Quantitative assessment of the sustainability of greenhouse gas mitigation strategies in the AFOLU sector at the global scale

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Évaluation quantitative de la durabilité de stratégies d'atténuation des émissions de gaz à effet de serre dans le secteur AFOLU à l'échelle mondiale

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Thèse de doctorat

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Résumé

L'implémentation à large échelle de stratégies d'atténuation des émissions dans le secteur de l'agriculture, la forêt et autres usages des sols (AFOLU) pose des questions sur la durabilité de ces stratégies. Par exemple, les bio-fuels de seconde génération menacent la biodiversité et la reforestation d'espaces agricoles augmente le prix de l'alimentation. De plus, ces stratégies d'atténuation des émissions dépendent fortement des conditions socio-économiques décrivant le reste du système alimentaire (libéralisation du commerce agricole, développement économique, augmentation de la population...). Dans cette thèse, nous cherchons à préciser les impacts sur la biodiversité, l'alimentation et les émissions de gaz à effet de serre de différentes stratégies d'atténuation à large échelle dans le secteur AFOLU au regard de différentes situations socio-économiques. Pour cela, nous utilisons la modélisation prospective qui nous permet de simuler des scénarios décrivant l'évolution de l'usage des sols à l'échelle mondiale à l'horizon 2030, 2050 et 2100. Le couplage du modèle d'usage des sols Nexus Land-Use (NLU) avec le modèle de biodiversité Projecting Responses of Ecological Diversity In Changing Terrestrial Systems (PREDICTS) permet d'étudier l'impact de ces stratégies d'atténuation sur différentes composantes de la biodiversité. Le calcul de bilan d'azote permet quant à lui de préciser le lien entre l'intensification et sont impact environnemental.

Dans la première partie du manuscrit de thèse, nous testons des scénarios d'augmentation de la production de légumineuses en Europe en évaluant les effets sur les émissions de gaz à effet de serre du secteur AFOLU. Nous montrons que le principal avantage environnemental des légumineuses est de fournir des protéines comme substitut aux produits d'origine animale plutôt que de permettre une réduction de la consommation d'engrais de synthèse par une fixation biologique accrue de l'azote. La majorité de la réduction d'émission a lieu dans le secteur de la production animal et hors de l'Europe. Notons également l'importance des mécanismes indirects qui mènent à une réduction des émissions de N2O associées à la fertilisation azotée dans le secteur végétal. La sensibilité de ces résultats à la combinaison du scénario de changement de régime alimentaire avec un scénario de reforestation nous amène à nous intéresser ensuite aux interactions entre stratégies d'atténuation.

Dans la seconde partie, nous étudions les compromis et les synergies entre conservation de la biodiversité et maintien de la sécurité alimentaire pour différents scénarios d'atténuation. La production à large échelle de bioénergie a des effets négatifs à la fois sur différents indicateurs de biodiversité (richesse spécifique et l'indicateur d'intégrité de la biodiversité) et sur la sécurité alimentaire (prix de l'alimentation et coût de production). Bien que présentant un compromis entre protection de la biodiversité et sécurité alimentaire, les combinaisons de changement de régime alimentaire et de scénario de reforestation permettent d'améliorer la conservation de la biodiversité et la sécurité alimentaire dans de nombreux cas par rapport à une situation sans atténuation des émissions.

Dans la troisième partie, nous comparons différentes évolutions de l'usage des sols à l'échelle mondiale en identifiant les scénarios qui permettent de ne pas dépasser les limites de la planète au regard d'indicateurs renseignant le cycle de l'azote, l'intégrité de la biosphère, les émissions de CO2 du secteur AFOLU et la conservation des forêts. Nous montrons que malgré l'incertitude régnant autour de la détermination des limites planétaires, les scénarios environnementaux qui permettent de rester de manière robuste au sein de ces limites planétaires sont constitués majoritairement de dynamique de reforestation, de changement de régime alimentaire et d'augmentation de l'efficacité de l'utilisation des intrants dans la production végétale.

Résumé étendu

Le secteur agricole et forestier est au centre de nombreux enjeux. Bien qu'en recul ces 50 dernières années, la lutte cotre la faim dans le monde reste d'actualité puisque le nombre de personnes sous-nourries s'est accru de 17 million en 2017. Le secteur agricole constitue le premier facteur de réduction de la biodiversité mondiale. Il est également responsable de 25% des émissions mondiales de gaz à effet de serre (GES). Enfin, si l'augmentation de la fertilisation minérale dans le secteur de la production végétale depuis 1961 (+120%) a permis de lutter contre la faim dans le monde, elle s'est faite au détriment de l'eutrophisation des écosystèmes naturels. Dans cette thèse, nous allons chercher à étudier les différentes facettes de la durabilité du système agricole et comment relever durablement les défis que posent les problèmes évoqués ci-dessus. La mondialisation du système alimentaire mondial s'accompagne d'une délocalissation de la production alimentaire et ses conséquences loin des zones de consommation et nécessite d'étudier les problèmes de durabilité du secteur agricole dans un large périmètre. 50 % de l'azote contenu dans la production alimentaire est contenu dans le commerce international. Une part importante des terres situées dans l'alimentation des habitants de pays comme les USA, l'Europe ou le Japon (respectivement 33%, 50% et 92%) sont produits dans d'autres pays, notamment d'Amérique du Sud comme le Brésil ou l'Argentine (avec respectivement 47% et 88% de la surface de culture destinée à l'exportation). Un cinquième de l'eau utilisée pour la production agricole entre 1996 et 2005 est destinée au commerce international. Dans cette thèse, nous nous plaçons à l'échelle mondiale pour étudier à la fois les impacts mondiaux de l'agriculture, mais également pour prendre en compte les causes distantes de cette dégradation de l'environnement par l'agriculture.

En particulier, nous allons nous intéresser aux stratégies d'atténuation des émissions de GES dans le secteur de l'agriculture, la forêt et autres usages des sols (AFOLU en anglais) à large échelle. Les stratégies proposées actuellement pour réduire les émissions sont des stratégies de changement de régime alimentaire (réduction de la consommation de ruminant dans la ration alimentaire...), de changement structurelle (ré-allocation géographique de la production dans des régions faiblement émettrices, intensification de l'élevage...) ou de déploiement de technologies faiblement émettrices pour produire notre alimentation (amélioration de la fertilisation, digests anaérobiques...). Pour étudier ces stratégies d'atténuations qui ne sont encore que très peu déployées à large échelle, la communauté scientifique utilise des scénarios décrivant l'évolution de l'usage des sols à l'échelle mondiale à l'horizon 2030, 2050 et 2100. Or la durabilité de ces stratégies d'atténuations proposées par la communauté de modélisation du climat et basées sur l'usage des sols pose actuellement

question. Par exemple, les bio-fuels de seconde génération menacent la biodiversité et la reforestation d'espaces agricoles augmente le prix de l'alimentation.

Parmi les différentes composantes du système agricole, l'intensification agricole joue un rôle clé dans la durabilité de la production agricole. Elle est en effet responsable de la majeure part de l'augmentation de la production connue pendant la révolution verte, mais également de la pollution azotée. L'intensification agricole consiste en l'augmentation de la production par unité de terre. L'intensification étudiée dans cette thèse est une intensification conventionnelle réalisée à l'aide d'intrants chimiques. Dans cette thèse, nous cherchons à préciser les impacts sur la biodiversité, l'alimentation et les émissions de gaz à effet de serre de différentes stratégies d'atténuation à large échelle dans le secteur AFOLU au regard de différentes situations socio-économiques. Et plus spécifiquement, nous étudierons comment l'intensification agricole influence la durabilité de ces stratégies.

Pour représenter les usages des sols à l'échelle mondiale, l'intensification agricole et les émissions de gaz à effet de serre associé à la production agricole, nous choisissons d'utiliser le modèle d'usage Nexus Land-Use (NLU). Ce modèle d'usage des sols est un modèle d'équilibre partiel technico-économique puisque chacune des 12 régions du monde minimise son coût de production sous contraintes techniques. Le secteur de la production animale représente à l'aide de coefficients techniques la conversion de consommation d'herbe, fourrages et concentrés par les ruminants en produits animaux. Le secteur de la production végétale représente la production d'une culture représentative en fonction de son rendement potentiel local et du niveau d'intensification. Le niveau d'intensification résulte de la minimisation du coût et est défini en fonction du prix des intrants et du prix de l'alimentation. Le modèle NLU utilise donc en entrée des données biophysique et calcule des indicateurs économiques comme présenté dans le schéma suivant :

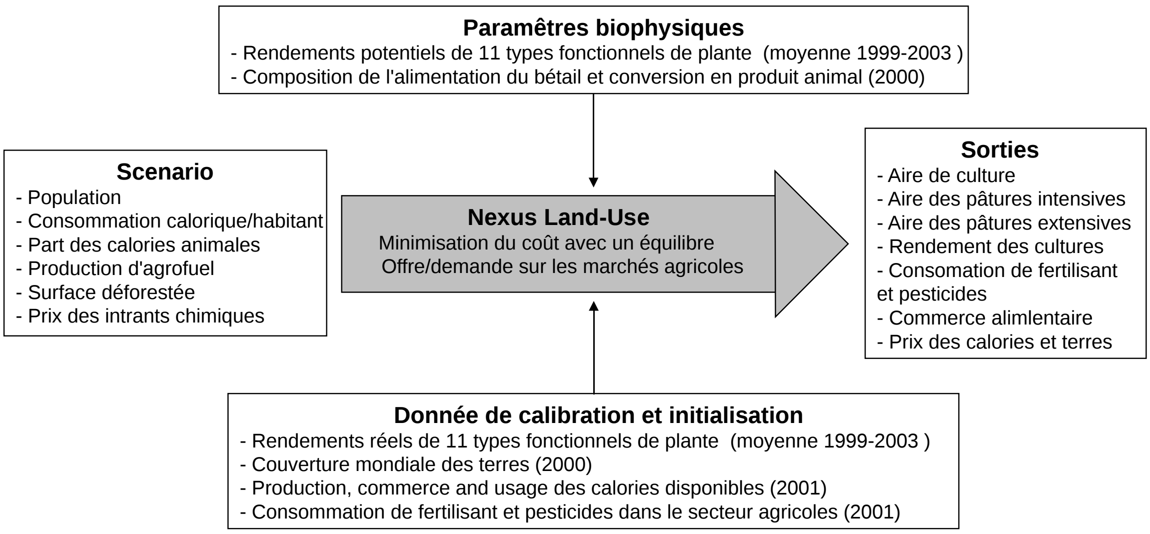


FIGURE 1 – Fonctionnement général du modèle Nexus Land-Use (NLU)

Dans la première partie du manuscrit de thèse, nous évaluons l'impact de scénarios d'augmentation de la production de légumineuse en Europe sur les émissions de gaz à effet de serre du secteur agricole et forestier. Pour cela, nous insérons dans le modèle NLU un calcul de bilan d'azote. Ces bilans d'azote consistent à calculer les différentes sources d'azote et en déduire les différents type d'azote en sortie du système à partir de l'équation suivante :

$$\mathrm{N}_{synth} + \mathrm{N}_{manure} + \mathrm{N}_{rot} + \mathrm{N}_{fix,l} + \mathrm{N}_{fix,o} + \mathrm{N}_{deposition} = \mathrm{N}_{récolté} + \mathrm{N}_{perdu} + \mathrm{N}_{laissé} \tag{1}$$

Avec $\mathrm{N}_{récolté}$ pour l'azote récolté, N_{perdu} pour l'azote perdu, $\mathrm{N}_{laissé}$ pour l'azote restant pour la rotation suivante, N_{synth} pour l'azote synthétique, N_{manure} pour l'azote du fumier, N_{rot} pour la quantité d'azote fournie par la rotation précédente, $\mathrm{N}_{fix,l}$ pour l'azote fixé biologiquement des légumineuses, $\mathrm{N}_{fix,o}$ pour l'azote fixé biologiquement des autres cultures,$\mathrm{N}_{deposition}$ l'azote provenant du dépôt atmosphérique. Par hypothèse, $\mathrm{N}_{laissé}$ correspond aux résidus de légumineuses, les autres résidus ne sont pas explicitement comptabilisés, dans l'hypothèse où ils annulent l'approvisionnement et l'utilisation.

Nous montrons que le principal avantage environnemental des légumineuses est de fournir des protéines comme substitut aux produits d'origine animale plutôt que de permettre une réduction de la consommation d'engrais de synthèse par une fixation accrue de l'azote par les légumineuses. Pour obtenir ce résultat, nous avons comparé un scénario de production de soja européen à destination de la consommation animale et un scénario de production de pois en substitution de la consommation de ruminant. Il apparaît que le scénario de substitution de consommation de ruminant par de la consommation de pois réduit les émissions de 211 MtCO2 alors que la production de soja réduit les émissions que de 10 MtCO2 de légumineuse. Nous montrons également une faible réduction d'émission permise par la substitution d'azote minérale par de l'azote biologiquement fixée dans le cas de la substitution d'un changement de régime alimentaire car cette réduction ne représente que 1 % de la réduction d'émission.

Nous montrons également que les émissions réduites par la production de légumineuse dépend en grande partie de l'usage des terres libérées par le changement de consommation. La substitution de la consommation de ruminant par des pois réduit en effet l'empreinte au sol et permet de libérer des terres. Ces terres peuvent alors être reforestées ou maintenues dans le système agricole sous forme de pâture ou de cultures. Dans le premier cas (reforestation des terres libérées), la reforestation représente 53% de la réduction d'émission, la réduction de la production animale 38% de la réduction animale et le reste de la réduction d'émission a lieu dans le secteur de la production végétale. Dans le second cas (terres maintenues dans le secteur agricole), 62% de la réduction d'émission a lieu dans le secteur de la production animale et le reste de réduction d'émission dans le secteur de la production végétale. Bien que la quantité d'émission réduite pour le même niveau de production de légumineuse soit similaire dans les deux scénarios (1%ou 0.5% pour respectivement le scénario avec et sans reforestation), le type d'émission atténué est fortement différent et les mécanismes à l'œuvre varient.

L'utilisation du modèle NLU mène à une expansion agricole sur les terres marginales de qualité moindre suivant la théorie ricardienne. La moindre expansion agricole en 2050 lié à un changement de régime alimentaire permet ainsi d'augmenter le rendement par hectare et de réduire la consommation d'intrant en concentrant la production sur les terres de meilleures qualités par rapport à la baseline en 2050. Nous présentons ici un exemple d'intensification durable de la production végétale.

La sensibilité de ces résultats à la combinaison du scénario de changement de régime alimentaire avec un scénario de reforestation nous amène à nous intéresser dans la suite aux interactions entre stratégies d'atténuation.

Dans la seconde partie, nous étudions les compromis et les synergies entre conservation de la biodiversité et maintien de la sécurité alimentaire pour différents scénarios d'atténuation. Pour évaluer les impacts des changements d'usage des sols associés à la mise en place de stratégies d'atténuation sur la biodiveristé, nous choisissons d'utiliser les modèles de biodiversité à effet mixte construit à partir de la base de donnée Projecting Responses of Ecological Diversity In Changing Terrestrial Systems (PREDICTS) couvrant 47 000 espèces dans le monde entier. Cette base de données permet d'évaluer l'impact sur différents indicateurs de biodiversité de différents usages des sols anthropique et de différents niveaux d'intensification de ces usages par rapport à un état de l'écosystème pristine. Dans ces modèle, la végétation (forêt et non forêt) secondaire, les cultures annuelles, pérennes et fixant biologiquement l'azote, les pâtures, les parcours et la zone urbaine constituent les différents usages des sols possibles. 3 niveaux d'intensité d'usage des sols sont associés à 3 types de culture (annuelles, pérennes et fixant l'azote), et 2 niveaux sont associés aux pâtures. Les indicateurs de biodiversité estimés dans ces modèles sont le nombre d'espèce présent dans le milieu, l'abondance d'individus et la composition des communautés écologiques. Les modèles associés à ces 3 indicateurs sont présentés dans le tableau suivant :

TABLEAU 1 – Coefficient des modèles mixtes de biodiversité estimé à partir de la base de donnée PRDICTS.

Usage des sols	Modèle d'abondance	Modèle de richesse spécifique	Modèle de similarité
Intercepte	0.65895	2.65435	2.189599
Végétation Secondaire	-0.01415	-0.15875	-0.223229
Parcours	-0.03463	-0.09300	-1.122190
Pâture à usage léger	-0.05364	-0.23153	-3.398944
Pâture à usage intense	-0.08303	-0.21634	-3.398944
Annuelle à usage minimum	-0.12289	-0.37063	-1.557422
Annuelle à usage léger	-0.11470	-0.47360	-1.557422
Annuelle à usage intense	-0.15255	-0.41606	-1.557422
Pérennes à usage minimum	0.02072	-0.21912	-0.294046
Pérennes à usage léger	-0.09749	-0.42456	-1.063739
Pérennes à usage intense	-0.06351	-0.51682	-1.801390
Fixatrice d'azote à usage minimum	-0.04453	-0.37003	-1.280273
Fixatrice d'azote à usage léger	-0.16470	-0.72871	-1.280273
Fixatrice d'azote à usage intense	-0.23775	-0.67512	-1.280273
Urbain à usage minimum	-0.01684	-0.15043	-1.319501
Urbain à usage léger	-0.10958	-0.34365	-1.319501
Urbain à usage intense	-0.15866	-0.39866	-1.319501

Pour faire le lien entre ces modèles de biodiversité et le modèle NLU, nous réalisons d'une part une spatialisation au point de grille du NLU et d'autre part nous calculons le lien entre les niveaux d'intensité d'usage des sols de PREDICTS (3 pour les cultures et 2 pour les pâtures) avec les classes d'intensité du modèle NLU (60 pour chacune des 12 régions pour les cultures et 2 par région pour les pâtures). La spatialisation au point de grille du NLU consiste à passer de distributions pour différentes qualités de la terre de différents usages des sols (culture,pâture et forêt) à la part de usage des sols au niveau d'un point de grille d'une carte terrestre mondiale. Dans ce cas, nous faisons le choix d'une règle simple qui consiste à répartir proportionnellement les changements calculés à l'échelle des distributions sur les classes de terre au niveau des points de grille. Pour faire le lien entre les intensité d'usage des sols de PREDICTS et du NLU, un modèle additif généralisé (GAM en anglais) est calculé à l'année de référence pour faire correspondre la proportion relative de terres cultivées minimales, légères et intenses avec les 60 classes de terres du NLU. Une GAM est utilisée pour éviter de faire des hypothèses sur la distribution des erreurs, pour lisser la relation entre les classes d'intensification des deux modèles et pour éviter de donner trop de poids dans la relation aux valeurs extrêmes incertaines. Cette relation n'est définie que pour les classes de terres ayant une superficie significative de terres. Pour les classes de terres extrêmes, les proportions des catégories intensives, légères et minimales sont définies constantes à la dernière valeur calculée (exemple des USA en Fig.2).

Dans l'année t, la classe de terre correspondant au rendement calculé par la NLU est liée aux 3 classes d'intensification PREDICTS par le GAM. Comme prévu, cette relation montre une augmentation des proportions relatives des classes PREDICTS les plus intensives avec l'augmentation de l'intensification dans les classes NLU.

La densité d'élevage des ruminants est utilisée comme lien entre l'intensité des pâturages du PREDICT et du NLU. Dans cette version du modèle, le changement de système n'a pas d'impact sur la biodiversité que par un changement d'usage des sols (mélange de terres cultivées et de pâturages pour le système ML et de pâturages pour le système pastoral uniquement) et non par un changement de l'intensité. Au cours de l'année de référence, la relation entre la proportion relative de pâturages légers et intenses et la densité du cheptel sur les cartes de la FAO est calculée en utilisant une GAM. Cette GAM est ensuite utilisée au cours de l'année t pour déterminer la proportion de pâturages légers et intenses à partir de la densité du bétail.

Dans cette partie, nous trouvons que la production à large échelle de bioénergie a des effets négatifs à la fois sur la biodiversité à travers différents indicateurs (richesse spécifique et língdicateur d'intégrité de la biodiversité) et sur la sécurité alimentaire (prix de lálimentation et coût de production) à l'échelle mondiale (Fig.2). Nous constatons qu'en axant l'atténuation sur une seule politique, on peut obtenir des résultats positifs pour un seul indicateur de la sécurité alimentaire ou de la conservation de la biodiversité, mais avec des effets secondaires négatifs importants sur les autres. Par exemple, les mesures d'atténuation dominées par le reboisement favorisent les critères de biodiversité, mais elles devraient entraîner une forte hausse des prix des denrées alimentaires. Un portefeuille équilibré des trois politiques d'atténuation, bien qu'il ne soit pas optimal pour un seul critère, minimise les compromis en évitant des effets négatifs importants sur la sécurité alimentaire et la conservation de la biodiversité.

A l'échelle régionale, le compromis entre biodiversité et sécurité alimentaire observé au niveau mondial est nuancé par les différents contextes régionaux. La combinaison de politiques d'atténua-

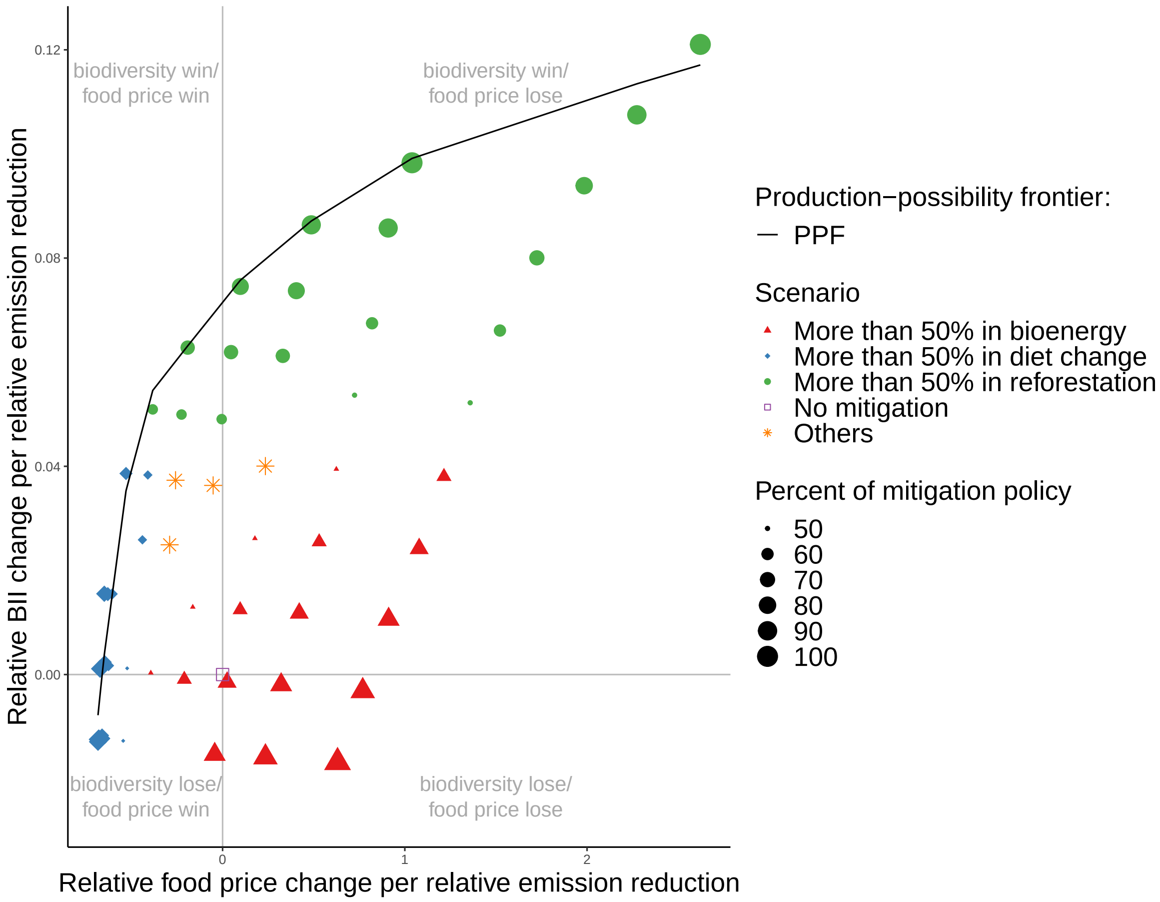


FIGURE 2 – Influence de différents mix d'atténuation sur le prix de lálimentation et sur le BII. Les résultats sont présentés comme le changement relatif de l'indice d'intégrité de la biodiversité (IIB) et du prix des aliments dans les scénarios d'atténuation du climat par rapport à la ligne de base sans aucune politique d'atténuation par changement relatif des émissions. A l'échelle mondiale, le changement relatif des émissions atténuées est constant à 0,3 dans tous les scénarios d'atténuation du climat par construction parce que les émissions de référence (13,9 GtCO2 en 2100) sont réduites de 4,3 GtCO2/an en 2100. Les variations relatives des IIB et des prix des denrées alimentaires peuvent être déduites de ce graphique en multipliant les valeurs obtenues par la variation relative des émissions pour chaque scénario, qui est constante à 0.3. Chaque scénario d'atténuation climatique est coloré en fonction de la politique d'atténuation dominante.La politique de réduction des émissions dominante dans chaque scénario est celle qui réduit le plus les émissions. Dans la légende, Autres représente les scénarios où aucune option ne représente plus de 50% de l'effort d'atténuation.Le pourcentage de la politique d'atténuation est la part des émissions atténuées de la politique d'atténuation dominante (production de biocarburants de deuxième génération, changement alimentaire et reboisement).

tion, qui maximise la biodiversité et la sécurité alimentaire, change donc d'une région à l'autre. Cependant, dans toute la région, la production élevée de bioénergie augmente le prix des aliments et réduit la biodiversité par rapport à un niveau de référence sans atténuation.

Dans la troisième partie, le but est de sélectionner les futurs possibles pour le secteur agricole qui permettent de rester au sein des limites planétaires. Les limites planétaires ont été définies à l'échelle mondiale comme des limites biophysique au-delà desquels des changements irréversibles peuvent avoir lieu. Ces changements irréversibles peuvent menacer la sûreté de l'espèce humaine. Dans ce chapitre, les limites planétaires retenus sont la préservation de 3.7 Gha de forêt, un niveau d'intégrité de la biosphère au-delà de 0.9, une atténuation des émissions en 2030 de 1 GtCO2 et des pertes mondiales d'azote dans le milieu en deçà de 62 Tg d'azote.

Les futures possibles pour le secteur agricole sont définis par l'ensemble de scénario combinant différents drivers : des niveaux de population, de demande alimentaire, de surface de forêt, d'efficacité de l'utilisation de l'azote, d'efficacité du secteur animale et d'ouverture au commerce. Les différentes valeurs prises par ces drivers sont résumées dans le tableau suivant :

TABLEAU 2 – Description des scénarios descriptifs des différents secteurs AFOLU possibles en 2030.

Scénarios	Bas	Moyen	élevé
Population (Mia de personnes)	8.10	8.37	8.61
Demande végétale (Mkcal/cap/year)	920	917	914
Demande animale (Mkcal/cap/year)	107	84	60
Surface de forêt	3568	3653	3730
Efficacité d'utilisation de l'azote	0.34	0.38	0.38
Productivité animale (kcal/kcal)	13	10	
Imports de produits végétaux (Tkcal)	2961	3305	3645

Pour sélectionner les futurs secteurs agricoles qui permettent de rester de manière robuste à l'intérieur des limite planétaires, nous utilisons l'analyse de cluster appelée "Scenario discovery". Cette méthode vise à caractériser les combinaisons de paramètres d'entrée incertaines ou drivers les plus prédictifs pour rester dans les limites planétaires. Contrairement aux approches exploratoires ou aux approches d'optimisation telles que la frontière possibilité-production, nous prenons explicitement en compte avec cette méthode l'incertitude entourant les conditions socio-économiques du secteur AFOLU et l'adoption d'autres politiques environnementales dans le choix de la stratégie robuste pour rester dans les limites planétaires. Par exemple, lorsqu'une politique de protection de l'environnement est adoptée, nous ne savons pas dans quelles conditions socio-économiques elle sera mise en œuvre. Les bénéfices environnementaux du déploiement d'une politique d'amélioration de l'efficacité de l'utilisation de l'azote peuvent donc être totalement compensés par une augmentation de la consommation de produits d'origine animale.

Pour définir les caractéristiques du secteur AFOLU qui restent de manière robuste à l'intérieur des limites planétaires, nous sélectionnons un ensemble de scénarios (aussi appelé boîte) à l'aide d'une méthode d'analyse de cluster appelé PRIM (Patient Rule Induction Method). La boîte est définie par les limites qui limitent l'ensemble des futures qui définissent les secteurs agricoles restant dans les limites planétaires. Par exemple, une boîte est un secteur de l'AFOLU avec une population comprise entre 8,37 et 8,61 milliards de personnes en 2030, qui consomme en moyenne 84 Mkcal/cap/an de produits animaux par an, avec un taux de reboisement élevé et une forte libéralisation des échanges. Pour mesurer la qualité de la boîte, deux indicateurs mesurent la capacité des secteurs sélectionnés à rester dans les limites planétaires : la couverture et la densité. La couverture mesure dans quelle mesure les scénarios définis par la boîte capturent complètement les secteurs agricoles et forestiers qui permettent de rester à l'intérieur des limites planétaires (également appel'es le Type I ou le faux négatif). La densité mesure la pureté de la boîte. Elle est exprimée comme la part de scénarios dans la boîte qui permet de rester dans les limites planétaires (appel'ee Type II ou faux positif).

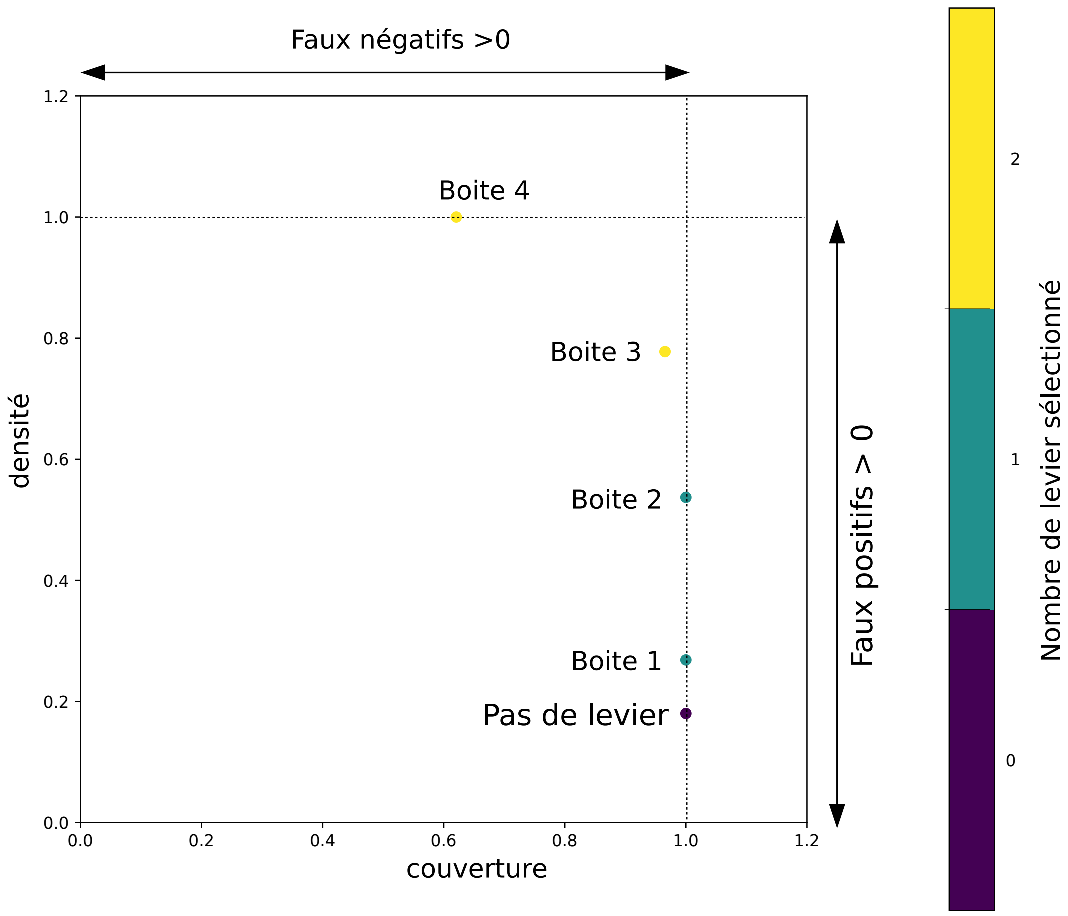
L'objectif de l'algorithme PRIM est de minimiser les incertitudes de type I et de type 2 pour définir les stratégies robustes dans le secteur AFOLU qui restent dans les limites planétaires. Pour ce faire, l'algorithme PRIM est un processus itératif, qui élimine les faces minces de l'espace d'entrée pour générer des régions plus petites contenant chacune une couverture et une densité moyennes plus élevées. La densité augmente avec le nombre d'effets de levier, car de moins en moins de stratégies d'aménagement du territoire sont mises en œuvre dans la boîte sélectionnée. Inversement, la couverture diminue avec la réduction de la taille de la boîte parce que le PRIM ne tient pas compte des stratégies d'utilisation des terres qui restent dans les limites des limites planétaires. Enfin, nous réalisons une analyse de sensibilité autour des limites planétaires définies précédemment.

Nous montrons que la combinaison d'un changement de régime alimentaire conséquent (réduction de la consommation de ruminant de par rapport à la baseline sans changement de régime alimentaire en 2030) et de reforestation (jusqu'à atteindre 3730 Mha) permet de rester de manière robuste au sein des limites planétaires définies précédemment. Cette combinaisons permet de limiter les faux positifs le plus possible (densité de 1 pour la boite 4 dans la figure 5).

L'analyse de sensibilité montre que le choix des seuils modifie la sélection des secteurs agricoles et forestiers durables. Plus on est exigeant en terme de réduction de l'azote perdu et en terme de réduction des émissions non-CO2, plus on sélectionne des stratégies d'augmentation de l'efficacité de l'utilisation de l'azote.

Pour conclure, la principale contribution de cette thèse est de prendre en compte dans un même cadre de modélisation (i) l'impact des différentes couvertures terrestres (forêts, pâturages et terres cultivées) et des différentes intensités d'occupation des sols des pâturages et des terres cultivées sur la biodiversité, (ii) le calcul des bilans azotés, (iii) le calcul des émissions de GES par source selon la méthode IPCC ; (iv) un modèle de production agricole issue des secteurs végétaux et animaux. Grâce à l'intégration de ces différents éléments, nous étudions commet le secteur AFOLU influence les compromis entre ces quatre objectifs de développement durable : SDG 2 (Faim zéro), SDG 6 (Eau et assainissement), SDG 13 (Changement climatique), SDG 15 (Biodiversité, forêts, désertification).

Dans le premier chapitre,nous étudions les compromis entre réduction des émissions CO2 et les émissions nonCO2 lors d'un changement de régime alimentaire. Nous montrons en effet que l'utilisation des terres économisées par le changement de régime alimentaire détermine fortement le type d'émission atténuée. Dans le cas où les terres sont utilisées pour de la reforestation, le carbone est principalement stocké dans le sol et dans le cas où les terres sont maintenues dans le système



Boite	Levier	Min	Max	Couverture	Densité
1	Forêt	0.05	0.2	1	0.26
2	Forêt	0.15	0.2	1	0.54
3	Forêt	0.15	0.2	0.97	0.78
	Diet	0.25	1		
4	Forêt	0.15	0.2	0.62	1
	Diet	0.75	1		

FIGURE 3 – Arbitrage entre la couverture et la densité des stratégies pour rester à l'intérieur des limites planétaires dans la minimisation de l'incertitude en 2030.

agricole, les émissions sont réduites principalement dans le secteur de la production animale.

Dans le second chapitre, nous étudions le compromis entre sécurité alimentaire et biodiversité lors du déploiement de stratégies d'atténuation basée sur l'usage des sols. Dans ce cas, nous montrons que le déploiement de forte quantité de biofuel produite à la place de pâtures est néfaste à la fois à la biodiversité et à la sécurité alimentaire. Un portfolio de plusieurs stratégies permet de limiter ces compromis et il est même possible de réduire les émissions de 4.3 GtCO2/an en 2100 tout en augmentant le biodiversité et réduisant le prix de l'alimentation par rapport à une baseline sans atténuation.

Enfin dans le dernier chapitre, nous étudions les compromis entre le SDG 2, le SDG 6, le SDG 13 et le SDG 15 dans le secteur agricole et forestier. Nous montrons quún secteur agricole et forestier qui reforeste et augmente son efficacité de l'utilisation de l'azote accompagné d'un changement de demande permet de rester au sein des limites planétaires.

Abstract

The large-scale implementation of emission reduction strategies in the agriculture, forestry and other land uses (AFOLU) sector raises questions about their sustainability. For example, second-generation bio-fuels threaten biodiversity and the reforestation of agricultural land increases food prices. In addition, these emission reduction strategies are highly dependent on socio-economic conditions describing the rest of the food system (agricultural trade liberalization, economic development, population growth, etc.). For example, an increase in food demand, due to population growth and economic development, can increase pressures on the food system, leading to ecosystem degradation and increased greenhouse gas emissions.

In this thesis, we seek to clarify the impacts on biodiversity, food and greenhouse gas emission of large-scale mitigation strategies in the AFOLU sector under different socio-economic conditions. To do this, we used prospective modeling to simulate various global land uses in 2030, 2050 and 2100 under different scenarios. More specifically, to study the impact of different mitigation strategies on biodiversity indicators, we coupled the Nexus Land-Use (NLU) model with the Projecting Responses of Ecological Diversity In Changing Terrestrial Systems (PREDICTS) biodiversity model. A nitrogen balance is also built to specify the link between intensification and environmental impact.

In the first chapter, we assessed the impact of scenarios of increased legume production in Europe on greenhouse gas emissions in the AFOLU sector. We found that the main environmental benefit of legumes is to provide proteins as a substitute for animal products rather than enabling a lower consumption of synthetic fertilizer through the increased leguminous nitrogen fixation. Most of the emission reduction takes place in the animal production sector and outside Europe. This first chapter also highlights the importance of indirect mechanisms that lead to a reduction in N2O emissions associated with nitrogen fertilization in the plant sector. The sensitivity of these results to different reforestation scenario led me to then focus on the interactions between mitigation strategies.

In the second chapter, we analyzed the trade-offs and synergies between biodiversity and food security for different combinations of mitigation scenarios. Large-scale bioenergy production had negative effects on different biodiversity indicators (species richness and biodiversity intactness index) as well as on different food security indicators (food prices and production costs). Although presenting a trade-off between biodiversity protection and food security, a combination of diet change and reforestation scenarios can improve biodiversity and food security in many cases compared to a situation without mitigation.

In the third chapter, we identified global land-use scenarios that ensure to stay within planetary boundaries in terms of nitrogen cycle, biosphere integrity, non-CO2 emissions from the AFOLU sector and forest conservation. We showed that despite the uncertainty surrounding the determination of global boundaries, the most robust environmental scenarios that ensure to stay within these global boundaries are mainly composed of reforestation, dietary changes and increased efficiency in the use of inputs in crop production.

Contents

List of figures

List of Tables

Introduction

« *All happy families are alike ; each unhappy family is unhappy in its own way.* »

Leo Tolstoy

« *All domestic animals are alike ; each non-domestic animals is non-domestic in its own way.* »

Jared Diamond

« *All sustainable societies are alike ; each unsustainable society is unsustainable in its own way.* »

This Thesis

3.3 million hectares were deforested annually between 2010 and 2015 (Keenan et al. 2015) ; greenhouse gas emissions increased from 339 to 397 ppm between 1981 and 2014 ; the species extinction rate is 100 to 1000 times higher than the extinction rate observed in the past (Pimm et al. 2014) ; the use of mineral nitrogen in agriculture increased by 120% between 1961 and 2008 (Galloway et al. 2008). This global environmental damage has led the international community to implement global environmental protection policies. In the first part of the introduction, I will explain the choice of the global scale to study current environmental problems ; in the second part, I will present the main questioning of the thesis on the interactions between environmental protection strategies, land use change and sustainability indicators ; and I will end this introduction by raising our thesis problematic.

The growing influence of environmental policies on a global scale

The global dimension of environmental problems

Since the beginning of the 20th century, the effect of humankind on the earth environment has reached unprecedented levels. The specificity of this period is the abundance of inter-connections in a globalized socio-economic system, the increasing role of global ecological and economic mechanisms and the existence of international institutions to address certain aspects of global environmental protection (Donges et al. 2017).

Inter-connections within the global socio-economic system are firstly due to a disconnection between the locations of production and consumption that are connected to each other through international trade. This leads to a geographical separation of causes and environmental impacts (Liu et al. 2013). This disconnection can be exacerbated by global dynamics specific to natural systems such as climatic, hydrological or ecological dynamics which connect geographically distant elements of the earth system (Kitzberger et al. 2007).

In the framework of the Driver-Pressure-State-Impacts-Response (DPSIR) (Kristensen 2004), socio-economic drivers (population, diet, economic development...) are translated into distant pressures. For example, the increase in the urban population has impacts on land use in areas that produce food for these urban centers (Seto et al. 2012).

In addition, global ecological dynamics are becoming increasingly important in ecological dynamics in general with climate change proving to be a growing driver of ecosystem health (Newbold 2018) and ecosystem homogenization representing a growing threat to biodiversity (Millennium Ecosystem Assessment Board 2005, Newbold et al. 2016, 2018, Kuemmerle et al. 2013). In the case of climate change, the distribution of damage is not related to the source of emissions but rather to the redistribution of effects by the dynamics of the climate system and the ability of individuals to adapt (Tol et al. 2004). Human impacts on biodiversity depends on scales (Chase et al. 2018) and overall, climate directly impacts biodiversity (Bellard et al. 2012, Thomas et al. 2004).

The expansion of global environmental policies

The first Earth Summit, organized by the United Nations in Stockholm in 1972, and the creation of the United Nations Environment Program (UNEP), were the first to address global environmental problems in international institutions. Subsequently, conventions specific to the various environmental problems were adopted.

In 1992, at the Earth Summit in Rio, the United Nations Framework Convention on Climate Change (UNFCCC) was signed by 154 states. This convention, whose supreme body is the Confederation of Parties (COP), commits the signatory countries to fight climate change. As an extension of this convention, the Kyoto Protocol was signed in 1997 by 184 countries that committed themselves between 1995 and 2015 to reduce emissions by at least 5% compared to 1990 levels by 2008-2012. To pursue the effort to reduce emissions after 2015, an agreement was reached at the COP21 in Paris between 195 countries. Signatory countries have since committed to reduce their emissions according to the intentions stated in the national contributions (INDC). In parallel with these international climate negotiations, a Convention on Biological Diversity (CBD) was created in 1992. This convention on the protection of biodiversity was then implemented through National Biodiversity Strategies, notably in France in 2004. In 2010, Aichi's global objectives were adopted and in April 2012, the Intergovernmental Platform on the Protection of Biodiversity and Ecosystem Services (IPBES) was created to provide multidisciplinary expertise on biodiversity and ecosystem service issues to the 194 CBD signatory countries.

In 2015, global environmental policies were integrated into a broader framework of sustainability of socio-economic and natural systems through the Sustainable Development Goals (SDGs), which followed Agenda 21 by setting 17 targets by 2030. These objectives include environmental objectives such as objectives 6 (clean water), 7 (clean energy), 12 (responsible consumption and production), 13 (measures to combat climate change), 14 (aquatic life) and 15 (terrestrial life). But sustainability also includes a social dimension, as evidenced by many other sustainable development objectives. There has therefore been a change in the approach to environmental policy by trying to integrate the achievement of several objectives simultaneously because of the many interactions between environmental and socio-economic objectives. I will try to take into account in this thesis the influence of these environmental policies on the social dimensions of sustainability, including the quantification of food security.

Description of the environmental policies studied in this thesis

During this thesis, since I were not able to study all the environmental policies corresponding to the environmental SDGs mentioned above, I chose to focus on 4 policies : reducing greenhouse gas emissions from the Agriculture, Forestry and Other Land Use (AFOLU) sector, conserving biodiversity, protecting forests and combating nitrogen runoff in surface waters, which present interesting relationships. Indeed, nitrogen cycle and biodiversity are linked by the negative effect of nitrogen deposition on land (Alkemade et al. 2009). The protection of forests has a beneficial effect on biodiversity because of the exceptional ecological value of forests (Watson et al. 2018). More generally, the link between biodiversity, nitrogen use through nitrogen fertilization and forest protection is formalized in the conceptual framework of land-sharing/land-sparing (Phalan 2018). These policies were also chosen because of the current high pressures on the nitrogen cycle and biosphere integrity, as evidenced by the crossing of the boundaries associated with these two dimensions defining the safe space for human societies (Steffen et al. 2015).

Challenges in the global assessments of land-use based scenario of GHG reduction in the AFOLU sector

Until recently, among the global environmental protection strategies, the literature has focused on the assessment of the impacts of global GHG mitigation strategies in the AFOLU sector on food security and biodiversity. We present here an overview of this literature.

Assessing the impacts of land-use based scenario of GHG reduction in the AFOLU sector on biodiversity

As mentioned above, the integrity of the biosphere is currently one of the most threatened global boundaries. This biodiversity loss is largely due to the significant changes in land use that have taken place over the past century (Foley et al. 2005). These land use changes consist of both an increase in the area under cultivation, that currently covers a quarter of the planet (Ramankutty et al. 2008), an increase in the intensification of these uses as evidenced by the increase in global nitrogen flow that has doubled since 1960 (Galloway et al. 2008) and an increase in irrigation which represents 70% of global water withdrawals (Millennium Ecosystem Assessment Board 2005). This pressure on land use is expected to increase with increasing involvement for land use in climate policies (Obersteiner et al. 2016). In particular, in the scenarios developed in the AR5 (IPCC 2014) and in the IPCC 1.5° report (IPCC 2018), the IPCC highlighted the significant increase in the production of second generation biofuels in the second half of the 20th century which represents a risk for water management (Bonsch et al. 2016) and biodiversity (Hill et al. 2018, Heck, Gerten, Lucht & Popp 2018).

Although the direct impacts of biofuel production are taken into account, the impacts of intensification on biodiversity are often neglected in most of these studies (Kehoe et al. 2017).

Assessing the impacts of land-use based scenario of environmental protection on food price

The large-scale deployment of emission mitigation strategies in the AFOLU sector impacts the price of food through : (i) the use of land for other purposes than food production land use for other purposes than food production (Humpenöder et al. 2018, Heck, Gerten, Lucht & Popp 2018), (ii) the reduction of land availability through the protection of carbon sink ecosystems like forests (Kreidenweis et al. 2016, Stevanović et al. 2017), (iii) a shift towards less GHG-intensive agricultural commodities (Popp et al. 2010, Stehfest et al. 2009), and (iv) the adoption of GHG-efficient management practices in the livestock sector (Havlík et al. 2014) or the crop sector (Frank et al. 2018).

The increase in food insecurity due to the large-scale implementation of GHG mitigation strategies in the AFOLU sector must be compared to a baseline where climate change also has impacts on food prices through reduced yields (Hasegawa et al. 2018). Thus, scenarios of a uniform carbon tax leads to an increase of the number of people at risk of hunger by 78 million in the SSP2+RCP2.6 scenario, corresponding to a carbon tax of 65$ compared to the 24 million in the case of the SSP2/RCP 6.0 corresponding to a carbon tax of 2$ (Hasegawa et al. 2018). For a 1.5°, potential of people undernourished can raise to 260 million people (Frank et al. 2017).

However, there are many solutions to limit the negative effects of these GHG mitigation strategies in the AFOLU sector. First of all, the redistribution of the income from the uniform carbon tax can compensate the regions suffering the most from hunger (Hasegawa et al. 2018).

The explicit study of interactions between several emission mitigation strategies to reduce negative impacts on food prices is only very recent. A combinaison of forest and water protection, improved nitrogen use efficiency, and agricultural intensification next to large-scale bioenergy production allows to reduce the impacts of climate policies in the AFOLU sector on food prices (Humpenöder et al. 2018).

Some land-use based climate policies can have synergistic effects with food and nutritional security (Popp et al. 2010, Stehfest et al. 2009). For example, taxes on red meat and dairy products are expected to cut emissions and improve nutritional health. Targeting land-use change hotspots can be a way to reduce emissions by minimizing food security impacts and avoiding emissions leakages. The sequestration on agricultural land (Frank et al. 2017) could lower carbon prices and costs in terms of calorie decrease.

Modeling interactions between the natural system and the socio-economic system

To study the effects of different environmental protection strategies on a global scale, I used prospective modelling. Prospective modelling is an approach that aims to represent possible futures using scenarios and models to provide public policy actors with decision-making tools. Unlike forecasting, prospective modelling proposes a systemic diagnosis, in long-term exercises, integrating disruptions such as the implementation of a specific environmental policy. While models allow different elements of reality to be grouped together in a coherent framework, scenarios make it possible to either fix the uncertainty around certain elements of the model, or to explicitly represent the evolution of certain aspects of particular interest. In this thesis, I will focus on this type of modeling.

Modelling interactions between society and the climate system

When tackling environmental problems, it is important to take into account natural dynamics and to confront them with the dynamics of the socio-economic system. To do this, integrated models articulate different models that each have their own dynamics. The integrated models used in the reports of the International Panel on Climate Change (IPCC) are the GCAM (Calvin et al. 2019), AIM (Fujimori et al. 2014), IAM IIASA framework (Havlík et al. 2011, Amann et al. 2011, Messner & Schrattenholzer 2000, Meinshausen et al. 2011), REMIND-MagPIE (Luderer et al. 2015, Popp et al. 2014), WITCH (Bosetti et al. 2011) and IMAGE (Bouwman et al. 2006). These models combine top-down economic models to represent the links between the different sectors of the economy, with bottom-up sectoral models to represent the dynamics within a sector. These models can be integrated with climate models to take into account climate dynamics or simply use climate model results in their simulations.

Modeling the socio-economic system of the AFOLU sector through land-use [1] models

Although they affect the economy as a whole, the problems I have chosen to address in this thesis concern above all the AFOLU sector. In the case of nutrient cycle management issues, agriculture is the main nitrogen emitter in the environment through mineral nitrogen fertilization of crops and massive intensification in some regions of the world (Galloway et al. 2008). The nitrogen cycle is also disrupted through the trade of agricultural raw materials that separates production sites from consumption sites (Galloway et al. 2008). For biodiversity, one of the main threats are the changes in land-use and in intensity of land-use due to agriculture (Foley et al. 2005). For climate, the AFOLU sector was responsible for 25% of anthropogenic greenhouse gas emissions in 2005 (Tubiello et al. 2015) and represents a mitigation potential through the carbon sinks it exploits : soil and vegetation. I will therefore focus on this sector, which incorporates many anthropogenic and natural dynamics.

Land use models integrate several human activities and associated ecological dynamics in the same framework (Millennium Ecosystem Assessment Board 2005, Newbold et al. 2016, 2018, Verburg et al. 2002). In this thesis, I used the land use model called NLU (Souty et al. 2012, Brunelle et al. 2015).

The NLU has a global scope that explicitly takes into account tele-coupling mechanisms around the world (Liu et al. 2013), unlike regional models (Jayet et al. 2018, Chakir & Le Gallo 2013). A representative crop is used, so the representation of trade is less detailed than models that represent the markets of each culture such as GCAM (Calvin et al. 2019), but the NLU allows to represent crop and livestock intensification processes which is important for environmental impacts assessment. The NLU is not well suited for the study of redistributive effects of national or regional agricultural policies, for instance taxes on input or on agricultural products, as agent heterogeneity is not very detailed, regional models such as AROPAJ (Jayet et al. 2018) are more suitable for such a task.

The NLU is not only an economic model, it also include environmental characteristics with plant growth models (Souty et al. 2012) and scenarios of climate change impacts on yields (Müller & Robertson 2014). In this thesis, I added nitrogen balances which estimate the natural nitrogen fertilization of crops through biological fixation of legumes and deposition. The dynamic of climate, water cycle and phosphorus cycle are still not represented in NLU as in an earth dynamic model (Calvin et al. 2019).

Also, the use of linear programming in NLU makes it possible to bring together in the same modelling framework economic mechanisms subject to technical constraints (Souty et al. 2012). Unlike statistical models such as the CLUE-S model (Verburg et al. 2002) or a spatial econometrics model (Chakir & Le Gallo 2013), NLU represents relatively well the long-term processes and impacts of public policy on the agricultural system. The use of an explicit production function with nitrogen balances makes it possible to take into account many mechanisms, such as crop fertilization from the manure of the livestock production sector, which cannot be taken into account by a price-induced

1. Land use is the combination of a land-cover or type of land use (forest, crops, pastures, city...) with its intensity (use of inputs on cropland, amount of timber extracted from the forest, livestock stocking per hectare...). A land-use change therefore consists of a change in the level of fertilization of a crop or the conversion of a forest into a crop. I will try to specify as often as possible the type of land-use change that is considered

intensification function.

Finally, to a lesser extent than the integration of the AFOLU sector into the GCAM model (Calvin et al. 2019), the coupling of the NLU with the general equilibrium model IMACLIM (Waisman et al. 2012) still makes it possible to understand certain cross-sectoral links, such as the impact on energy prices of biofuel production, in the study of climate change mitigation policies.

To summarize, the use of the NLU therefore presents the advantage of a precise representation of agricultural intensification, a modelling framework that integrates technical constraints among economic choices and a world coverage of the AFOLU sector. It also integrates many aspects of the land-use theory in the agricultural sector. This last point is described in the method section. On the other hand, it has the disadvantage of having a rough representation of the forest, international trade and inter-temporal dynamics. The land use models can be classified more exhaustively according to 8 sources of variations (Briassoulis 2000) : the objective, the theory underlying the interactions between land-uses, the level of description, the sector represented, the temporal dynamics and the type of modelling. I describe in Table. 3.15, presented in appendix, 5 land-use models according to these 7 criteria and describe where the NLU stands in this classification.

Modelling the human-ecosystem link through biodiversity and ecosystem service models

While the importance of biodiversity for human activities has been documented in several reports (Millennium Ecosystem Assessment Board 2005, TEEB 2010, FAO 2019), biodiversity and ecosystem services are still rarely integrated in global assessments. In the different conceptual frameworks describing the link between ecosystems and human systems, the focus can be set on the ecological processes (translated into biodiversity models) or on the interactions between human and ecosystems (translated into models of ecosystem services). To define which of these two types of model to use, the conceptual framework of Societal Determinants, Pressure, State, Impact and Response (DPSIR) can be useful. Biodiversity models represent the drivers, pressures and state of biodiversity. Thus, they focus on the natural system by detailing the pressure-state relationship. Ecosystem service models represent drivers, pressures, states and impacts on human societies and therefore focus more on the link between nature and human society.

Due to its multiple aspects, biodiversity can be described using different approaches (functional ecology, taxonomic ecology or phylogenic ecology) and at different scales (species, community and ecosystem). Thus, the choice of indicators skews the assessment of biodiversity towards one of these aspects.

Each species responds differently to different types of land-use (Newbold et al. 2014), therefore the impacts of a land-use on biodiversity cannot be obtained by generalizing the results of a species-specific study. Nonetheless, species-specific studies improve predictions by taking into account species-specific responses to anthropogenic pressures (Visconti et al. 2016, Merow et al. 2013, Thuiller et al. 2009).

Focusing on communities increases the scope of species-specific studies but presents challenges in accounting for community composition changes, in their response to anthropogenic pressures and in scale effects. A first approach combines the use of species-specific responses with population dynamics to take into account landscape effects (Martins & Pereira 2017, Chaudhary et al. 2015). This approach has the advantage of considering scale effects and population composition effects. As the species-specific models, the scope of these models is depending on the number of community represented in the models. Another approach is to no longer focus on the evolution of a species on a territory over time but to look at the state of communities in different land uses at some point in time. This approach allows local observations to be generalized to large scales using land use data (a detailled description of this approach is provided in Purvis et al. (2018) and De Palma et al. (2018)). In this framework, anthropogenic pressures are an aggregate of different anthropogenic pressures such as, in the case of crops, use of chemical inputs, field size, crop diversity and level of mechanization (Hudson et al. 2014). A final possible approach is to define the impacts of anthropogenic pressures from a meta-analysis linking biodiversity (here through the "mean species abundance" indicator) to specific anthropogenic pressures such as the distance to the closest human infrastructure, nitrogen deposition, fragmentation, and climate change (Alkemade et al. 2009). That kind of statistical model allows to take into account multiple indicators, but is not well suited to a long term assessment of the link between biodiversity and people activity as the processes are not explicit at all.

A more recent path considers the study of entire ecosystems by taking into account the relationships between individuals, the dynamics of species, their relationship with the environment etc., and defining groups according to their biological traits (Harfoot et al. 2014). The development of models on this scale is very recent and at the beginning of this thesis did not yet have a version advanced enough to be used on a global scale.

Another focus can be set on the interaction between ecosystems and humans using models such as Dynamic global vegetation models (Sitch et al. 2003, Haverd et al. 2018) or ecosystem services models (Sharp et al. 2016, Alkemade et al. 2009, Guerra et al. 2016, Martínez-López et al. 2019, Jackson et al. 2013).

In this thesis, I chose to emphasize the link between people and ecosystems by choosing a biodiversity model because nature's feedback on people is still in its early stages on a global scale with very recent advances (Dainese et al. 2019). Moreover, since the purpose of this thesis is to study environmental strategies, I chose the PREDICTS model which has the advantage of representing rather accurately the influence of different anthropogenic pressure intensities (Purvis et al. 2018). Overall, the coupling of the NLU, a model of agricultural intensification, and the PREDICTS model appears as a relevant association to study the impact of environmental management strategies on biodiversity. However, I did not represent the link between climate and biodiversity. .

General problem and plan

My main goal was to investigate the influence of the combination of environmental protection strategies in the AFOLU sector on its global sustainability.

To address this issue, I will assume that the environmental protection strategies that maximize the global sustainability of the AFOLU sectors can be evaluated using a set of sustainability indicators and by considering the land-use changes due to these strategies.

To clarify the methodology used to test this hypothesis, I will first describe the chosen modelling framework : the coupling between the NLU global land-use model and the PREDICTS biodiversity model. In this section I will also describe the land-use theories used to explain the land-use change mechanisms at work in the NLU.

In the chapter. 1, I will study how indirect effects influence the greenhouse gas reduction performance of a strategy to increase legume production in Europe. In this first part, I first focused on a single environmental protection strategy and studied the influence of these indirect effects on the environmental performance of an indicator. Then, I investigated the effects of a combination of two scenarios, dietary change and a reforestation scenario, on the reduction of GHG emissions. The sensitivity of the dietary change effect on the reduction of GHG emissions led us to focus on the combination of different strategies for protecting the environment.

In the chapter. 2, I will explore how the interaction between second-generation biofuel production, reforestation of pastures and dietary change reduces the trade-off between food security and biodiversity conservation in a context of reduced emissions in the AFOLU sector. This section will illustrate how mitigation strategies interact and will clarify the relationships between different environmental sustainability indicators. This study led us to investigate the robustness of this conclusion by comparing it with different possible futures for the AFOLU sector and different sustainability indicators.

In the chapter. 3, I will determine levels for land use drivers that allow us to stay within some planetary boundaries. This section will cover a wide range of scenarios and provide information on many indicators, in order to understand the relative importance of socio-economic drivers in the sustainability of land use and to deepen the links between different environmental indicators.

Finally, I will conclude with the main findings of this thesis and describe some perspectives relative to the work initiated with this thesis.

This thesis is based on a collection of articles. The three chapters following the methodology descriptions are articles that can also be red independently of the whole manuscript. The method section and some elements of the conclusion provide a cross-cutting analysis across all chapters.

Modelling framework

In this chapter, we describe the modeling framework used in the chapters of this thesis to avoid redundant presentations of some parts of the model common to the different chapters. In a first part, we describe the version of NLU before the beginning of this thesis. In the second part, we describe the contributions of this thesis to the modelling framework. More precisely, we describe (i) the nitrogen balances introduced into NLU, (ii) the calculation of GHG emissions from the AFOLU sector in NLU and (iii) the coupling between NLU and the PREDICTS biodiversity model.

NLU model

Here, we provide a general description of the version of NLU before this thesis. More details can be found in Souty et al. (2012) and Brunelle et al. (2015).

General description

NLU model is a partial equilibrium model in which the agricultural sector is divided into 12 regions of the world (Fig. 4), inter-connected with each other by international trade.

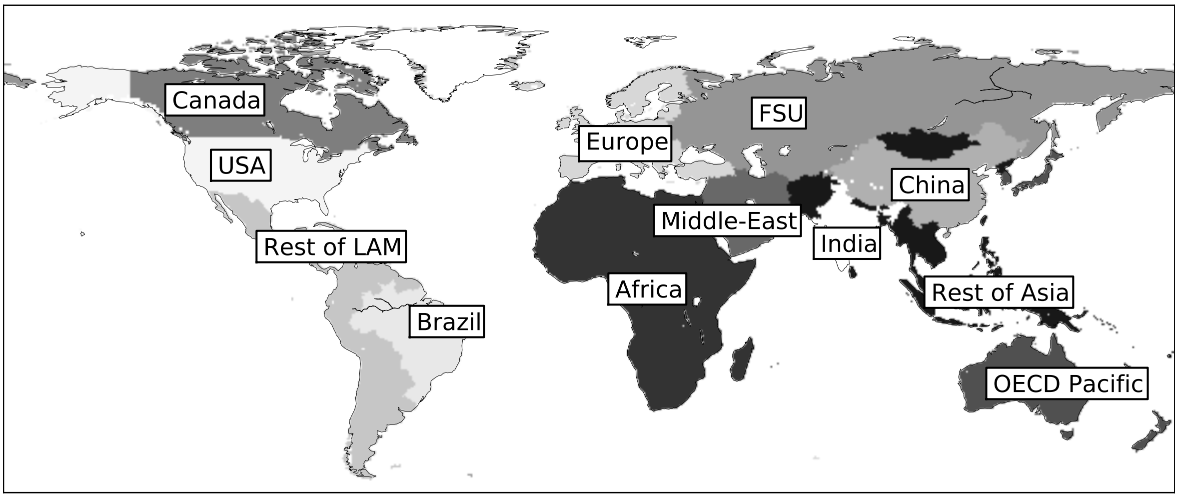


Figure 4 – Map of the 12 regions as defined in NLU

Model inputs are scenarios of population, diet, agrofuel production, deforestation rate and fertilizer prices and its outputs are cropland area, mixed crop-livestock system area, pastoral area, crop yield, fertilizer consumption, land price and calorie price (See Fig. 5). NLU provides a simple representation of the main processes of agricultural intensification for crop and livestock production: the substitution between (i) land and fertiliser for the crop sector and (ii) grass, food crops, residues and fodder for the livestock sector. It does so by minimising the total production cost under a supply-use equilibrium for food and biofuel markets. A detailed description can be found in Brunelle et al. (2015).

The NLU model simulates changes in the agricultural sector at the global scale (food price, land rent, profit, crop yield and cropland as a percentage of total agricultural land) with a non-linear response of yield to fertilizer prices, as well as an explicit representation of livestock systems and international trade. For the base year, a representative potential yield is computed on a $0.5° \times 0.5°$ grid from the potential yields given by the vegetation model LPJmL for 11 crop functional types (CFT).

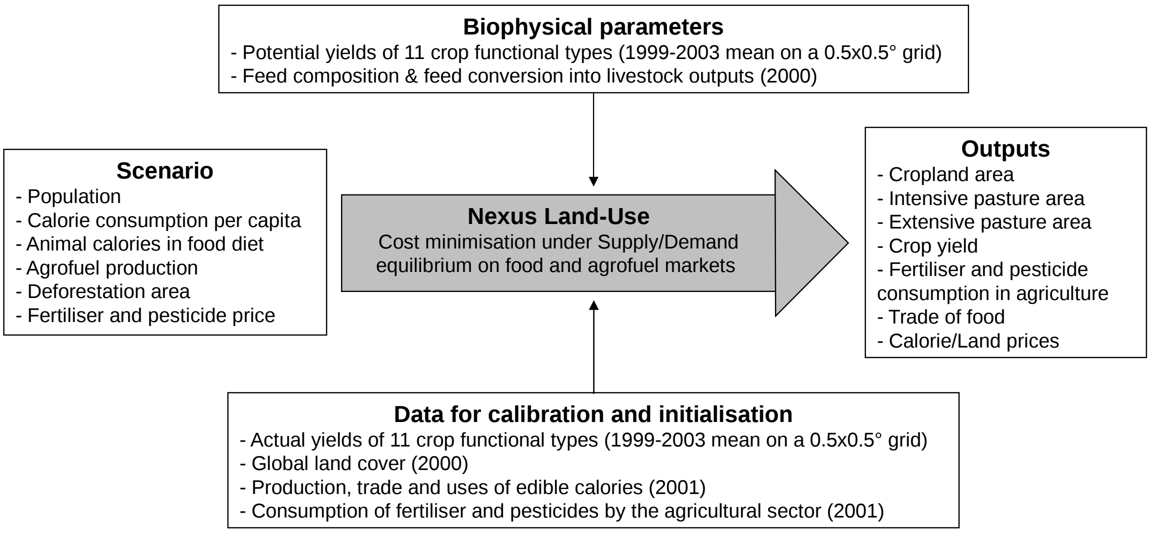


Figure 5 – Description of the modelling system in NLU.

Land classes are set up that group together grid points with the same potential yield. Yield in each land class is dynamically determined by a fertilizer function for the 11 CFT (hereafter referred to as dynamic crops). This function asymptotes toward the potential yield and is characterised by decreasing returns. In each land class, consumption of chemical inputs and associated yield are determined by cost minimization under the constraint of a global supply-demand balance for plant food (Eq. 4) and ruminant calories (Eqs. 5-8) and a land constraint (Eq. 10).

Variables and indices

Indices

j	Land class number.
j_{limit}	Limit land class between the mixed crop-livestock and the pastoral production systems.
j_{max}	Index of the highest land class.

Parameters in each region

ω^{fc}_{swo}, ω^{m}_{swof}, ω^{r}_{swof}	Ratio of Seed, Waste at the farm level, Other uses of food crops (excluding Feed) in total production of Food Crop, Monogastric and Ruminant products.
$Q^{fc}_{\text{other crop}}$	*Other* production of food crops which is not dynamically modelled (i.e. difference between the total production from Agribiom and LPJmL production in 2001).
α_{IC}	Initial slope of the intermediate consumption function in $\$\,\text{kcal}^{-1}$.

$\mathrm{FC_{tot}}$	Fixed cost per hectare in \$ $\mathrm{ha^{-1}\,yr^{-1}}$ corresponding to capital, labour, business services, pesticides and energy consumption for vehicles, buildings (heating, etc.) and other on-farm operations (drying of crops, etc.). Recalibrated to account for the costs of the mixed crop-livestock system and the pastoral systems.
$\rho^{grass}_{past,int}$, $\rho^{grass}_{past,ext}$	Grazed grass per hectare of pastures in the mixed crop-livestock and pastoral systems in $\mathrm{kcal\,ha^{-1}\,yr^{-1}}$.
$\rho^{r,int}_{past}$, $\rho^{r,ext}_{past}$	Production of ruminant product per hectare of pasture in the mixed crop-livestock and pastoral systems in $\mathrm{kcal\,ha^{-1}\,yr^{-1}}$ ($\rho^{r,int/ext}_{past} = \frac{\rho^{grass}_{past,int/ext}}{\beta_{r,int/ext}\phi^{grass}_{r,int/ext}}$).
$\mathrm{Imp^m}$, $\mathrm{Exp^m}$	2001 imports and exports of monogastric products in $\mathrm{kcal\,yr^{-1}}$.
ρ^{max}_j, ρ^{min}_j	Potential yield and minimum (no inputs) yield in $\mathrm{kcal\,ha^{-1}\,yr^{-1}}$.
β_m, $\beta_{r,int}$, $\beta_{r,ext}$	Feed conversion factor for monogastrics, ruminants from the mixed crop-livestock and the pastoral systems in kcal of feed/kcal of animal product.
ϕ^{fc}_m, ϕ^{fodder}_m, $\phi^{fc}_{r,int}$, $\phi^{fodder}_{r,int}$, $\phi^{grass}_{r,int}$, $\phi^{grass}_{r,ext}$	Share of feed categories in animal rations (fc: food crops, fodder: residues and fodder, grass: pasture grass, monog: monogastrics, r,int: ruminants from the mixed crop-livestock system, r,ext: ruminants from the pastoral system).

World level variables

p^w_{cal}	World calorie price in \$ $\mathrm{kcal^{-1}}$ (endogenous).
p_χ	Index of fertilizer and pesticide price (exogenous).

Exogenous regional variables

D^{fc}_h, D^m_h, D^r_h	Demand of food crops (fc), monogastrics (m) and ruminants (r) products for humans (h) in $\mathrm{kcal\,yr^{-1}}$.
$D^{fc}_{agrofuel}$	Demand of food crops for agrofuel production in $\mathrm{kcal\,yr^{-1}}$.
S_{surf}	Supply of agricultural area excluding *other* croplands, including *dynamic* croplands, residual pastures and pastures from the crop-livestock and pastoral systems in ha.

Endogenous regional variables in each land class

ρ_j	Yield of the land class j minimizing farmer's production cost in $\mathrm{kcal\,ha^{-1}\,yr^{-1}}$.
IC_j	Intermediate consumption of chemical and mineral inputs of the land class j in \$ $\mathrm{ha^{-1}\,yr^{-1}}$.
f^{crop}_j, f^{Pint}_j, f^{Pres}_j, f^{Pext}_j	Area of *dynamic* cropland (i.e. where crops modelled in the LPJmL model are grown), pastures from the crop-livestock system, residual pastures, pastures of the pastoral system of the land class j expressed as a fraction of D_{surf}.

Endogenous regional variables

p_{cal}	Food crop calorie price in $\$\,\text{kcal}^{-1}$.
λ	Land rent in $\$\,\text{ha}^{-1}\,\text{yr}^{-1}$.
p_{r}	Price of ruminant calories in $\$\,\text{kcal}^{-1}$.
D_{surf}	Demand of agricultural area excluding *other* croplands, including *dynamic* croplands, pastures from the crop-livestock system, residual pastures and pastures of the pastoral system in ha.
$Q_{\text{r,int}}$, $Q_{\text{r,ext}}$	production ruminant from the crop-livestock system and the pastoral system in $\text{kcal}\,\text{yr}^{-1}$.
D^{fc}_{m}, $D^{\text{fc}}_{\text{r,int}}$	Demand of food crops for monogastrics and ruminant production from the crop-livestock system in $\text{kcal}\,\text{yr}^{-1}$.
D^{fc}	Total demand of food crops in $\text{kcal}\,\text{yr}^{-1}$.
Imp^{fc}, Exp^{fc}	Imports and exports of food crops in $\text{kcal}\,\text{yr}^{-1}$.
Imp^{r}, Exp^{r}	Imports and exports of ruminant products in $\text{kcal}\,\text{yr}^{-1}$.

Regional optimization programm

Yield-fertilizer function:

$$\text{IC}_j(\rho_j) = \alpha_{\text{IC}}(\rho_j^{\max} - \rho_j^{\min})\left(\frac{\rho_j^{\max} - \rho_j^{\min}}{\rho_j^{\max} - \rho_j} - 1\right) \tag{2}$$

Objective function: Cost minimization of total production costs in each region:

$$\underset{\substack{\rho_j, j_{\text{limit}}, D^{\text{fc}}_{\text{r,int}} \\ Q_{\text{r,int}}, Q_{\text{r,ext}}, D_{\text{surf}}}}{\text{Min}} \left(\int_{j_{\text{limit}}}^{j_{\max}} (p_\chi \text{IC}_j(\rho_j) + \text{FC}_{\text{tot}}) f_j^{\text{crop}} \, \text{d}j\right) D_{\text{surf}} \tag{3}$$

Regional constraints:

$$Q^{fc}_{other} + \int_{j_{limit}}^{j_{max}} f^{crop}_j \rho_j \, dj \, D_{surf} =$$

$$(D^{fc}_{r,int} + D^{fc}_h + D^{fc}_m + D^{fc}_{agro} + Exp^{fc} - Imp^{fc})(1 + \omega^{fc}_{swo}) \quad (4)$$

$$D^r_h + Exp^r - Imp^r = Q_{r,int} + Q_{r,ext} \quad (5)$$

$$D^m_h + Exp^m - Imp^m = Q_m \quad (6)$$

$$Q_{r,ext} = \left(\int_0^{j_{limit}} f^{Pext}_j \, dj + \int_{j_{limit}}^{j_{max}} f^{Pres}_j \, dj \right) \rho^{r,ext}_{past} D_{surf} \quad (7)$$

$$Q_{r,int} = \frac{D^{fc}_{r,int}}{\beta_{r,int} \phi^{fc}_{r,int}} \quad (8)$$

$$Q_m = \frac{D^{fc}_m}{\beta_m \phi^{fc}_m} \quad (9)$$

$$S_{surf} = D_{surf} \quad (10)$$

The constraint on food crop production (Eq. 4) is associated with the Lagrangian multiplier interpreted as the calorie price p_{cal}. The constraints on total ruminant production (Eq. 5), ruminant production from the pastoral system (Eq. 7) and ruminant production from the mixed crop-livestock system (Eq. 8) are associated with Lagrangian multipliers that are all equal and can be interpreted as the ruminant price p_r. The constraints on monogastric production (Eq. 6 and 9) are associated with Lagrangian multipliers that are all equal and can be interpreted as the ruminant price p_m. Finally, the land constraint (Eq. 10) is associated with the Lagrangian multiplier interpreted as the land rent λ.

First order conditions yields:

$$p_{cal} = p_\chi IC'_j(\rho_j)(1 + \omega^{fc}_{swo}) \quad (11)$$

$$p_r = p_{cal}(1 + \omega^r_{swo}) \beta_{r,int} \phi^{fc}_{r,int} \quad (12)$$

$$p_m = p_{cal}(1 + \omega^m_{swo}) \beta_m \phi^{fc}_m \quad (13)$$

$$p_r f^{Pext}_{j_{limit}} \rho^{r,ext}_{past} =$$

$$(p_{cal} \rho_{j_{limit}} - p_\chi IC_{j_{limit}}(\rho_{j_{limit}}) - FC_{tot}) f^{crop}_{j_{limit}} + p_r f^{Pres}_{j_{limit}} \rho^{r,ext}_{past} \quad (14)$$

$$\lambda = p_{cal} \int_{j_{limit}}^{j_{max}} f^{crop}_j \rho_j \, dj - \int_{j_{limit}}^{j_{max}} (p_\chi IC_j(\rho_j) + FC_{tot}) f^{crop}_j \, dj \ldots \quad (15)$$

$$\ldots + p_r \left(\int_0^{j_{limit}} f^{Pext}_j \, dj + \int_{j_{limit}}^{j_{max}} f^{Pres}_j \, dj \right) \rho^{r,ext}_{past} \quad (16)$$

The land rent λ is the sum of the scarcity rent, denoted μ, and the differential rent, denoted δ, defined as following:

$$\mu = p_{cal} f^{crop}_{j_{limit}} \rho_{j_{limit}} - (p_\chi IC_{j_{limit}}(\rho_{j_{limit}}) + FC_{tot}) f^{crop}_{j_{limit}} + p_r f^{Pres}_{j_{limit}} \rho^{r,ext}_{past} \quad (17)$$

$$\delta = \lambda - \mu \quad (18)$$

In the following we present the novelty brought by this thesis to NLU.

Adaptation of the modelling framework for each chapter

Chapter 1: Adding the computation of GHG emissions of the AFOLU sector and the nitrogen balance

Nitrogen balance, synthetic nitrogen use and nutrient cost

To represent emission changes associated with nitrogen leaching and fertilizer emissions as a result of the increase in legume consumption, we incorporated a nitrogen balance into the NLU model based on Zhang et al. (2015). This nitrogen balance represents the different sources and outputs of nitrogen in the crop system. Legume production influences nitrogen balance through different mechanisms: (i) an increase in biologically fixed nitrogen (BFN), (ii) a decrease in synthetic nitrogen fertilization, (iii) an increase in harvested nitrogen per calorie where legumes are introduced, and (iv) an increase in nitrogen contained in residues.

The following nitrogen balance is used:

$$\mathrm{IC}_{\mathrm{N}}^{\mathrm{T}} = \mathrm{N}_{synth} + \mathrm{N}_{manure} + \mathrm{N}_{rot} + \mathrm{N}_{fix,l} + \mathrm{N}_{fix,o} + \mathrm{N}_{deposition} \quad (19)$$

$$\mathrm{IC}_{\mathrm{N}}^{\mathrm{T}} = \mathrm{N}_{harvest} + \mathrm{N}_{lost} + \mathrm{N}_{left} \quad (20)$$

With $\mathrm{N}_{harvest}$ for harvested nitrogen, N_{lost} for lost nitrogen, N_{left} for nitrogen left for the next rotation, N_{synth} for synthetic nitrogen, N_{manure} for manure nitrogen, N_{rot} for nitrogen quantity provided by the previous rotation, $\mathrm{N}_{fix,l}$ for legumes biologically fixed nitrogen, $\mathrm{N}_{fix,o}$ for other crops biologically fixed nitrogen, $\mathrm{N}_{deposition}$ the nitrogen coming from the atmospheric deposition and $\mathrm{IC}_{\mathrm{N}}^{\mathrm{T}}$ the total supply of nitrogen, also equal to the total use. By assumption, N_{left} corresponds to legumes residues, other residues are not explicitly accounted for, under the assumption that they cancel out in supply and use.

Harvested nitrogen, nitrogen biologically fixed by legumes, and nitrogen left by legumes usable in the next rotation are set proportional to the energy yield ρ:

$$\mathrm{N}_{fix,l} = \rho \times \alpha_{\mathrm{N}}^{fix,l} \quad (21)$$

$$\mathrm{N}_{harvest} = \rho \times \alpha_{\mathrm{N}}^{harvest} \quad (22)$$

$$\mathrm{N}_{left} = \rho \times (\alpha_{\mathrm{N}}^{fix} - \alpha_{\mathrm{N}}^{harvest,legumes}) \quad (23)$$

with $\alpha_{\mathrm{N}}^{harvest}$ the harvested nitrogen per calorie produced and $\alpha_{\mathrm{N}}^{fix}$ the nitrogen fixed by legumes per calorie produced. These coefficients are obtained by aggregating crops based on crop coefficients from Zhang et al. (2015) for harvested nitrogen, Herridge et al. (2008) for fixed nitrogen, and FAO (2001) for crop energy content.

N_{dep}, $\mathrm{N}_{fix,o}$ and N_{manure} are modelled as constant rates per hectare in a given region, using Zhang et al. (2015) coefficients for N_{dep} and $\mathrm{N}_{fix,o}$. Annual changes in manure nitrogen are set based on the nitrogen applied from the monogastric and ruminant intensive systems. The nitrogen of legumes remaining in residues is left for the next rotation, $N_{rot} = N_{left}$.

To link crop yield to the different sources of nitrogen, phosphate, and potassium, we use an explicit production function in the NLU:

$$IC^T(\rho) = \alpha_{IC}^T \rho^{max}\left(\left(\frac{\rho^{max}}{\rho^{max}-\rho}\right)-1\right) \tag{24}$$

with $IC^T(\rho)$ the total nutrient requirement to reach the actual yield ρ, α_{IC}^T the original slope and ρ^{max} the potential yield.

Total nitrogen (N), phosphorus and potassium (PK) requirements are a share of the total nutrient requirement, with shares α_N and α_{PK}, $\alpha_N + \alpha_{PK} = 1$. These shares are assumed to be independent of yield level, thus making the nutrients complementary, with an assumption that the efficiency change is the same for both nutrient types along the production function. We deduce total nitrogen (IC_N^T) and total phosphorus and potassium (IC_{PK}^T):

$$\begin{aligned} IC_N^T(\rho) &= \alpha_N IC^T(\rho) \\ IC_{PK}^T(\rho) &= \alpha_{PK} IC^T(\rho) \end{aligned}$$

Synthetic nitrogen is determined based on the total nitrogen requirement minus other supply sources, as described by the nitrogen supply balance (19) (with $N_{synth} \equiv IC_N$, i.e., synthetic nitrogen in the supply balance is equal to fertilizer demand in the cost function). For the PK balance, a free source PK_f (corresponding especially to rock weathering) is considered in addition to mineral fertilizer. Synthetic nitrogen and mineral PK are introduced into the agricultural sector, while the other sources are either free renewable resources or agricultural co-products.

$$\begin{aligned} IC_N(\rho) &= \alpha_N IC^T(\rho) - (N_{manure} + N_{fix,o} + N_{dep}) - \alpha_{fix}^N \rho \\ IC_{PK}(\rho) &= \alpha_{PK} IC^T(\rho) - PK_f \end{aligned}$$

with $IC_N(\rho)$ the synthetic nitrogen, $IC_{PK}(\rho)$ the mineral P and K.

The price of nitrogen and the price of phosphate and potassium are calculated using the methodology described in Brunelle et al. (2015) Total nutrient cost $CI_{NPK}(\rho)$ for a given ρ is thus:

$$\begin{aligned} CI_{NPK}(\rho) &= p_N IC_N(\rho) + p_{PK} IC_{PK}(\rho) \\ &= (\alpha_N p_N + \alpha_{PK} p_{PK}) IC^T(\rho) \\ &\quad - p_N\left((N_{manure} + N_{fix,o} + N_{dep}) + \alpha_{fix}^N \rho\right) - p_{PK} PK_f \end{aligned} \tag{25}$$

The intensification level is determined by a microeconomic criterion of equality of marginal cost and marginal benefit:

$$
\begin{aligned}
p_{cal} &= ((\alpha_{\mathrm{N}} p_{\mathrm{N}} + \alpha_{\mathrm{PK}} p_{\mathrm{PK}})\mathrm{IC}^{\mathrm{T}\prime}(\rho) - p_{\mathrm{N}}\alpha^{\mathrm{N}}_{fix})(1+\omega^{fc}_{swo}) && (26)\\
\rho_j &= \mathrm{IC}^{\mathrm{T}\prime -1}\left(\frac{p_{cal} + (1+\omega^{fc}_{swo})p_{\mathrm{N}}\alpha^{\mathrm{N}}_{fix}}{(1+\omega^{fc}_{swo})(\alpha_{\mathrm{N}} p_{\mathrm{N}} + \alpha_{\mathrm{PK}} p_{\mathrm{PK}})}\right) && (27)
\end{aligned}
$$

with p_{cal} which is the calorie price, $\mathrm{IC}'^{\mathrm{T}}$ the marginal consumption of inputs and ω^{fc}_{swo} the rate of plant food production used for seeds, lost as waste, or used for other purposes.

Computation of GHG emissions by NLU

GHG emissions are estimated using an emission factor linking an emission source to its equivalent emission value following the method of the Intergovernmental Panel on Climate Change (IPCC) (see Table 4 in Supplementary Material for detailed emission factors).

Direct emissions from manure and synthetic nitrogen application ($\mathrm{N_2O}_{soil}$), indirect emissions from manure and synthetic nitrogen application ($\mathrm{N_2O}_{indirect,s}$) and indirect emissions from the leguminous crops residues ($\mathrm{N_2O}_{indirect,legumes}$) are computed based on the method of IPCC (2006*a*):

$$
\begin{aligned}
\mathrm{N_2O}_{soil} &= \mathrm{EF}_{soil} \times (\mathrm{N}_{synth} + \mathrm{N}_{manure}) && (28)\\
\mathrm{N_2O}_{indirect,s} &= (\mathrm{F}_{leach} \times \mathrm{EF}_{leach} + \mathrm{F}_{volat} \times \mathrm{EF}_{volat,s}) \times \mathrm{N}_s \qquad \forall s \in \{synth, manure\} && (29)\\
\mathrm{N_2O}_{indirect,legumes} &= \mathrm{F}_{leach} \times \mathrm{EF}_{leach} \times \mathrm{N}_{left} && (30)
\end{aligned}
$$

with EF_{leach} the emission factor of N_2O per N leached, F_{leach} the share of leached nitrogen, F_{volat} and $\mathrm{EF}_{volat,s}$ similar coefficients for volatilization, all set to the IPCC coefficients values. EF_{soil} is the emission factor per nitrogen fertilizer (organic or mineral) applied.

The IPCC methodology proposes the use of an explicit term for N in residues and a separate term for applied synthetic and manure nitrogen for direct N_2O emissions, with zero emissions when no nitrogen is applied. Other approaches use a non-zero intercept, because N_2O from applied nitrogen cannot be distinguished from N in previously applied nitrogen found in the soil and crop residues (Stehfest & Bouwman 2006, Van Groenigen et al. 2010). Similarly to the IPCC methodology, we consider that there are no emissions when no nitrogen is applied, but we use the factors of Stehfest & Bouwman (2006), intercept e_i and slope e_s, to calibrate the emission factor, including the effect of both applied nitrogen and residual nitrogen from previously applied nitrogen, using the application rate of developed countries where nitrogen in soil mostly involves previously applied nitrogen :

$$
\begin{aligned}
\mathrm{N}^{ref}_{applied} &= \sum(\mathrm{N}_{synth,\mathrm{DevCountries}} + \mathrm{N}_{manure,\mathrm{DevCountries}})\\
\mathrm{N_2O}^{ref}_{\mathrm{DevCountries}} &= e_i + e_s \mathrm{N}^{ref}_{applied}\\
\mathrm{EF}_{soil} &= \frac{\mathrm{N_2O}^{ref}_{\mathrm{DevCountries}}}{\mathrm{N}^{ref}_{applied}}
\end{aligned}
$$

We also take into account emissions from land-use changes (LUC) due to the conversion of one land cover (cropland, pasture, or forest) to another, the methane $(\mathrm{CH_4})$ emissions from enteric fermentation, the nitrous oxide $(\mathrm{N_2O})$ and methane emissions from manure management, the nitrous oxide emissions from pasture fertilization, and the methane emissions from rice cultivation (see Table 4 in supplementary material for detailed emission factors). Non-CO_2 emissions are converted into equivalent CO_2 emissions using the Global Warning Potential calculated for 100 years.

For livestock emissions, tier 2 emissions factors are computed following IPCC (2006*a*), based on feed requirements from Bouwman et al. (2005). The share of each product reported in FAOSTAT at the country level is used for every livestock sector for the food crop and byproduct category and the animal products category in Bouwman et al. (2005). For these feed types FAO composition coefficients are used to determine nitrogen and digestible energy content (FAO 2001). When such coefficients are missing and for the other categories of feed, feedipedia composition coefficients are used (INRA and CIRAD and AFZ and FAO 2015). For crop residues and fodder crops the quantity of fodder crops and the share of fodder products are taken from Monfreda et al. (2008). Nitrogen retention factors from the table in IPCC (2006*a*) are used and are not recomputed. Manure management system shares from IPCC (2006*a*) are used, selecting preferentially pasture range and paddock manure management system for ruminants pastoral systems.

Table 4 – Emissions factors as represented in NLU and variables used to compute emissions in Europe.

Emissions		Emissions factor	Emissions factors unit	Emissions factors source	Total emissions in 2001 (tCO_{2eq})
Land-use change	Pasture to cropland	0.68	$tCO_{2,eq}$/ha	Le Quéré et al. (2009)	2.7×10e9
	Forest to pasture	419.2	$tCO_{2,eq}$/ha	Le Quéré et al. (2009)	
	Forest to cropland	420.6	$tCO_{2,eq}$/ha	Le Quéré et al. (2009)	
Nitrous emissions					
from enteric fermentation	Extensive system	[1]	Tier 2 IPCC (2006*a*)	1.6×10e8	
from manure	Extensive system	-	$tCO_{2,eq}$/Mkcal	Tier 2 IPCC (2006*a*)	
from enteric fermentation	Intensive system	[1]	$tCO_{2,eq}$/Mkcal	Tier 2 IPCC (2006*a*)	1.6×10e8
from manure	Intensive system	[1]	$tCO_{2,eq}$/Mkcal	Tier 2 IPCC (2006*a*)	
Methane emissions					
from enteric fermentation	Extensive system	[1]	$tCO_{2,eq}$/Mkcal	Tier 2 IPCC (2006*a*)	5.6×10e8
from manure	Extensive system	[1]	$tCO_{2,eq}$/Mkcal	Tier 2 IPCC (2006*a*)	
from enteric fermentation	Intensive system	[1]	$tCO_{2,eq}$/Mkcal	Tier 2 IPCC (2006*a*)	24.6×10e8
from manure	Intensive system	[1]	$tCO_{2,eq}$/Mkcal	Tier 2 IPCC (2006*a*)	
Direct nitrous emissions from crop fertilization		0.006	$tCO_{2,eq}$/kg N_{fert}	Stehfest & Bouwman (2006)	1.2×10e9
Indirect nitrous emissions from mineral fertilization		0.0014	$tCO_{2,eq}$/kg N_{fert}	IPCC (2006*a*)	6.68×10e8
Indirect nitrous emissions from manure		0.0018	$tCO_{2,eq}$/kg N_{fert}	IPCC (2006*a*)	6.68×10e8
Nitrous emissions from legumes residues		0.011	$tCO_{2,eq}$/kg N_{fert}	IPCC (2006*a*)	2.6×10e7
Emissions from rice cultivation		0.047	$tCO_{2,eq}$/ha	Yan et al. (2009)	5.416740×10e6
Nitrous emissions	Intensive system	0.0067	$tCO_{2,eq}$/ha	IPCC (2006*a*)	4.4×10e7
from pasture fertilization	Extensive system	0	$tCO_{2,eq}$/ha	IPCC (2006*a*)	0

[1] regional emissions factors values are presented in Table. 5

Table 5 – Regional emissions factors for livestock production. Here manure management emissions factors and entheric fermentation are aggregated per animal type (ruminant/monogastric), emission type (N_2O/CH_4) and system type (Intensive/Pastoral/mixed). Here emissions factors are expressed in $tCO_2.Mkcal^{-1}$.

Regions	Nitrous emissions			Methane emissions		
	Monogastric	Ruminant		Monogastric	Ruminant	
	-	Pastoral	Mixed	-	Pastoral	Mixed
USA	0.108	2.055	0.981	0.687	6.470	3.660
Canada	0.103	2.403	1.265	0.937	7.510	4.441
Europe	0.124	1.055	0.383	0.630	3.606	2.873
FSU	0.283	1.646	0.822	0.355	5.872	3.771
OECD Pacific	0.146	1.0553	0.307	0.375	3.606	2.831
China	0.339	3.637	1.620	0.283	13.444	7.171
India	0.570	2.172	0.202	0.488	7.926	6.018
Brazil	0.506	5.224	1.810	0.187	16.718	8.315
Middle-East	0.683	1.3078	0.708	0.119	4.583	3.370
Africa	0.546	5.575	1.795	0.224	18.640	8.298
Rest of Asia	0.443	3.447	0.526	0.642	15.197	5.477
Rest of LAM	0.548	4.515	1.411	0.192	14.735	6.185

Chapter 2: Coupling of NLU and the PREDICTS models

Presentation of the PREDICTS framework

The PREDICTS project (Projecting Responses of Ecological Diversity In Changing Terrestrial Systems) built a data base of abundance and occurrence data for over 50,000 species and over 30,000 sites in nearly 100 countries from hundreds of published biodiversity comparisons of sites facing different land-use and related pressures (Hudson et al. 2017). The purpose of this survey is to represent animal, plant, and fungal diversity by avoiding geographic bias with the overrepresentation of economically developped and accessible regions and taxonomic biases with overepresentation of vertebrate. The data-base focuses on local biodiversity because most of the ecosystem services are provided locally.

The specificity of the PREDICTS approaches are "the space-for-time" assumptions and collating raw data rather than results-based meta-analysis (Purvis et al. 2018). These assumptions are discussed by De Palma et al. (2018) and we try to summarize it in the following.

The method to estimate the effect of land-use on a biodiversity indicator which is used in the PREDICTS project is to measure biodiversity in different land-uses in a landscape in the same time. By making this "space-for-time" assumptions, the biodiversity measure depends explicitly of the land-uses present in the region of interest. This option present the inconvenient to not take into account dynamics (De Palma et al. 2018) and "diffuse pressures acting across the whole landscape" (Purvis et al. 2018). This approach is therefore not well suited to measure impacts of climate change on biodiversity. To understand the impact of land-use changes, another method would be to measure in one location a biodiversity indicator under two different land-uses. To up-scale the results, the chosen sites for the biodiversity measure have to be representative of the region of interest. At global scale, the diversity of landscape, climate, human pressure... become impossible to cover and the need of data extremely big.

The second specificity of the PREDICTS project is to collate raw data rather than do a results-based meta-analysis (Purvis et al. 2018). A meta-analysis use a single indicator as response variable. Due to the different aspects of biodiversity, we can be interested by different biodiversity metrics that a meta-analysis can not deal with. The PREDICTS database (Hudson et al. 2017) can be used to calculate the α (inside a land-use) or β (along a pressure gradient) taxon-diversity. Crossed with phylogenitic tree and functionnal trait base, this data-base can also be used to compute phylogenetic measures of diversity or functional diversity.

With the special design, the PREDICT project allows, therefore, to improve the global biodiversity models, indicators, and projections of land-use impacts on biodiversity.

Reasons for coupling PREDICTS and NLU

NLU model explicitly specifies agricultural intensification. Spatial heterogeneity is taken into account by representing land of different quality on the basis of its potential yield. This local heterogeneity does not prevent the model from having a global representation of interactions between regions of the world using international trade. NLU model also represents two livestock systems with different intensities: the mixed crop-livestock system and pastoral system with grass-fed livestock (Bouwman et al. 2005). NLU's focus on agricultural intensification led us to use the PREDICTS model because of its ability to represent the impact of intensification of agricultural land-uses on biodiversity which is still missing from global modeling of land-use impact on biodiversity.

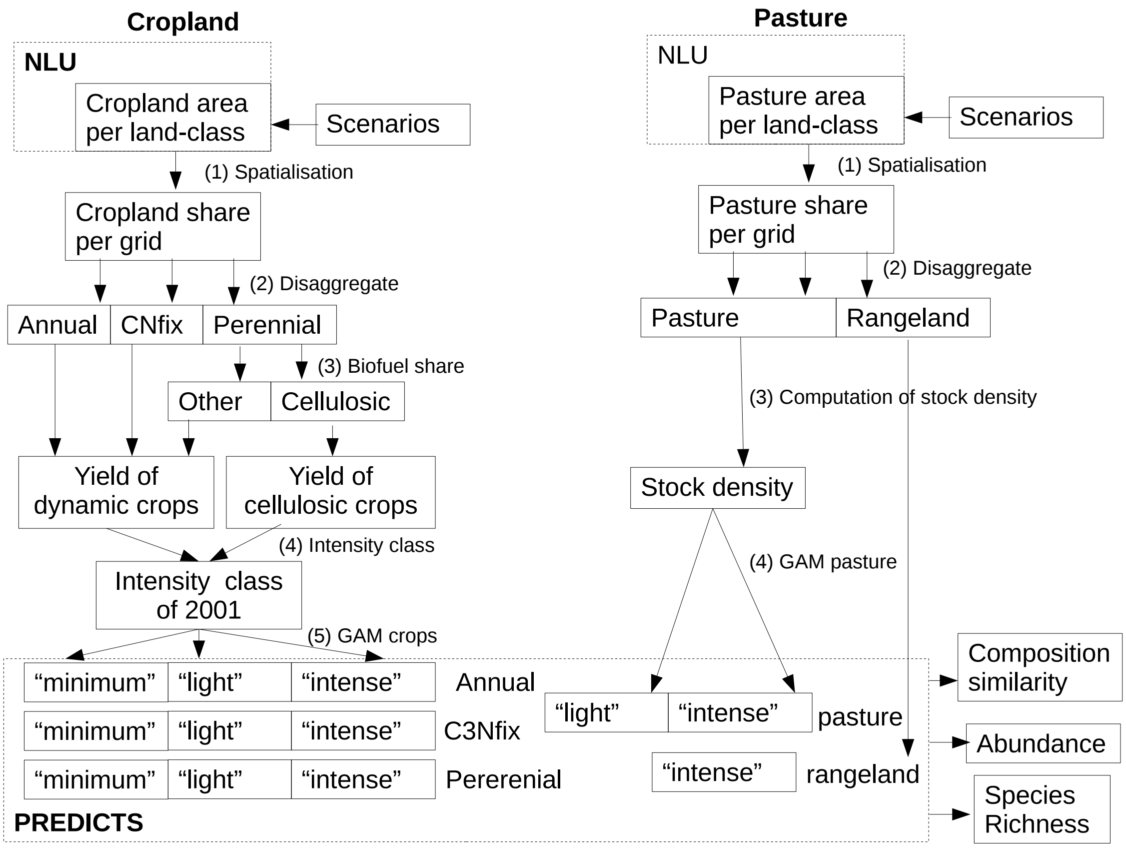


Figure 6 – Links between NLU and PREDICTS models. Steps (1), (2) and (3) match NLU land-use categories to PREDICTS land-use categories. Details are provided in the section. . Steps (4) and (5) link the intensities of NLU land-use to PREDICTS land-use intensities. Details are provided in the following. GAM means General Additive Models. CNfix corresponds to plants which are biologically fixing nitrogen as defined in Hurtt et al. (2011)

Link between land-use categories of NLU and PREDICTS

Coupling the Nexus Land-Use (NLU) and PREDICTS models required changes to both compared to their previous versions, presented respectively in Souty et al. (2012) and Hill et al. (2018). NLU includes numerous agricultural land uses. For example, crops are broken down into 2 classes (dynamic crops and other crops) and pastures are taken from two systems (pastoral system and mixed crop-livestock system). On the other hand, "forest" is a single class.

For this study, a downscaling from land classes (land with similar potential yield) to the 0.5° grid point scale has been added to NLU. This is step (1) in Fig. . To this end, land use changes are distributed in proportion to the distribution of land-cover categories within the land class in the reference year t_0 (2001). For every grid point i belonging to the land class j, the fraction of land-use LU in year t is computed as:

$$\alpha_{\mathrm{LU},i,t} = \frac{\hat{\alpha}_{\mathrm{LU},j,t}}{\hat{\alpha}_{\mathrm{LU},j,t_0}} \times \alpha_{\mathrm{LU},i,t_0} \tag{31}$$

with $\alpha_{\mathrm{LU},i,t}$ the fraction of land-cover LU in grid cell i at time t, $\hat{\alpha}_{\mathrm{LU},j,t}$ the fraction of land-cover LU in land-class j at time t, $\alpha_{\mathrm{LU},i,t_0}$ the fraction of land-cover LU in grid cell i in the reference year and $\hat{\alpha}_{\mathrm{LU},j,t_0}$ the fraction of land-cover LU in land-class j in the reference year. LU can be cropland, pasture or forest.

This downscaling method has been chosen for its transparency and consistency at regional and global scales (Vuuren et al. 2010). We do not analyse results at the grid point scale in particular because of the simplicity of this method.

The two crop categories of NLU, dynamic and other crops, are aggregated into a cropland categrory that matches cropland as defined in Hurtt et al. (2011). The proportions of annual, perennial and nitrogen-fixing crops in the crop mix are calculated as the ratio of the annual, perennial and nitrogen-fixing area to the total cropland area in 2001 from Hurtt et al. (2011) (“Disaggregate” step in Fig. 6). The production of second-generation biofuel is included in “perennial”, otherwise the ratio is kept constant through time.

NLU pasture categories are split into pasture and rangeland based on the rangeland map produced by Hurtt et al. (2011) regardless of whether they belong to pastoral system pasture or mixed crop-livestock system (“Disaggregate” step in Fig. 6). The proportion of rangeland is set to be constant over time.

The PREDICTS model is more accurate with respect to natural area and forest land classes (see the following section). Biodiversity indicators are strongly influenced by the primary or secondary character of natural areas (Watson et al. 2018). In addition, forest and other natural areas, such as savannah, have very different ecological functions. In this PREDICTS model, natural areas (forest and non forest) are aggregated into secondary and primary natural areas. To match this classification of natural areas, NLU forest is split into primary and secondary forest based on the relative proportion of each forest type in 2001 as defined in Hurtt et al. (2011) and assumed to be constant over time. In the scenarios, reforestation consists only of secondary forest.

Table 6 – Abundance, composition similarity and species richness models based on the PREDICTS data base

Land-Use	Abundance model	Species richness model	Composition model
Intercept	0.65895	2.65435	2.189599
Secondary	-0.01415	-0.15875	-0.223229
Rangelands	-0.03463	-0.09300	-1.122190
Pasture Light use	-0.05364	-0.23153	-3.398944
Pasture Intense use	-0.08303	-0.21634	-3.398944
Annual Minimal use	-0.12289	-0.37063	-1.557422
Annual Light use	-0.11470	-0.47360	-1.557422
Annual Intense use	-0.15255	-0.41606	-1.557422
Perennial Minimal use	0.02072	-0.21912	-0.294046
Perennial Light use	-0.09749	-0.42456	-1.063739
Perennial Intense use	-0.06351	-0.51682	-1.801390
Nitrogen Minimal use	-0.04453	-0.37003	-1.280273
Nitrogen Light use	-0.16470	-0.72871	-1.280273
Nitrogen Intense use	-0.23775	-0.67512	-1.280273
Urban Minimal use	-0.01684	-0.15043	1.319501
Urban Light use	-0.10958	-0.34365	1.319501
Urban Intense use	-0.15866	-0.39866	1.319501

Link between land-use intensities

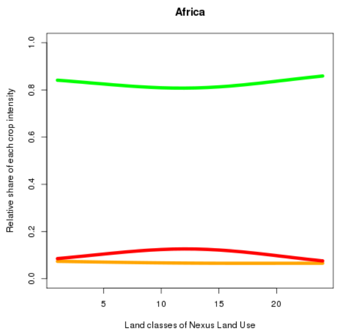
In NLU, 60 land classes are defined in the reference year according to their potential yield (Brunelle et al. 2018). In each land class, the yield is proportional to the potential yield according to a coefficient recalculated each year based on the price of inputs and land. In PREDICTS, three intensification classes break down perennial crops, annual crops and nitrogen-fixing crops into “minimal”, “light” and “intense” use categories. This classification is based on information about field size, inorganic fertiliser and pesticide application rates, use of irrigation and mechanisation of agriculture (Hudson et al. 2014).

In the reference year, a Generalized Additive Model (GAM) is computed to match the relative proportion of “minimal”, “light” and “intense” cropland with the 60 land classes of NLU. A GAM is used to avoid making assumptions on the error distribution, to smooth the relationship between the intensification classes of the two models and to avoid giving too much weight in the relationship to uncertain extreme values. This relationship is defined only for land-classes with a significative amount of land. For extreme land-classes, proportions of intensive, light and minimal categories are set constant (Fig. 7).

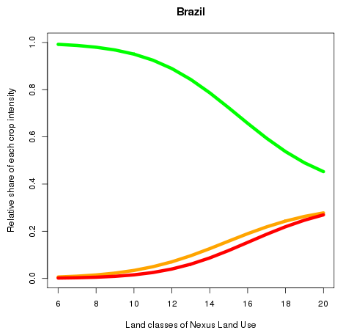
In year t, the land class corresponding to the yield calculated by NLU is linked to the 3 PREDICTS intensification classes by the GAM. This step is called “GAM crops” in Fig. 6. As expected, this relation shows an increase on the relative proportions of the most intensive PREDICTS classes with the increase of the intensification in NLU classes (see Fig. 7).

Ruminant livestock stocking density is used as a link between PREDICT’s pasture intensity and NLU grassland areas. In NLU, there is no geographical distribution of livestock density. Therefore we used the livestock density from the maps produced by Robinson et al. (2014). In this version of the model, the change of system therefore impacts biodiversity only through a change in land-cover (mix of cropland and pasture for ML system and pasture only for the pastoral system) and not through a change in intensity. This step corresponds to the “computation of stock density” step in Fig. 6. In the reference year, the relationship between the relative proportion of “light” and “intense” pasture and livestock density from FAO maps is calculated using a GAM (Fig. 8). This GAM is then used in year t to determine the proportion of “light” and “intense” pasture from the livestock density. This step corresponds to the “GAM pasture” step in Fig. 6.

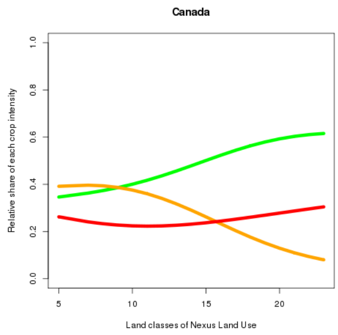
Finally, intensification of forest logging is not represented in this framework because NLU is not able to provide information about changes in forest intensification over time.



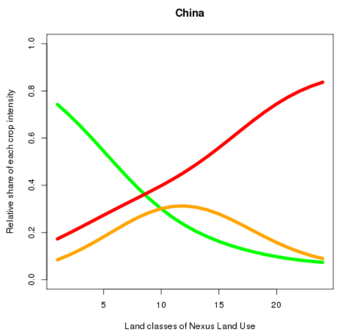
(a) Africa



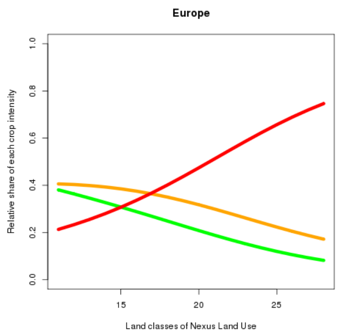
(b) Brazil



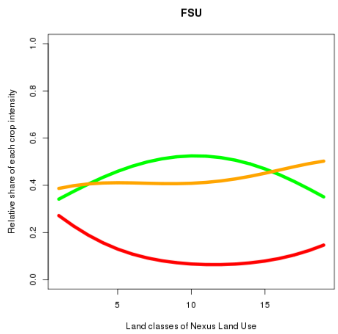
(c) Canada



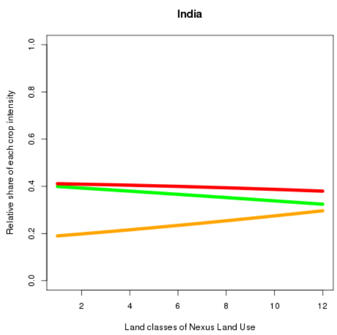
(d) China



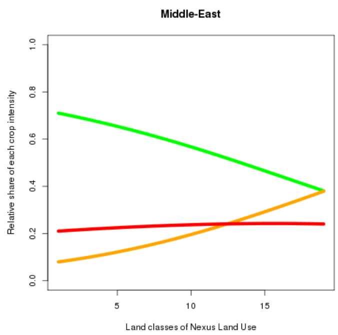
(e) Europe



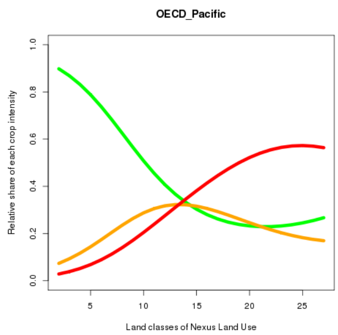
(f) Former Soviet Union



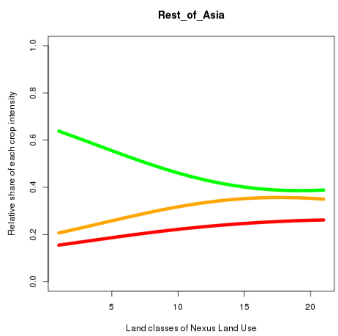
(g) India



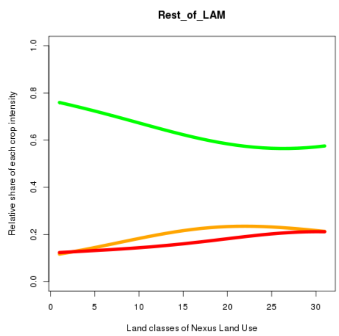
(h) Middle-East



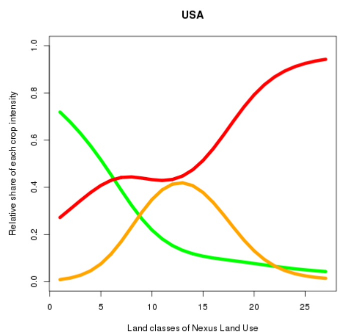
(i) OECD Pacific: Japan + Australia



(j) Rest of Asia

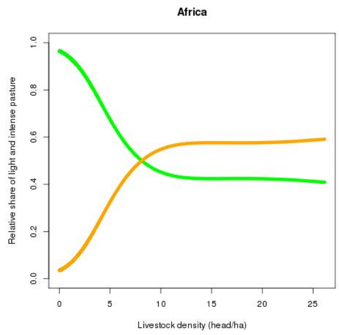


(k) Rest of LAM

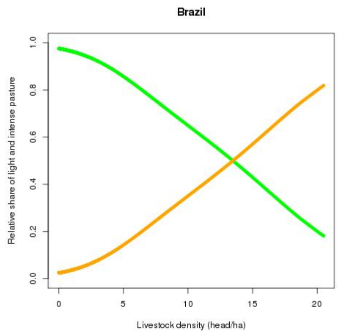


(l) USA

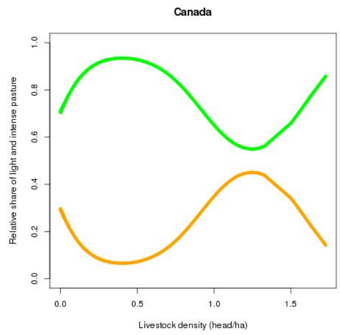
Figure 7 – Regional relative share of crop intensity in PREDICTS (intense, light, minimum respectively in red, orange and green) depending on the crop intensity in NLU.



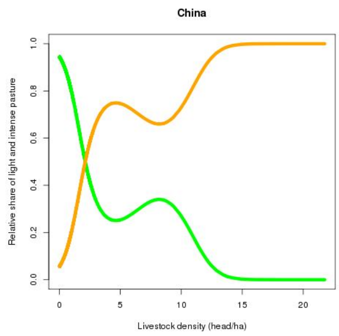
(a) Africa



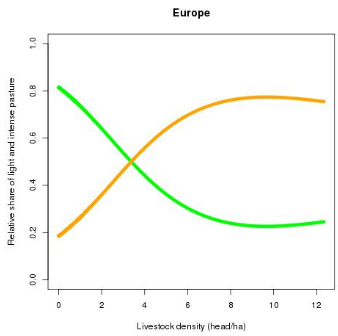
(b) Brazil



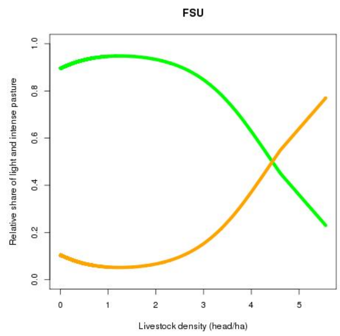
(c) Canada



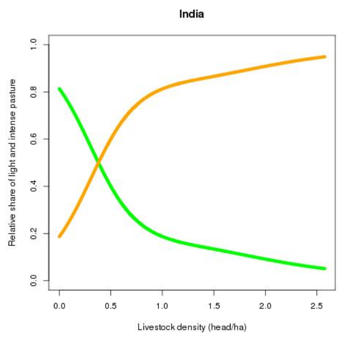
(d) China



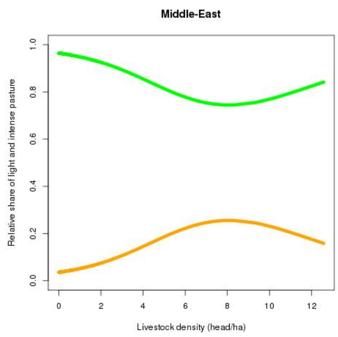
(e) Europe



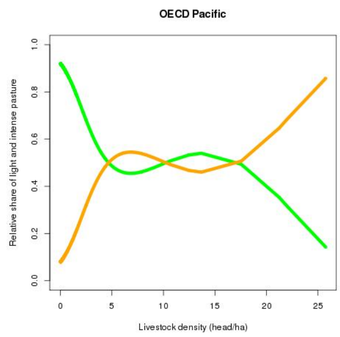
(f) Former Soviet Union



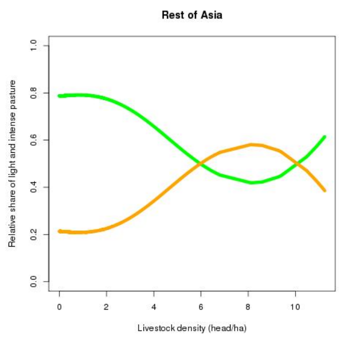
(g) India



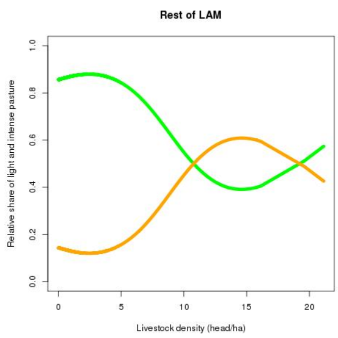
(h) Middle-East



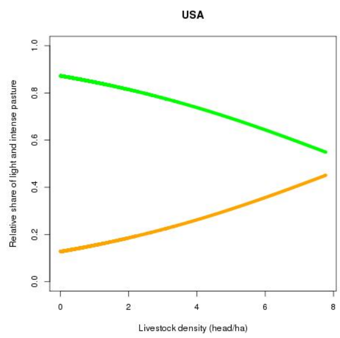
(i) OECD Pacific: Japan + Australia



(j) Rest of Asia



(k) Rest of LAM



(l) USA

Figure 8 – Regional relative share of pasture intensity in PREDICTS (intense, light respectively in orange and green) depending on livestock density in head/ha.

Land-use theories applied in NLU

Presentation of land-use theories applied in NLU

Based on the recent litterature review of land-use theory (Meyfroidt et al. 2018), we describe here the theories found in NLU: Boserup's theory, induced intensification theory, rent theory and access theory.

The Boserup's theory assume that intensification occurs to satifsfy a growing demand. In this case, technology are available. In NLU, intensification of cropland is represented by a choice between chemical inputs and land depending on their price.

Theory of induced intensification adds to Boserup's theory the importance of technology, land policies and biophysical conditions. In NLU, the yield response function to chemical inputs is concave with an asymptote corresponding to the potential yield. This potential yield is calculated from a dynamic vegetation model called LPJmL (Bondeau et al. 2007). Forest scenarios are fully described with exogenous scenarios. We therefore consider that the forest sector is controlled by institutions that limit the land available for agriculture. Replacement of pastures by crops and vice-versa are endogenously determined. The replacement of forest by pastureland and cropland areas is based on exogenous forest scenarios.

In Ricardian theory, rent theory depends on the productivity of the factors of production (land, inputs, labour and capital). In NLU, only land and inputs are explicitly represented and the cost of other factors are included in a fixed cost per hectare defined at the regional level. The rents and prices calculated in NLU are "shadow rents" or "shadow prices". The scarcity rent, representing the rent from the limited amount of land available, and the differential rent, representing the rent along the gradient of land quality, are endogenously calculated by NLU. The scarcity rent is used to define the boundary between the mixed crop-pasture system and the pastoral system. The differential rent is calculated along a land quality gradient represented by the potential yield.

Finally, in the theory of accessibility, land-owners have access to some production factorswhich frame their ability to make land-use choices. In NLU, some pastures are exploited in the pastoral system while they are located on land with good potential yields (land-use maps from Ramankutty et al. (2008)). This exception to the ricardian theory is explicated by the unaccessibility to some production factors. The intensification of these pasture is endogenously based on cropland expansion.

Presentation of middle-range land-use theories applied in NLU

The theories presented above combine to form middle-range theories synthesized by Meyfroidt et al. (2018). Middle-range theories are "contextual generalizations presenting causal explanations of delimited aspects of reality-events or phenomena" (Merton 1968). Meyfroidt et al. (2018) classify them into 3 types of middle-range theories: spillovers, indirect land-use changes (iLUC) and the land-use transitions. Here we focus on the spillovers that are represented in NLU. According to Meyfroidt et al. (2018), spillovers refer to land-use changes occurring following a first land-use and distant geographically and we can distinguish 3 types:

— leakages. These are land-use changes resulting from an environmental policy and mitigating the initial effect of this environmental policy. They are transmitted through the activity leakage market, the land market leakage market, the commodity market leakage market and along the supply chain leakage.

— indirect land-use changes (iLUC). These are land use changes geographically distant from the initial land use change. Here, we will group together the leakages that are happening in another region of the world.

— rebound effects. The rebound effect which consists of an agricultural expansion following an intensification (like Jevon's paradox). It may result from reinvesting the additional income resulting from intensification in agricultural expansion or the increased consumption due to an higher land-use efficiency.

In NLU, we can distinguish 4 mechanisms of land use change (Brunelle et al. 2018):

— a reduction in the area of pasture resulting from the transition from the pastoral system to the mix crop-pasture system

— an increase in crop yield due to an increase in the use of inputs, and vice versa (intensive margin)

— an reduction of the average crop yield resulting from the abandonment of previously cultivated land, and vice versa (extensive margin)

— a relocation of production between regions through international trade

In NLU, land use change mechanisms are due to economic choices that involve the marginal cost of inputs, the price of land (which is actually a shadow price), the food price (which is also a shadow price) and the profit per hectare. All these elements interact with each other, but some mechanisms are directly linked to specific prices and indirectly to others (Table. 7).

Table 7 – Links between land-use changes and prices in NLU. A cross indicates a direct link and empty cell indicates an indirect link.

NLU processes	Land price	Food price	Fertilizer marginal cost	Profit
Reduction of pasture	X			X
Intensive margin		X	X	
Extensive margin	X			X
Reallocation of production		X		

We deduce from this the following table which represents the way spillovers are represented by the land-use exchange mechanisms represented in NLU:

Table 8 – Links between land-use changes and prices in NLU. A cross indicates a direct link and empty cell indicates an indirect link.

NLU processes	Land-market leakage	Commodity-market leakage	Activity leakage	rebound effect
Reduction of pasture	X			
Intensive margin		X	X	
Extensive margin	X			X
Reallocation of production		X		

Chapter 1

Assessing the impact of increased legume production in Europe on global agricultural emissions

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Abstract

The increased use of legumes is viewed as a promising option to mitigate climate change, as they are a source of proteins and provide nitrogen to the soil. In this paper, we evaluate a strategy for the increased use of legumes in Europe until 2050 by integrating a large array of food and natural system processes into a consistent modeling framework. Two contrasting senarios are studied: a supply-side scenario entailing a change in the animal feed mix and a demand-side scenario entailing a shift in human diet. Our results show that the main environmental benefit of legumes is to provide proteins as a substitute for animal products rather than enabling a lower consumption of synthetic fertilizer through increased leguminous nitrogen fixation. In the diet shift scenario, the reduction in emissions at global scale was mainly achieved in the livestock farming sector through a reduction in emissions due to enteric fermentation (38%) and forage reduction (27%). We also show that most of the emission reduction in the plant food farming sector relates to N_2O emissions due to the reduced fertilization (31%), mainly linked to economic choices regarding production allocation and intensification levels. The main part of the emission reduction is exported out of Europe, as Europe re-imports emissions

in the plant food sector by reducing its domestic land needs (mainly pastures) and improving its trade balance.

Keywords: Diet Change; Mitigation Policy; Land-use; Legume; Livestock

1.1 Highlights

- ◇ Mitigated emissions of a european dietary change scenario are 20 greater than in a scenario of soybean production
- ◇ In a dietary change, main source of emission reduction (65% of mitigated emissions) is the ruminant farming sector.
- ◇ In a european dietary change, major part of the emission reduction occurs outside Europe because Europe re-imports initially saved emissions by increasing its plant food production.
- ◇ In a european dietary change, emission reduction in the plant food farming sector results from reduced fertilization, mainly linked to economic choices regarding production allocation and intensification levels.
- ◇ Substitution of mineral by biologically fixed nitrogen has a very limited impact on the emission reduction.
- ◇ In a combinaison of a reforestation scenario and a diet change scenario, the emissions reduction mainly occur through a reduction in CO_2 emissions (54%) and the mitigation of no-CO_2 emissions is reduced compared to a diet change alone scenario.

1.2 Introduction

Reactive nitrogen (Nr) is an indispensable nutrient for agricultural production, since half the global production depends on anthropogenic nitrogen fertilization (Ladha et al. 2005), and 70% of production relies on synthetic nitrogen fertilizers (FAOSTAT 2011). However, Nr dependency has important environmental consequences linked to Nr pollution, including nitrous oxide (N_2O) emissions. The cost of this damage could account for 0.3% to 3% of global GDP (Bodirsky 2014), with N_2O emissions from crop fertilization representing around 3.1 $GtCO_{2,eq}$/year. Based on the middle-of-the-road scenario of shared socioeconomic pathways (SSP2), N_2O emissions will rise from 3 Tg N_2O-N in 1995 to 7-9 Tg N_2O-N in 2045 (Bodirsky et al. 2014). To tackle this nitrogen mitigation challenge, both demand- and production-side policies are necessary (Bodirsky et al. 2014, Clark & Tilman 2017, Davidson 2012, Reis et al. 2016). Supply-side policies consist of improving nitrogen use efficiency (NUE) by means of several options such as genetic improvement and precision farming (Kanter et al. 2016, Lassaletta et al. 2016, Zhang et al. 2015). Demand-side measures mostly comprise waste reduction and a lower consumption of animal products (Popp et al. 2010, Stehfest et al. 2009, Westhoek et al. 2014).

The increase in the share of legumes in agricultural rotation seems to be a promising option for mitigating N_2O emissions from nitrogen fertilization, because: (i) legumes fix nitrogen from the atmosphere through biological reactions, thus avoiding emissions related to the application and synthesis of nitrogen fertilizer (Jeuffroy et al. 2012), and (ii) their high nitrogen content makes them good candidates to replace animal products in human diets (Poore & Nemecek 2018) and be used in animal feed (Davis et al. 2015). The introduction of legumes into rotations can therefore be associated with either demand-side policies in the case of diet shift or supply-side policies in the case of nitrogen-rich livestock feed production. Moreover, debates on protein import dependence led by the European Commission concluded that increasing domestic legume production could reduce environmental pollution, provide economic advantages to European farmers, and improve food quality for consumers (Häusling 2011).

However, some effects of increased legume consumption could reduce the effectiveness of legumes in reducing greenhouse gas (GHG) emissions, such as emissions due to nitrogen leaching, the yield differential between the legumes and replaced crops, and the changes in Europe's trade balance. First, the introduction of legumes into rotations produces nitrogen-rich residues that disrupt the carbon/nitrogen ratio of the soil and release Nr into the environment through leaching (Cassman et al. 2002). This leaching leads to indirect N_2O emissions (IPCC 2006*a*). Second, the development of legumes at the expense of existing crops can lead to an increase in the cultivated area because of the poor yield performance of legumes compared to other existing crops (FAOSTAT 2015) and thus higher CO2 emissions due to the land-use change. In this respect, the expansion of soybean has played an important role in Brazilian deforestation by pushing cattle ranching to deforestation frontiers (Arima et al. 2011, Bowman et al. 2012, Fehlenberg et al. 2017). Finally, the development of legumes in Europe affects trade flows across regions, especially given that 50% of European consumption is produced in other regions of the world (Yu et al. 2013). This effect has been illustrated in the case of diet changes in Europe where the European Union has become a net cereal exporter

(Westhoek et al. 2014).

In this study, we seek to quantify the emission reduction subsequent to the development of legumes in Europe by integrating the abovementioned effects into a consistent analytical framework. For a comprehensive representation of the increase in legume consumption in Europe, we studied two contrasting scenarios: (i) an increase in legume consumption from 2.7 to 11.4 kg/capita/year between 2020 and 2050 (demand-side scenario), equivalent to 11 Pkcal of field pea (called in the following the diet shift scenario), and (ii) a feed change scenario (called in the following the feed mix scenario) from rapeseed to soybean (supply-side scenario), equivalent to an 11 Pkcal increase in soybean in between 2020 and 2050. The impacts of these scenarios on global emissions are then represented using the Nexus Land-Use (NLU), a global agricultural intensification model (Souty et al. 2012). This model is a global partial equilibrium model that represents the regional markets for plant and animal products, while also accounting for international trade. The explicit representation of the livestock and plant sectors considers changes resulting from the development of legumes in Europe and how the sectors and their interrelations evolve. A nitrogen balance was added to the model in this study. Nitrogen leaching emissions from legume residues are determined with the nitrogen balance. The production function was also modified to distinguish nitrogen from potash and phosphorus and explicitly account for the different sources of nitrogen. Finally, to take into account the impact of these scenarios on N_2O and other GHG emissions in the agricultural sector, the calculation of emissions associated with agricultural production and land-use changes was also added to the model using a Tier 1 methodology for the plant sector and Tier 2 methodology for the livestock sector based on the IPCC (2006*a*).

To present the emission changes resulting from increased legume consumption in Europe, we (i) analyze the emission changes region by region, (ii) separate the plant and livestock sector emissions, and (iii) explain how changes in the nitrogen balance impact nitrogen emissions.

1.3 Methods

1.3.1 Overview of the modelization framework

Emission changes from increased legume consumption in Europe are assessed using the NLU model. NLU is a partial equilibrium model representing the agricultural sector comprising crop and livestock production at a global scale. This model is suited to our study, as it entails an explicit nitrogen balance, the calculation of GHGs associated with agricultural production and land-use changes (LUC), and a basic representation of international trade (See Fig. 1.1).

An extensive description of the NLU model is provided in Souty et al. (2012). In this paper, we provide a comprehensive description of how the nitrogen balance (Section) and emissions associated with agricultural production and land-use changes (Section) are introduced in the model, because they are important novelties of the NLU model in addition to being a central aspect of our analysis. A description of the modeling of international trade and livestock production is provided in the Supplementary Material.

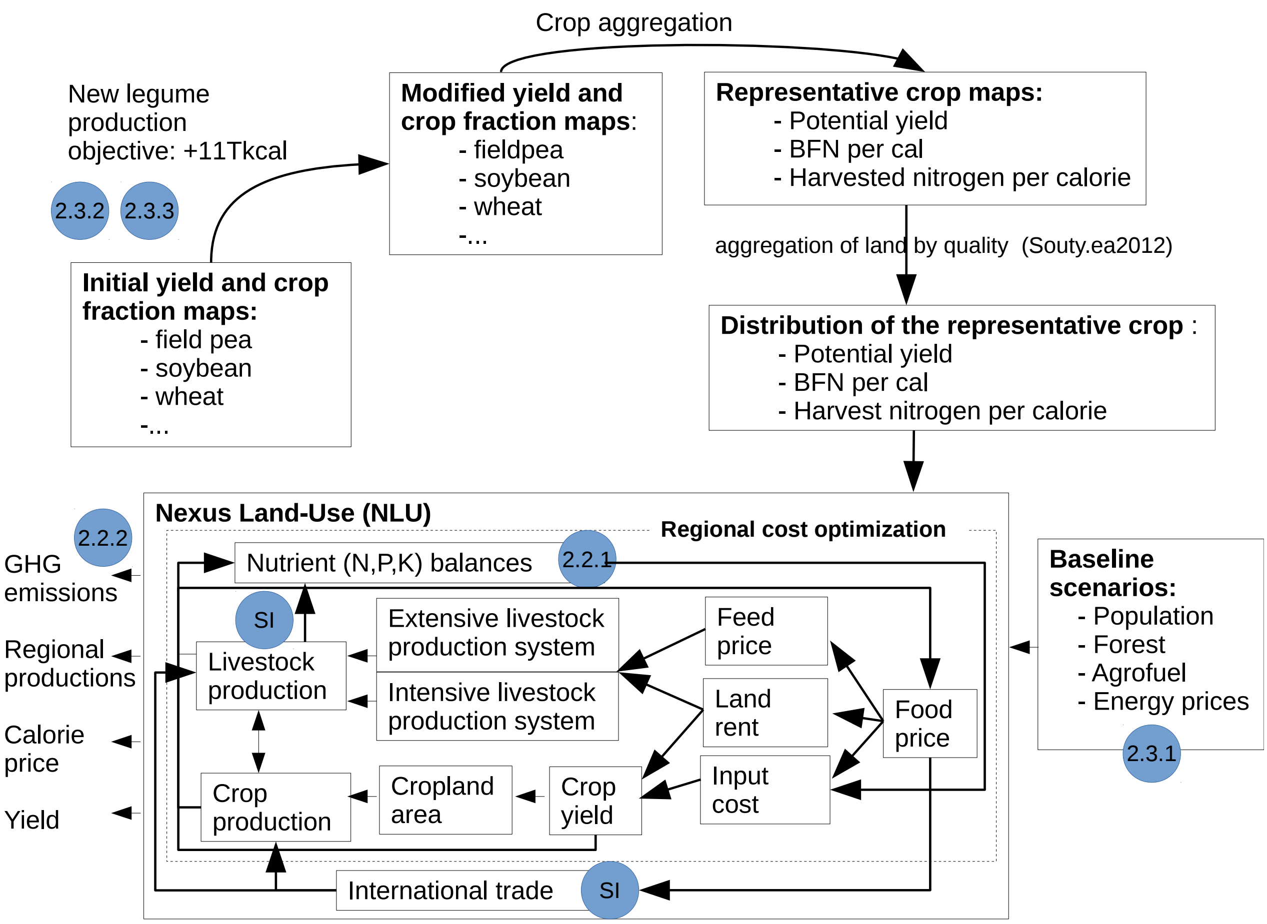


Figure 1.1 – Influence of land-use and mitigation strategies on GHG emissions mitigation in the chapter. 1

1.3.2 The Nexus Land-Use (NLU) model

Nitrogen balance, synthetic nitrogen use and nutrient cost

A detailled description of the nitrogen balance, synthetic nitrogen use and nutrient cost is provided in section.

Computation of GHG emissions by the NLU

A detailled description of the computation of GHG emissions by the NLU is provided in section.

1.3.3 Scenarios

To study the impacts of increased legume consumption in Europe (geographical scope of Europe is defined Fig. 4 in the Supplementary Material), we distinguish one baseline scenario as well as two scenarios of increased legume consumption taking place between 2020 and 2050.

Baseline

At the baseline (Table 1.1), the main features follow the shared socioeconomic pathway 2 (SSP2) and are presented in Table 1.1.

Table 1.1 – Parameters describing the baseline scenario.

	Unit	2010	2050
Population scenario (SSP2)	10^9 heads	7.63	9.28
Plant consumption per capita	kcal/cap/day	2447	2595
Animal consumption per capita	kcal/cap/day	494.97	612
Agrofuel production	10^8 Mkcal	2.89	6.4
Annual reforestation rate (global average)	%	0.1	0
Oil price	$2005/Gjoule	26.18	75.58
Natural gas price	$2005/Gjoule	9.46	18.2

Diet shift scenario

In the diet shift scenario, the European ruminant protein consumption is partly substituted with legume protein consumption. Legume consumption in Europe increases from its current level of 2.7 kg/capita/year to 11.4 kg/capita/year. This objective is derived from a Canadian food policy regarding legume consumption by 2030 (Solagro 2016). This food policy objective was used, because Canada and Europe have similar development levels, with the Canadian objectives seem to be both ambitious and achievable. In the European context, field pea, as a traditional European crop, is a relevant replacement for ruminant products (Cernay et al. 2016).

To determine the decrease in ruminant protein consumption corresponding to an increase in field pea consumption, we used a coefficient of available protein content per crop energy unit based on FAOSTAT Food Balance Sheets (FAOSTAT 2015). Protein digestibility coefficients of 0.8 for crop proteins and 0.9 for animal proteins (FAO and WHO 2001) are also used to determine the quantities substituted. The diet shift described in this scenario corresponds to an increase in the consumption of plant products of 108 kcal/cap/day (+4.2%) and a decrease in the consumption of animal products of 94 kcal/cap/day (-12%) (Fig. 1.2).

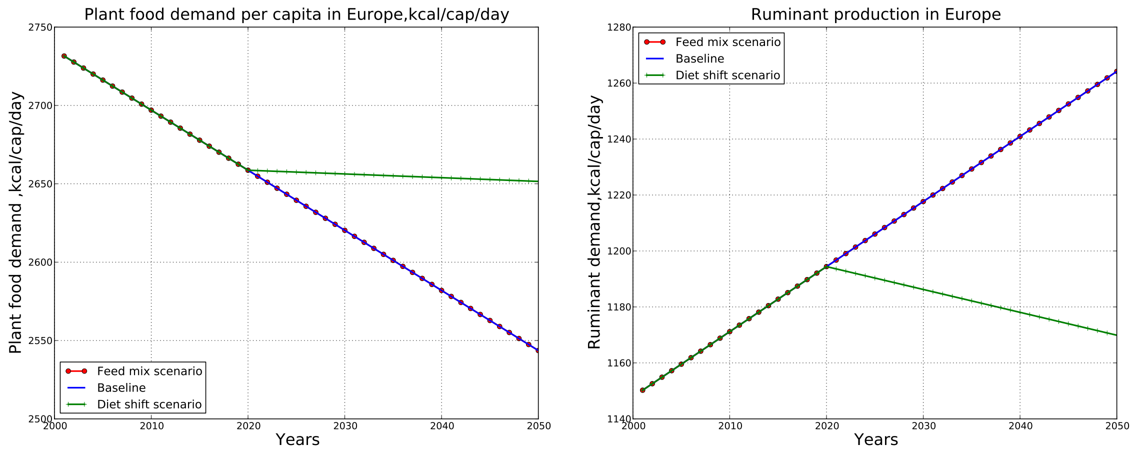


Figure 1.2 – Change in European plant and animal demand at baseline and in the scenarios of increased legume consumption.

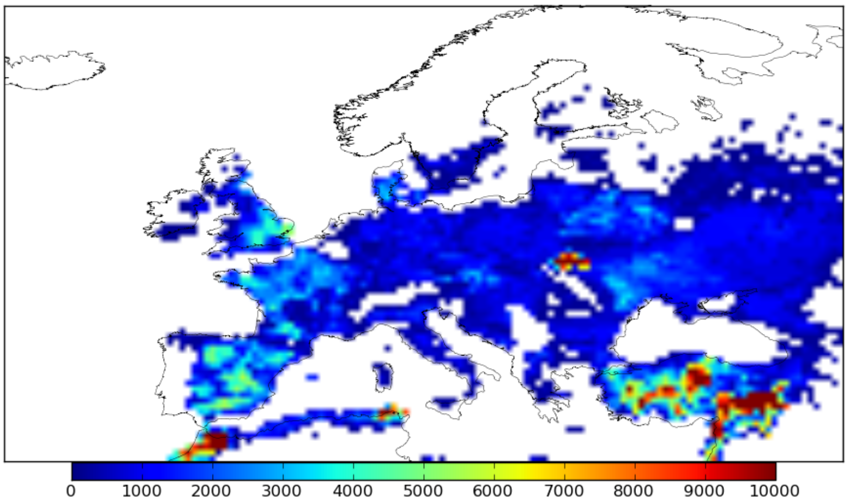
In the diet shift scenario, field pea is introduced by modifying the European mix of crop production to reflect the additional production of 11 Pkcal of field pea. Within the region, field pea is introduced to land with the best potential yield for field pea (Fig. 1.3) as defined by the Lund-Potsdam-Jena managed Land (LPJmL) vegetation model (Bondeau et al. 2007) at the expense of surrounding crops while maintaining the same cultivated area. This scenario corresponds to a 3.4 Mha increase in the area of field pea, representing 4% of European cropland (Fig. 1.4). The crop mix of other regions is not modified.

The crops from each grid point on the map are then aggregated into a representative crop based on their energy content and representative crops of grid points with the same potential yields are aggregated into land classes (Souty et al. 2012). The increased consumption of field pea in Europe corresponds to an increase in the potential calorie fixation of 272 kgN/Pkcal (+27%), an increase in potential harvested nitrogen per calorie of 226 kgN/Pkcal (+4.2%) and a decrease in the average potential yield of 492 kcal/ha (-0.05%) of the aggregate crop (Fig. 1.2).

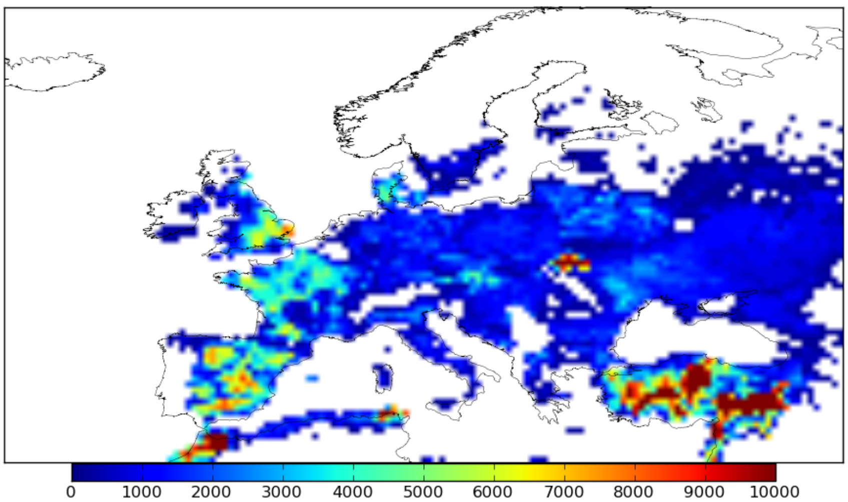
Feed mix scenario

In the second legume scenario, we increase the share of European soy production in animal rations and we maintain the same demand for protein and energy and the same total cultivated area. The total amount of soybean introduced in this scenario is 11 Pkcal to ensure that a change in this scenario is comparable to that of the dietary change scenario. This scenario allows us to estimate the impact of increased soybean use in Europe to feed livestock and replace non-leguminous crops.

As in the previous scenario, the geographical distribution of crops in Europe is modified, assuming that Europe produces all the required soybean, and the crop mixes in other regions remain unchanged. In Europe, soybean is introduced where its potential yield is the highest (i.e., the Danube region and southern European countries as represented in Fig. 1.4) at the expense of surround-



(a) Field pea area at baseline



(b) Field pea area change in diet shift scenario

Figure 1.3 – Field pea area in Europe at baseline (a). Changes in the field pea area in Europe due to an increase in legume consumption of 11 Pkcal of field pea compared to the field pea area in Europe at baseline (b)

ing crops in the same grid cell. To maintain the same protein demand, the increase in soybean as a protein-rich crop is compensated by a reduction in wheat, which is also a protein-rich crop (Table 1.3). To maintain the same energy production, rapeseed production, which has a higher yield per hectare than soybean, also increases to compensate the soybean increase, since soybean has a relatively low energy yield (Table 1.3). Lastly, to keep the total areas unchanged, the other crop areas are also changed.

Table 1.2 – Average European energy and protein yields of the aggregated crop at baseline based on the feed mix and diet shift scenarios in 2050.

	Feed mix	Baseline	Diet shift
Potential energy yield (Mkcal/ha)	25.1	25.2	25.1
Potential protein yield (tN/ha)	143.1	142.2	138.2
Harvested N per cal[4]	5.7	5.6	5.5
BFN per cal[4]	0.5	0.3	0.5

Table 1.3 – Average European energy and protein yields of a selection of crops at baseline

	soybean	wheat	rapeseed	other crop
Actual energy yield (Mkcal/ha)	9.1	14	13.6	4.6
Potential energy yield (Mkcal/ha)	10.9	23.5	22.3	5.8
Actual protein yield (tN/ha)	0.86	0.49	0.52	0.49
Area in baseline (Mha)	0.4	56.4	4.4	83.3

In the final crop mix, the rainfed soybean area increased by 1.05 Mha, the rainfed wheat area decreased by 5.2 Mha, the rainfed rapeseed area increased by 0.84 Mha, and the area of other crops increased by 3.27 Mha.

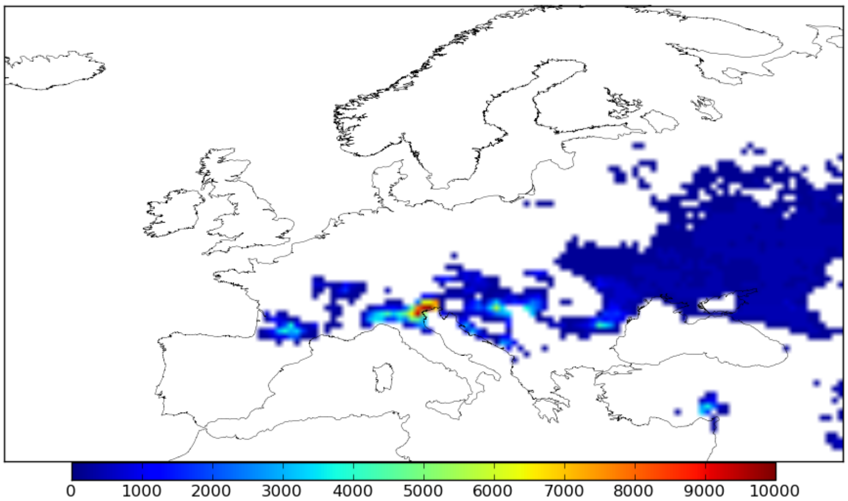
The resulting geographical crop distribution has the same amount of energy and nitrogen as the baseline geographical crop distribution. Thus, average potential yield and average potential harvested nitrogen are similar in the feed mix scenario and at baseline (Table 1.2). The BFN per calorie increases by 0.2 kgN/kcal (+40%) in the feed mix scenario compared to the baseline.

1.3.4 Decomposition of GHG emissions

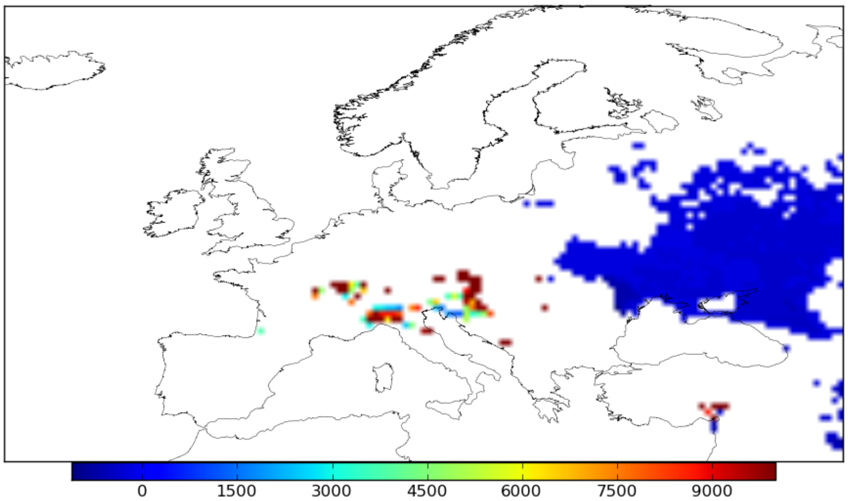
The increased use of legumes in Europe triggers different types of changes, particularly in terms of the nitrogen balance, ruminant production, and international trade. To disentangle these different channels of emission changes, we decomposed emission changes depending on (i) their geographic origin, (ii) the production sector involved their production, and (iii) the change in the nitrogen balance in relation to N_2O emissions.

1.3.5 Comparing nitrogen balance from the literature

To evaluate our representation of the nitrogen balance, we compared our outputs with other studies – Smil (1999), Bouwman et al. (2009), Bodirsky et al. (2014), Liu et al. (2016) – for the reference



(a) Soybean area at baseline



(b) Soybean area change in feed mix scenario

Figure 1.4 – European soybean area at baseline (Portmann et al. 2010) (a). Changes in soybean areas in Europe due to an 11 Pkcal increase in soybean production compared to the soybean area in Europe at baseline (b).

year and 2050. To present a more meaningful comparison of the nitrogen balance established in this study, we describe here a modified balance based on Bodirsky et al. (2014) in which nitrogen from aboveground crop residues, belowground crop residues, seeds, and soil organic matter is not included in nitrogen sources, while crop residues are not included in outputs. The N sources omitted amount to 50 TgN/yr over a total of 326 TgN/yr. In addition, the NUE is recalculated from the modified balance nitrogen sources and output.

For the reference year, the nitrogen balance computed in this study using the NLU is in the lower part of the range of total nitrogen input (see Table 1.4). Differences may be explained by the use of different reference years (in Bodirsky et al. (2014), a later reference year in 2010 partly explains the larger total nitrogen inputs), different scopes (Smil (1999) additionally consider forages, while Bouwman et al. (2009) also include pasture areas), and the computation of nitrogen from rotations (lower in the NLU, because we consider nitrogen left only in legume residues).

In 2050, the differences of total nitrogen input (between 271 TgN and 396 TgN in 2050) are partly due to the differences of scope: pastures are included in the nitrogen balances of Bouwman et al. (2009). The main reason for the differences, however is visible in nitrogen use efficiency: 0.24 in this study compared to 0.42 in Bouwman et al. (2009) and 0.48 in Bodirsky et al. (2014)). Part of the difference in NUE is attributable to discrepancies in harvested nitrogen. The difference in NUE between studies however mainly reveals different assumptions based on the evolution of nitrogen use efficiency between 2020 and 2050. While NUE increases in Bouwman et al. (2009) because total fertilizer use is taken as an inverse function of GDP in IMAGE 2.4 (Bouwman et al. 2006) and increases in Bodirsky et al. (2014) because of investments in yield-increasing research and technology (Dietrich et al. 2014), NUE decreases until 2050 in the NLU due to an increasingly inefficient use of nitrogen with yield increase (Souty et al. 2012). This inefficiency in the use of nitrogen for high yields is reflected in a more substantial use of synthetic nitrogen in the NLU compared to other studies (Table 1.4).

Table 1.4 – Comparison of NLU's nitrogen balance with other studies'nitrogen balance

					Reference year		Projected year	
Study	Reference year	Projected year	Scope	Elements	Element quantity (TgN/yr)	Total nitrogen input (TgN/yr)	Element quantity (TgN/yr)	Total nitrogen input (TgN/yr)
NLU	2001	2050	Cropland	Mineral fertilizer	75	136	399	271
				Manure	28		79	
				Rotation	5		8	
				Fixation	24		32	
				Deposition	18		16	
				Harvest	58		98	
Liu	2000	2030	Cropland	Mineral fertilizer	97	156	-	271
				Manure			-	
				Rotation	17		-	
				Fixation	25		-	
				Deposition	17		-	
				Harvest	92		-	
Bodirsky	2010	2045	Cropland	Mineral fertilizer	116	204	119	276
				Manure	31		62	
				Rotation	0		0	
				Fixation	36		57	
				Deposition	21		38	
				Harvest	84		133	
Bouwman	2001	2050	Cropland and pasture	Mineral fertilizer	83	249	122	396
				Manure	101		190	
				Rotation	0		0	
				Fixation	30		35	
				Deposition	35		49	
				Harvest	93		166	
Smil	1995	-	Cropland and forages	Mineral fertilizer	78	169	-	-
				Manure	18		-	
				Rotation	14		-	
				Fixation	33		-	
				Deposition	20		-	
				Harvest	50		-	

1.4 Results

1.4.1 Comparison of emission reduction between the diet shift and feed mix scenarios

Emission reductions in 2050 reach 211 $MtCO_{2,eq}$ in the diet shift scenario, while they are much more limited in the feed mix scenario (10 $MtCO_{2,eq}$) (Fig. 1.5). This result is explained by the nature of substitutions in each scenario: in the diet shift scenario, legumes substitute animal products, which are much more land and emission intensive, while in the feed mix scenario, legumes substitute other feed sources with more similar emission intensities.

Table 1.6 provides a comprehensive overview of the different components of the nitrogen balance at baseline and in the two legume scenarios. The main expected effect of the feed mix scenario is an increase in the biological fixation of nitrogen through the greater use of leguminous crops. Table 1.6 shows that this effect is nevertheless extremely limited (+1% compared to baseline; see first row of 1.6). The diet shift scenario involves a much more complex dynamic. For this reason, we focus on this scenario throughout the rest of this section.

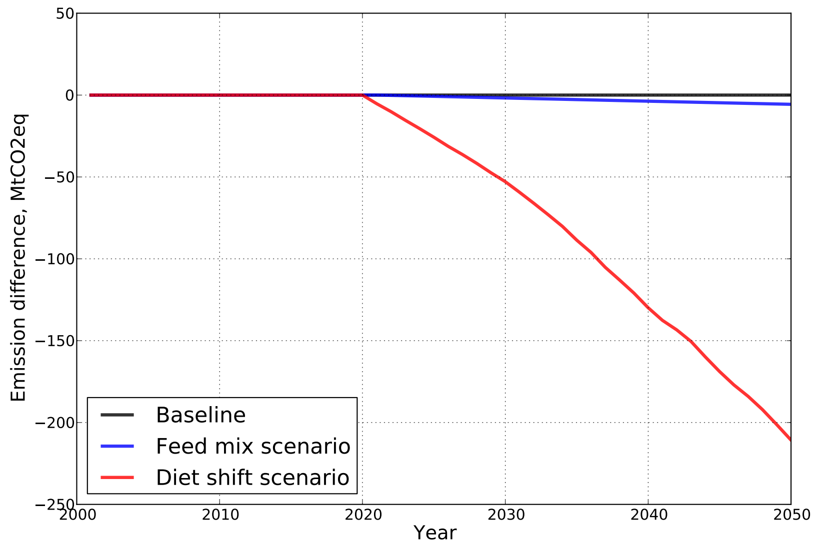


Figure 1.5 – Emission differences between the diet shift scenario and baseline, and between the feed mix scenario and baseline.

Table 1.5 summarizes the different effects at play in the diet shift scenario according to location (World/Europe) and sector (crop: BFN, plant fertilization, legume residues; livestock: fodder fertilization, enteric fermentation and manure management). These results are detailed in sections 1.4.2, 1.4.3 respectively.

1.4.2 Location of emission changes around the world

Figure 1.6 reveals that the major part of emission changes in 2050 (130 $MtCO_{2,eq}$ or 60% of mitigated emissions) following the diet shifts in Europe occurs outside Europe. This unbalanced re-

Table 1.5 – Distribution of emission changes among the main sources of emission mitigation in the diet shift scenario. A positive emission difference means an increase in emissions in the diet shift scenario compared to baseline (no diet shift). On the contrary, a negative emission difference means a decrease in emissions in the diet shift scenario compared to baseline (no diet shift).

	Emission difference ($MtCO_2$)	Percent of total mitigated emissions
Total mitigated emissions	-211	100%
Global emission difference:		
- by BNF	-1.0	0.48%
- due to plant fertilization for human food	-64.6	31 %
- due to legume residues	0.225	-0.11 %
- due to fodder fertilization	-57.6	27%
- due to enteric fermentation and manure management	-81.2	38%
- due to LUC	-0.97	0.46%
- due to other sources	-5.5	2.6%
European mitigated emissions	-81	38.6%
- by BNF	-3.8	1.8%
- due to plant fertilization for human food	-10	4.8%
- due to legume residues	0.726	-0.34%
- due to fodder fertilization	-8.5	4%
- due to enteric fermentation and manure management	-59	28%
- due to LUC	0.156	-0.07%
- due to other european sources	-5.5	2.6%

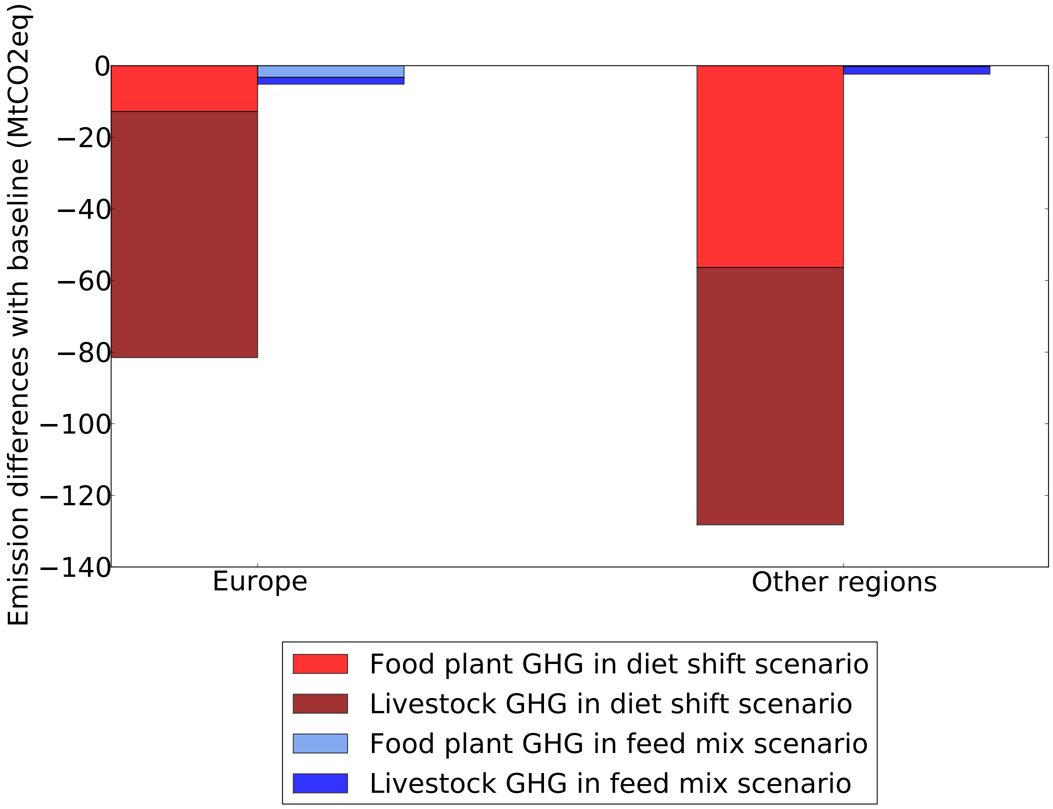
* other sources include CH_4 from rice cultivation and N_2O from pasture fertilization

duction in GHG emissions between Europe and the rest of the world is mainly due to changes in the global balance of trade involving increased crop production in Europe (+169 Pkcal), while the rest of the world produces less than at baseline (-104 Pkcal). The improved trade balance of Europe results from increased competitiveness in crop production, resulting from high quality land freed by the lower domestic livestock production and, to a lesser extent, from the reduction in fertilizer costs through an increase in biologically fixed nitrogen. Despite an increase in the food plant production, Europe emits less emissions in the plant farming sector because of a reduction of its emissions per unit of plant food production enabled by the increased biologically fixed nitrogen and an higher average yield linked to the reallocation of production on the high quality land freed by the lower domestic livestock production. Even though average yield increases, the use of input per unit area and the production price decrease in Europe.

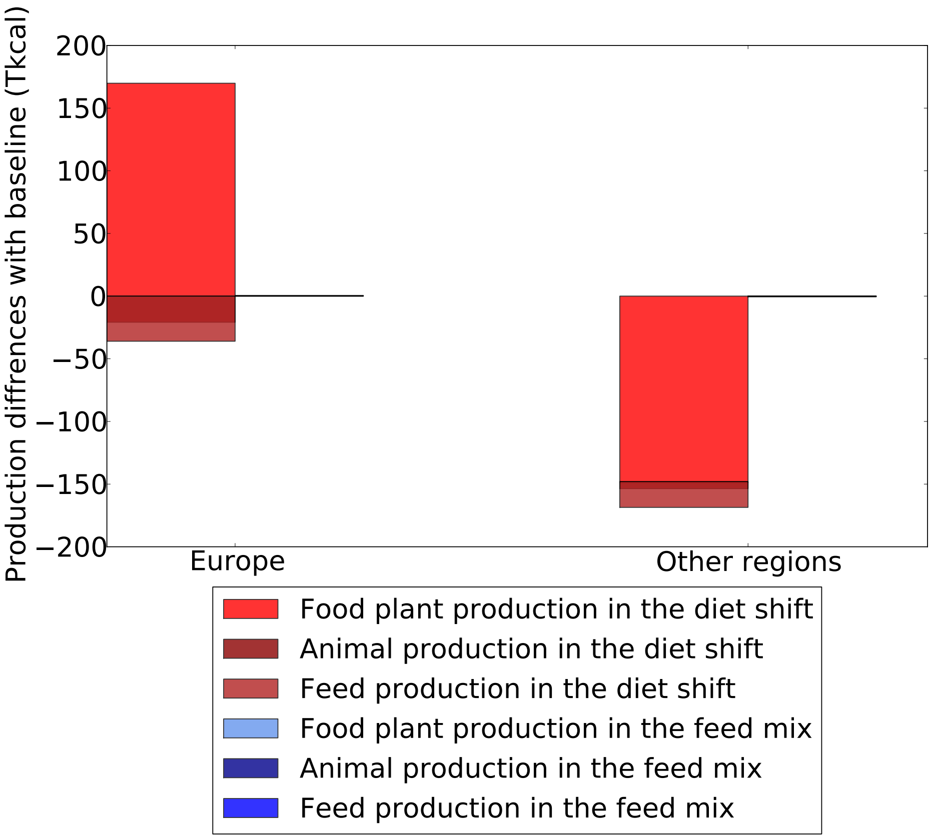
1.4.3 Emissions change in plant food farming and livestock farming

The decrease in global emissions in 2050 according to the diet shift scenario (211 $MtCO_{2,eq}$/year) is dominated by the emissions change in animal farming (139 $MtCO_{2,eq}$/year) followed by plant food farming (71 $MtCO_{2,eq}$/year, see Fig. 1.7). The reduction in emissions associated with land-use changes is almost negligible (1 $MtCO_{2,eq}$/year).

The major part of the emission reduction in food farming (79 %) comes from a reduction in N_2O emissions (-64 $MtCO_{2,eq}$). As shown on Fig. 1.8, the reduction in N_2O emissions results from a reduction in both direct (-60 $MtCO_{2,eq}$) and indirect emissions from fertilization (-4 $MtCO_{2,eq}$), as well as a slight increase in emissions due to the leaching of legume residues (+0.5 $MtCO_{2,eq}$).



(a) Regional crop and ruminant emissions



(b) Regional crop and ruminant production

Figure 1.6 – Production and GHG emissions in Europe and other regions for the two legume scenarios compared to baseline.

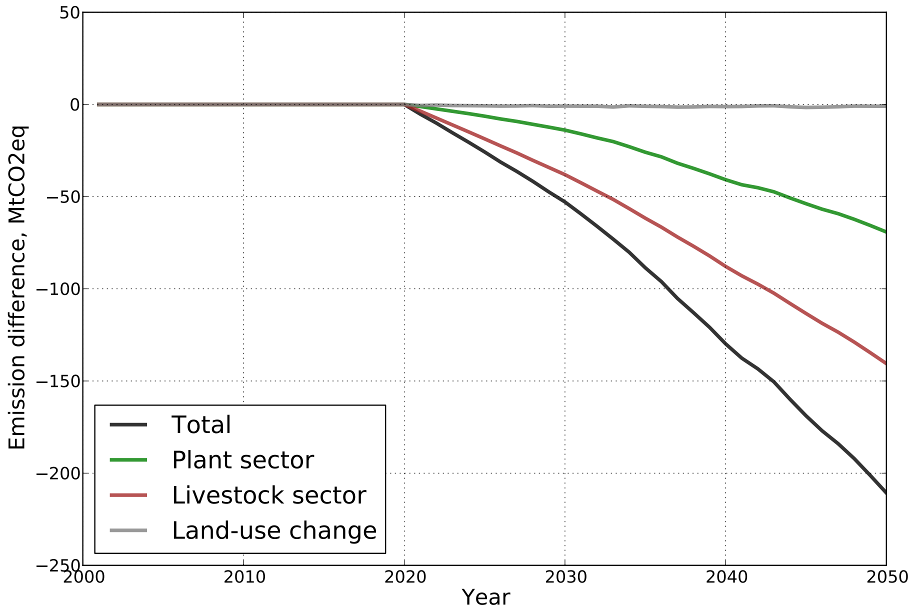


Figure 1.7 – Distinction between GHG emissions for the plant and livestock sectors according to the diet shift scenario.

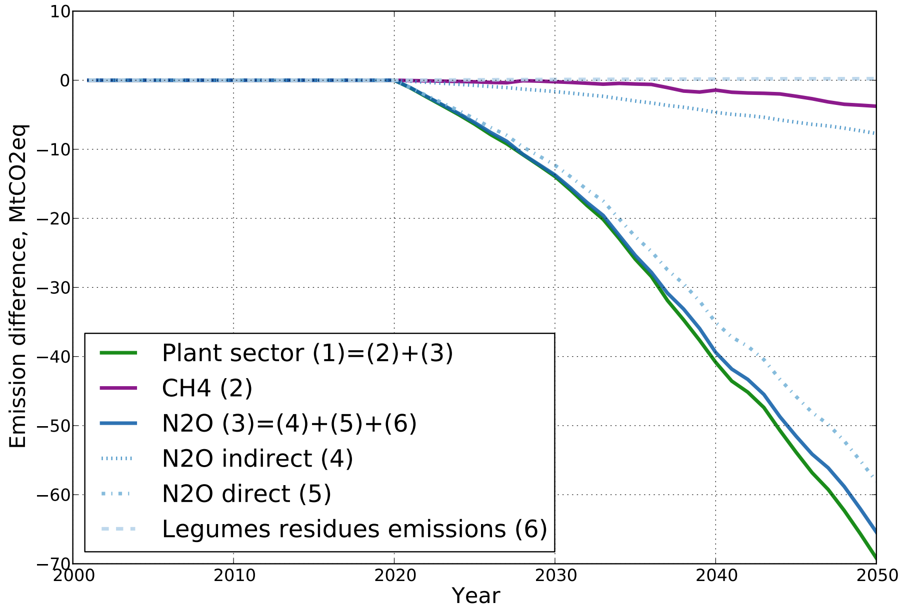


Figure 1.8 – Decomposition of GHG emissions from plant sector in the diet shift scenario.

The reduction in direct and indirect N_2O emissions from fertilization is due to the reduction in synthetic nitrogen fertilization (-8.5 TgN) and organic nitrogen fertilization (-0.51 TgN) (see the eleventh row of Table 1.6). The vast majority of this reduction results from changes in the international allocation of crop production and economic trade-offs that favor the use of land, which is relatively more abundant, over synthetic nitrogen. The substitution of fertilizers with BFN represents only a minor share of the decrease in nitrogen fertilization, corresponding to the equivalent of 4.05 $MtCO_{2,eq}$ (see Table 1.7).

Table 1.6 – Components of the nitrogen balance in 2050 computed using the NLU at baseline and according to the diet shift scenario (here, diet shift) and feed mix scenario (here, feed shift).

	World			Europe			Other regions		
N balance[1,2]	Baseline	Diet shift	Feed shift	Baseline	Diet shift	Feed shift	Baseline	Diet shift	Feed shift
Biologically fixed nitrogen (TgN)	32	32	32	1	1	1	31	31	31
Yield fixation N difference (%))	-0	0	1	-44	0	-7	1	0	1
Synthetic N (TgN)	136	127	135	11	10	11	125	118	125
Synthetic difference (%)	7	0	6	15	0	10	6	0	6
Area fixation N (TgN)	7	0	6	15	0	10	6	0	6
Area fixation N (%)	0	-1	0	0	8	0	0	-1	0
Deposition N (TgN)	16	16	16	1	2	1	15	14	15
Deposition N difference (%)	0	0	0	-8	0	-8	1	0	1
Rotation N (TgN)	8	8	8	0	0	0	8	8	8
Rotation N difference (%)	-0	0	1	-44	0	-7	1	0	1
Manure N (TgN)	79	79	79	6	6	6	73	73	73
Manure N difference (%)	1	0	1	6	0	6	0	0	0
Input N (TgN)	269	269	261	20	20	18	250	250	242
Input N difference (%)	0	0	-3	-0	0	-6	-0	0	-3
Harvest N (TgN)	98	98	98	10	11	10	88	87	88
Harvest N difference (%)	0	0	0	-6	0	-5	1	0	1
Left N (TgN)	8	8	8	0	0	0	8	8	8
Left N difference (%)	0	0	1	-44	0	-7	1	0	1
Lost N (TgN)	171	163	171	9	7	9	162	155	162
Lost Nitrogen difference (%)	5	0	5	24	0	22	4	0	4
NUE	0.236	0.246	0.237	0.529	0.597	0.537	0.224	0.231	0.224
NUE difference (%)	0	4	0	0	13	1.5	0	3	0

[1] Gray rows indicate relative differences from baseline, expressed as percentages

[2] A description of the nitrogen balance is provided in the Methods section above.

Table 1.7 – N_2O emissions from the plant sector ($MtCO_{2,eq}$) in 2050.

	World			Europe			Other regions		
Emissions	Baseline	Diet shift	Feed shift	Baseline	Diet shift	Feed shift	Baseline	Diet shift	Feed shift
N_2O emissions	1.8	1.7	1.8	0.1	0.1	0.1	1.7	1.6	1.7
Direct N_2O emissions	1.55	1.49	1.54	0.12	0.11	0.12	1.43	1.38	1.43
Indirect N_2O emissions	0.227	0.219	0.227	0.018	0.017	0.018	0.209	0.203	0.209
Saved N_2O emissions thanks BFN	0.955	0.959	0.968	0.02	0.035	0.033	0.924	0.936	0.936
N_2O emissions from legume residues	0.05	0.05	0.05	0.001	0.002	0.002	0.05	0.05	0.05

We distinguish three types of emissions in the livestock sector: emissions from enteric fermentation and manure management, emissions from intensive pasture fertilization, and emissions from fodder crop production (see Fig. 1.9). In the diet shift scenario, the major share of the emission change derives from a reduction in enteric fermentation and manure management (57% or 81 $MtCO_{2,eq}$ in 2050), followed by fodder crop production (36% or 55 $MtCO_{2,eq}$ in 2050) and pasture fertilization (-1.7 $MtCO_{2,eq}$). As expected, most of the emission reduction in enteric fermentation and manure management occurs in Europe (58 $MtCO_{2,eq}$ or 71% of emission reduction in enteric fermentation and manure managementin 2050). However, compared to the 12% decrease in ruminant calorie consumption simulated in our scenario in Europe in 2050, the reduction of this category of emissions appears to be small, even when including the emission reduction outside Europe (81 $MtCO_{2,eq}$ or 38% of mitigated emissions). This can be explained by the changes in the livestock production system simulated by the NLU. According to our results, the proportion of ruminants extensively produced is slightly higher in the diet shift scenario than at baseline (9% vs 8.7%) due to economic trade-offs between both systems represented in the NLU model. Given the larger emission factors associated with enteric fermentation and manure management that prevail in the extensive livestock system, this effect tends to mitigate the reduction in this type of emissions.

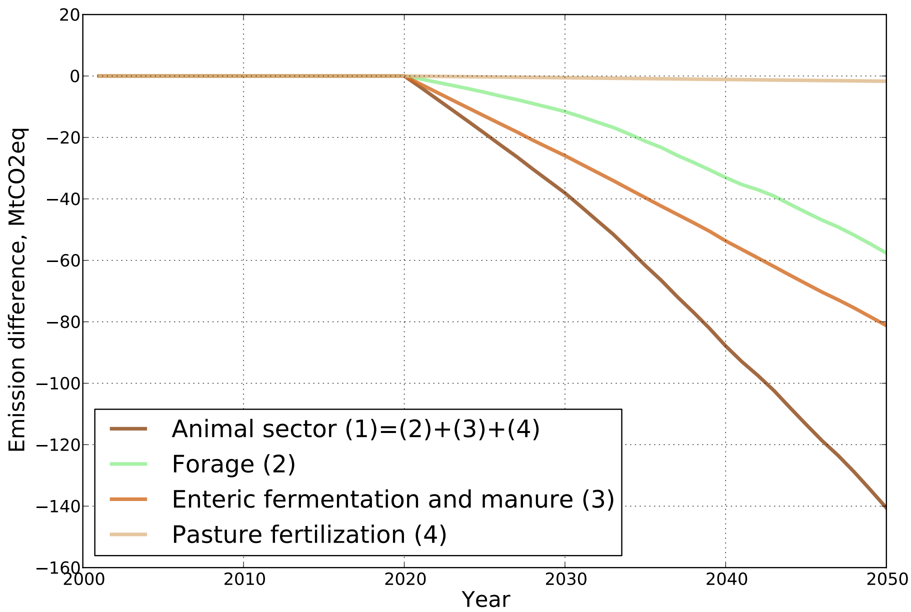


Figure 1.9 – Decomposition of GHG emissions from the livestock sector in the diet shift scenario

1.4.4 Sensitivity of the results

TTo assess the accuracy of our results, we first test their sensitivity to the uncertainty of some key model parameters and then to the uncertainty of the scenario selection.

Depending on the calibration of the NLU model (Souty et al. 2013), two uncertain key variables influence the results in important ways: the accessibility of residual pastures and the initial slope of the yield-fertilizer function. The sensitivity of the decomposition is therefore tested against these two parameters.

First, the parameter of residual pasture accessibility represents the permitted increase in cropland area with regard to residual pastures of high-quality land. In the NLU, the annual conversion rate of residual pasture into intensive pasture/cropland is linearly related to the pressure on land (approximated by the limit between the mixed crop-livestock and pastoral systems), up to a maximum of 5% based on benchmark simulations over 1961-2006 (see Souty et al. (2012) for more details). For our sensitivity test, we increased this maximum to 10% and then decreased it to 1% in two variants. We expected a reduction in the cultivated area in the case of an increase in the conversion rate of residual pastures into crops, as the crops are placed on land with a higher potential yield.

Second, we explored different initial slopes of the production function as a measure of the uncertainty of the intensive margin. The initial slope of the yield-fertilizer function drives the cost of increasing crop yields: the lower the slope, the larger the cost of increasing the yield. A first variant corresponds to the NLU default case in which the yield-fertilizer function parameter change is calibrated to reproduce approximately in 2050 the aggregate crop fertilizer application rate given for 2050 by the Food and Agriculture Organization projections of Alexandratos & Bruinsma (2012). We add a second, more pessimistic variant with no change to the yield-fertilizer function between 2001 and 2050. Given the less efficient use of inputs, we expect the intensification to be lower in this case. The sensitivity analysis of parameter uncertainty reveals a substantial sensitivity of the results to the parameters with results ranging from -186 $MTCO_{2,eq}$ to -525 $MTCO_{2,eq}$. However, the emission reduction remains robust to the uncertainty of the model parameters.

To explain the sensitivity of the results to model parameter uncertainty in the diet shift scenario, we decomposed the variance associated with the uncertainty around the slope of the yield-fertilizer function and the accessibility of residual pastures (Table 1.8) by computing a one-way ANOVA. The uncertainty surrounding the accessibility of residual pastures is responsible for the major part (89%) of the variability in emissions, with the initial slope of the yield-fertilizer function being 11%.

Table 1.8 – Analysis of variance of global emissions in the diet shift scenario

	Df	Sum Sq	Percent of Sum Sq
Accessibility of residual pastures	2	154.9e+17	89
Initial slope of the yield-fertilizer function	1	18.7+17	11

The area of forest is exogenously defined in the NLU based on external scenarios. The agricultural area (cropland + grassland) in each region is set constant in the baseline and our two legume introduction scenarios. In this setting, the lower pressure on land use in the diet shift scenario leads to a decrease in crop intensification and a reduction in emissions, mainly in terms of N_2O. Another possibility could be considered in which the areas spared in the diet shift scenario would be used for reforestation as opposed to decreasing crop production input use per unit area. For this purpose, we consider an alternative scenario with reforestation (referred as “Diet Shift + Reforestation” in the following) and compare it with the diet change scenario without reforestation already presented in details above (referred as "Diet Shift" again) (Table 1.9). The reforestation scenario consists of 9.4 Mha of reforestation in Europe in 2050 and no additional change of forest area in the rest of the world. This reforestation corresponds to the area of pasture land freed in Europe following the reduction in ruminant production equivalent to the reduction in ruminant products demand in the diet change scenario calculated as follows in 2050:

$$Forest_{2050,\mathrm{R}eforest} = Forest_{2050,\mathrm{D}iet\mathrm{S}hift} + \Delta\mathrm{D}_{rumi,\mathrm{E}urope,2050} \times \frac{\beta_{\mathrm{ML},2050} \times \phi_{\mathrm{ML},grass,2050}}{\rho_{\mathrm{ML},grass,2050}} \quad (1.1)$$

with $\mathrm{Forest}_{2050,\mathrm{D}iet\mathrm{S}hift}$ the european forest area in 2050 in the "Diet Shift" scenario (and in the baseline scenario), $\mathrm{Forest}_{2050,\mathrm{R}eforest}$ the european forest area in the "Diet Shift + Reforestation" scenario in 2050, $\Delta\mathrm{D}_{rumi,\mathrm{E}urope,2050}$ the change of ruminant demand in diet shift scenarios in 2050, $\beta_{\mathrm{ML},2050}$ conversion rate of plant calories into ruminant products calories, $\phi_{\mathrm{ML},grass,2050}$ share of the feed of ruminants belonging to the mixed crop-pasture system composed of grass, $\rho_{\mathrm{ML},grass,2050}$ the grass yield of pasture belonging to the mixed crop-pasture system. In Europe, all ruminant are produced in the mixed crop-pasture system. The additional forest change is only based on the 2050 area difference and occurs exponentially between 2020 and 2050.

The combination of a diet shift scenario with a reforestation scenario reduces emissions more than a diet shift scenario alone (respectively -253 $MtCO_2$ vs -211 $MtCO_2$). Popp et al. (2010) found similar results. Furthemore, the combination of these two scenarios changes the distribution of emission reduction sources. In the "Diet Shift + Reforestation" scenario, emission reduction take place mainly through reforestation (-53%), while in the "Diet Shift" scenario without reforestation, the emission reduction mainly occurs through a reduction in the non-CO_2 emissions (-99.5%). In a diet shift scenario, carbon sequestration resulting from reforestation therefore occurs at the expense of a reduction in non-CO_2 emissions.

Table 1.9 – Distribution of emission changes among the main sources of mitigated emissions in the diet shift scenario with and without deforestation, compared to a reference scenario without diet shift and without reforestation. A positive emission difference means an increase in emissions in the diet shift variants compared to the reference scenario (no diet shift and no reforestation).

Emission difference ($MtCO_2$)	Diet shift	Diet Shift + Reforestation
Total mitigated emissions	-211 (100%)	-253 (100%)
Global emission difference:		
- by BNF	-1.0 (0.48%)	-2.1 (0.8%)
- due to plant fertilization for human food	-65 (31%)	-16 (6%)
- due to legume residues	0.23 (-0.11 %)	0.59 (-0.23%)
- due to fodder fertilization	-58 (27%)	-18 (7%)
- due to enteric fermentation and manure management	-81 (38%)	-77 (30%)
- due to LUC	-1 (0.46%)	-134 (53%)
- due to other sources*	-5.5 (2.6%)	-6 (2.6%)
European mitigated emissions	-81 (39%)	-216 (85%)
- by BNF	-3.8 (1.8%)	-3.5 (1.4%)
- due to plant fertilization for human food	-10 (4.8%)	-4.5 (1.8%)
- due to legume residues	0.7 (-0.34%)	0.7 (-0.3%)
- due to fodder fertilization	-8.5 (4%)	-4.3 (2%)
- due to enteric fermentation and manure management	-59 (28%)	-69 (27%)
- due to LUC	0.2 (0.07%)	-134 (53%)
- due to other european sources*	-5.5 (2.6%)	-1.6 (0.6%)

* other sources include CH_4 from rice cultivation and N_2O from pasture fertilization

1.5 Comparison of the diet shift scenario with other studies

To further evaluate our results regarding the diet shift scenario, we compare them with similar studies and present the results in Table 1.10: two global scale land-use models, MAgPIE and IMAGE (Popp et al. 2010, Stehfest et al. 2009), one scenario-based model at the European scale (Westhoek et al. 2015) and one global balance model, GlobAgri-WRR (Ranganathan et al. 2016). To obtain a common measure of ruminant substitution in these studies, ruminant meat change was converted to ton of dry matter (tDM) for energy content and protein content following FAO and WHO (2001). The aim of this comparison is not to provide an exhaustive review of studies on dietary changes but rather compare the results obtained using various models of agricultural production.

Emissions per quantity of substituted ruminant span a very wide range between 1.65 and 0.025 $MtCO_{2,eq}$/ktDM. Part of this variability stems from the diversity of representations in the land-use change models (and the associated carbon sequestration) induced by the change in diet. For example, carbon sequestration is much greater in GlobAgri-WRR (98% of emission change) where the land-use change emissions result from the conversion of cropland into forests or savanna. By contrast, in the NLU and MAgPIE models (with respectively 0.5% and 0% of emission change), land-use change emissions either result from the conversion of cropland into pasture alone or are not taken into account (Table 1.10).

The variability of emission factors mitigated by the amount of substituted ruminant also derives from the non-linearity of the emission reduction with the amount of substituted ruminant. The concavity of the abatement curves as a function of mitigation effort (Frank et al. 2018) partly explains why studies with substantial dietary changes (here Westhoek et al. 2015 and MAgPIE) have lower abatement ratios per amount of substituted ruminant (0.025 and 0.03 respectively).

Table 1.10 – Comparison with other livestock substitution studies

Study	Emission type	Emission change $MtCO_{2,eq}$	Ruminant change	Ruminant change ktDM	Emission change $MtCO_{2,eq}$/ktDM
NLU	Production	211		2999	0.07
	LUC	1			
GlobAgri-WRR	Production	299	33% of ruminant in diet	4180	0.865
	LUC	3319			
Westhoek et al. 2015	Production	143	-50% beef and dairy, greening	6479	0.025
	LUC	25			
MAgPIE	Production	4000	Mediterranean diet	130000	0.03
IMAGE	Production	2100	Healthy diet	2300	1.65
	LUC	1700			

1.6 Conclusions

In this study, we assess the efficiency of increasing the use of legumes to reduce GHG emissions. The first key result concerns the difference in magnitude between the supply-side scenario (feed mix scenario) and the demand-side scenario targeting a reduction in the demand for ruminant products (diet shift scenario), with mitigated emissions being 20 greater in the latter compared to the former. The main environmental benefit of legumes is therefore providing proteins that can substitute animal products rather than enabling a lower consumption of synthetic fertilizer through increased leguminous nitrogen fixation.

Focusing on the diet shift scenario, most of the emission reduction results from the livestock sector through fodder fertilization on the one hand and enteric fermentation and manure management on the other. Let us note that a diet shift to leguminous crops is particularly effective in cutting this latter category of emissions in Europe, where it is a major source of emissions in the agricultural sector.

This study also stresses the pivotal role played by international trade, since the emission reduction in the diet shift scenario mainly takes place outside Europe. The mechanism at stake follows the same logic as that of indirect land-use changes for biofuel production (Searchinger et al. 2008), in which emissions are exported toward a third region. In the legume case, Europe re-imports emissions by reducing its land needs and improving its trade balance.

Finally, we showed that most of the emission reduction in the plant food farming sector results from reduced fertilization, mainly linked to economic choices regarding production allocation and intensification levels. This emphasizes the reduced role of the substitution of mineral nitrogen by BFN in legume production. It is noteworthy that the NLU is pessimistic on the effect of intensification on nitrogen use and N_2O emissions. Reductions of N_2O emissions in the diet shift scenarios are therefore likely in the high end of the spectrum.

Our results are robust to our main modeling assumptions, it is although important to note that the type of mitigated emissions (CO_2 or N_2O) is sensitive to the way in which spared lands are used. In our study, the lower pressure on land use in the diet shift scenario leads to a decrease in crop intensification and a reduction in emissions, mainly in terms of N_2O. In a combinaison of a reforestation scenario and a diet change scenario, the emissions reduction mainly occur through a reduction in CO_2 emissions (54%) and the mitigation of no-CO_2 emissions is reduced compared to a diet change alone scenario.

Despite our efforts to integrate different elements of the food system, some elements influencing the reduction in GHG emissions via legume introduction in Europe remain out of the scope of this study. One limitation is our focus on one part of the production chain: notably, the production of primary agricultural products. Up in the production chain, fat removal is associated to important losses for meat, especially in terms of energy. Meat also requires low-temperature storage, while legumes are dry products that require long cooking time. These differences, not taken into account in this study, could substantially influence the emission gain to substitute animal products for legumes as described in this study, but the direction is ambiguous (Poore & Nemecek 2018).

Finally, while our study includes a few drawbacks of legume production (increased nitrogen leaching of legume residues and lower average yields of leguminous crops compared to cereal crops), it does not account for the higher interannual variability in legume yields compared for instance to wheat yields in Europe (Cernay et al. 2015).

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Supplementary material

1.6.1 Modelling of international trade

The NLU model incorporates a simple representation of international trade based on relative regional prices. Imports and exports of plant and ruminant products are computed using a pool representation. In line with our observations, this representation allows a region to simultaneously import and export one category of goods, while other countries facing different production costs may be present on the market. The objective of this modeling is to capture basic adjustments to the changes in terms of trade (i.e., the ratio between domestic and international prices). However, we acknowledge that we may have overlooked some important features such as regional specialization. See (Souty et al. 2012) for more details (including equations).

1.6.2 Computation of changes in the livestock system

The NLU model represents the link between the plant and livestock production sectors in terms of the manure used to fertilize cropland, the crops used to feed livestock, and the competition between crop/pastureland and agricultural land. In the NLU, livestock can be produced through pastoral or mixed crop-livestock systems (Bouwman et al. 2005). In the pastoral system, livestock is only grass-fed, while in the mixed crop-livestock system, livestock is fed with grass, food crops, crop residues, and fodder. Pastures in the pastoral system are set on low-quality land, while those in the mixed crop-livestock system are set on higher-quality land in mosaic with cropland in the NLU.

The base year data shows that a fraction of the extensive production remains in high-quality land (Ramankutty et al. 2008). Market imperfection and lack of market accessibility can prevent the intensification of such land (Merry et al. 2008). These pastures are called residual pastures. The conversion of these residual pastures into the mixed livestock-crop system is determined by a conversion coefficient known as the accessibility parameter in this study.

1.6.3 Baseline information

At the baseline, the main features follow the shared socioeconomic pathway 2 (SSP2), adapted for the NLU model. The human food demand is calculated from the population scenario of socio-economic pathway 2 (SSP2) (Popp et al. 2017) and the consumption per capita from diet projections of FAO (Alexandratos & Bruinsma 2012). The agrofuel production scenario described by Alexandratos & Bruinsma (2012) is also added to the demand. The energy prices are taken from the computable general equilibrium called IMACLIM (Waisman et al. 2012).Forest areas are kept constant.

Chapter 2

Impact of land-use-based climate mitigation policies on biodiversity and food security

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This article has been written in a letter format with short description of method in the main text. A detailled description of the different steps of the methodology is provided in supplementary information of this section and in the chapter called "The modelling framework".

Abstract

Agriculture faces three great challenges: feeding a growing population, reducing its impact on biodiversity and minimizing greenhouse gas (GHG) emissions. Therefore, it is important to assess synergies and trade-offs in meeting these challenges. In this paper, we evaluate a broad range of scenarios that achieve 4.3 $GtCO_{2,eq}$/year GHG mitigation in the Agriculture, Forestry and Other Land-Use (AFOLU) sector by 2100. Scenarios include varying mixes of three GHG mitigation policies: biofuel crop production, dietary change and reforestation of pasture. We evaluated the impacts of these scenarios on food security and biodiversity conservation. We find that focusing mitigation on a single

policy can lead to positive results for one indicator, but with significant negative side effects on others. For example, mitigation dominated by reforestation favors biodiversity criteria, but is projected to lead to sharp increases in food prices. Mitigation scenarios focusing on biofuels have strong adverse effects on both biodiversity and food security indicators. A balanced portfolio of all three mitigation policies, while not optimal for any single criterion, minimizes trade-offs by avoiding large negative effects on food security and biodiversity conservation. At the regional scale, the projected impact of mitigation policies are similar to proection at global scale, except for Canada and Middle-East. Due to the small area of agricultural land in these regions, their average regional levels of biodiversity are mainly influenced by the state of their natural areas and not by agricultural land-use changes.

Keywords: Mitgation policies | Global land-use system | Biodiversity | PREDICTS

2.1 Highlights

- A mix of dietary change and reforest as global mitigation strategy improves biodiversity and food security compared to a baseline without mitigation.
- A large scale second-generation bioenergy production reduces both biodiversity (especially species richness) and food security compared to a baseline without mitigation.
- The use of several indicators is essential to take into account the impact of emission mitigation strategies in the AFOLU sector on the different aspect of biodiversity.
- The regional impact of mitigation strategies in the AFOLU sector on biodiversity depends mainly on regional agricultural areas.
- In this study, we provide a means of dialogue between the reaching climate objectives in the AFOLU sector and assessing impacts of this on biodiversity

2.2 Introduction

Land is a multi-purpose asset that may involve conflicts in its use. Formerly restricted to the local level, global conflicts have emerged over the last few decades because of the rapid intensification of international exchanges (Liu et al. 2013). Today, the joint challenges of global food security, climate change mitigation and conservation of biodiversity give a new dimension to this issue, involving new types of trade-offs and synergies while strenghtening the global dimension.

Assessments based on global land-use models show that mitigation policies relying on large-scale biofuel production have important environmental implications as well as adverse impact on food prices especially if forest protection measures are implemented (Humpenöder et al. 2018, Heck, Gerten, Lucht & Popp 2018). Afforestation is also associated with significant increase in food prices (Kreidenweis et al. 2016) while dietary change policies may have the opposite effect, with a reduction in the price of calories when implemented (Stevanović et al. 2017). Combining measures appears to be an appropriate solution to minimize negative effects, but the nature of the combinations does matter (Humpenöder et al. 2018, Obersteiner et al. 2016, Visconti et al. 2016).

This picture can be made more complex by taking into account the trade-offs between biodiversity and climate mitigation. While some mitigation policies such as carbon storage in forests can maintain biodiversity when appropriately implemented (Watson et al. 2018), other options could increase pressure on biodiversity indices. Mitigation policies in integrated scenarios of climate change (Representative Concentration Pathways) and human development (Shared Socio-economic Pathways) seem to be mostly harmful to biodiversity (Hill et al. 2018) with regard to numerous indicators and at the level of biodiversity hot-spots (Jantz et al. 2015). Ambitious mitigation scenarios involving substantial land use change or scenarios with strong climate change are particularly associated with high impacts on biodiversity (Newbold 2018).

This study provides a unique framework for understanding (i) the impact of different GHG mitigation policies (biofuel production, dietary change and reforestation of pastures) on both biodiversity and food security and (ii) the degree of conflict or synergy between such policies.

The food system (Erb et al. 2017) is represented by the Nexus Land Use (NLU) model (Souty et al. 2012). This global agricultural intensification model describes the worldwide land-use system, computes cost-optimal food security indicators (average cost of production per calorie produced and food price per calorie produced), calculates associated agricultural and land-use change wuth respect to GHG emission goals and generates land-use maps.

The PREDICTS models are used to convert these land-use maps into impacts on biodiversity through computation of the local Biodiversity Intactness Index (BII) (Scholes & Biggs 2005) and Species Richness indicator (SR) using a mixed-effect modelling structure (Hill et al. 2018). BII is an indicator of ecosystem naturalness and measures the proportion of species present in the ecosystem that are similar to the natural reference ecosystem. The SR reports the number of species present in the ecosystem. These two indicators are complementary because they provide an insight into the overall health of the ecosystemś specific diversity and the type of biodiversity present. To clarify the impacts of GHG mitigation policies on these indicators, we make some changes to the framework presented by Hill et al (2018) (Hill et al. 2018) by separating rangeland from other pasture and representing grassy and woody biofuel crops as highly intensified perennials.

With this framework, we assess the impact on biodiversity and food security of land-use-based mitigation scenarios that provide mitigation of 4.3 $GtCO_2$/year in 2100 (target for the AFOLU sector to reach 2° of global warming obtained by extrapolating the 2030 results to 2100 (Wollenberg et al. 2018)). To mitigate these 4.3 $GtCO_2$/year in 2100, we build scenarios that are combinations of second-generation biofuel production (between 0 and 112 EJ/year in 2100), dietary change (reduction of the proportion of animal products in food down to 314 kcal/cap/day except in Africa for nutritional reasons) and reforesting pastures (between 0 and 31% of global pasture reforested). The mitigation effort of each of these policies (second-generation biofuel production, dietary change and reforestation of pastures) is then defined as the percentage of each policy in total mitigated emissions (See Section. 2.6.2 in supporting information). To cover a broad range of scenarios and represent a uniform distribution of mitigation policies (biofuel dietary change and reforestation), the scenarios are constructed according to a complete factorial plan (See Section. 2.6.3 in supporting information). The experimental design involves taking mitigation efforts ranging from 0 to 100% for each policy in 10% steps while keeping the sum of efforts equal to 100%.

We infer from these scenarios whether the relationship between biodiversity and food security in the presence of mitigation policies is synergistic or antagonistic and how the policy mix influences this relationship.

Finally, we detail the distribution of these impacts across 12 large regions of the world. In this study, the mitigation effort is unequally distributed among the regions but depends on the amount of pasture to reforest, the current diet and the regional cost of second-generation biofuel production. To compare the impacts of these heterogeneous mitigation efforts between regions and with the global figures, we calculate the relative change in biodiversity and food security divided by the relative change in regional emissions (See supporting information for details of these indicators).

This downscaling highlights the influence of the regional context on the sensitivity of regional biodiversity and food security responses to mitigation policies.

2.3 Material and methods

2.3.1 Estimating agricultural production

The global land-use model known as NLU is used to represent the agricultural sector (See (Souty et al. 2012) for more details). It allows us to represent agricultural intensification and the distribution of cropland, pastures and forest at the global scale. Crop intensification is explicitly represented in the NLU with a concave production function and fertilizer prices are computed from energy prices (Brunelle et al. 2015). Two livestock systems are considered: a grass-based system and a mixed crop-livestock system.

Regional production cost is minimized under a supply-use equilibrium with a simplified representation of international trade. Based on an interpretation of the Ricardian theory, the boundary between the mixed crop-livestock system and the grass-fed livestock system changes according to the equalization of rent. In the mixed crop-livestock system, cropland distribution is based on potential yield, with rent increasing with land quality. In this model forest area is exogenously defined by scenarios.

2.3.2 Estimating agricultural emissions

Agricultural emissions are calculated by the NLU model using the IPCC Tier 1 method for production in the plant food sector and the IPCC Tier 2 method for the livestock sector (IPCC 2006*a*).

In the livestock sector, emissions from manure management (CH_4 and N_2O) and enteric fermentation (CH_4) are computed. In the plant food sector, emissions from fertilization(N_2O) and rice cultivation(CH_4) are computed. Carbon dioxide (CO_2) emissions are also computed for land-use changes based on Le Quéré et al. (2009) and for fossil fuel substituted by second-generation biofuel (detailed in the description of biofuel scenarios in supporting information).

2.3.3 Estimating biodiversity impacts

Biodiversity impacts are estimated by the PREDICTS modeling framework (Purvis et al. 2018) which considers land-use to be the main driver of biodiversity losses (Foley et al. 2005).

The statistical models linking biodiversity to drivers are underpinned by a large, global and taxonomically broad database of terrestrial ecological communities facing land-use pressures (Hudson et al. 2017). Among the biodiversity models provided by the PREDICTS framework (Purvis et al. 2018), we chose BII because of its use in the Planetary Boundaries framework (Steffen et al. 2015) and SR because of its wide use despite its known limitations. The species richness model (SR) is a mixed-effect model computing the number of species present. The total abundance model computes the sum of all individuals of all species presentin the ecosystem. The compositional similarity model computes the percentage of individuals common to the studied ecosystem and the reference ecosystem (De Palma et al. 2018) for each grid of a 0.5° map. The abundance map was then multiplied by the compositional similarity map to produce the map of abundance-based BII (Newbold et al. 2016). These three PREDICTS models include different levels of management (intensive, light or minimal) and different types of land cover (forest, pasture, rangeland, annual cropland, perennial cropland and urban zones).

2.3.4 Estimating the link between PREDICTS and NLU

In the NLU, 60 land classes are defined in the reference year according to their potential yield (Brunelle et al. 2018). Different crop types are defined for each land-class: “Dynamic” crops and “other” crops (See supporting information). In PREDICTS, three levels of intensification break down perennial crops, annual crops and nitrogen-fixing crops into a “minimal”, “light” and “intense” use category (Hudson et al. 2014). NLU crop types are aggregated into a single category and then split into PREDICTS crops categories (perennial, annual and nitrogen-fixing crops) based on their relative proportion of the crop mix in the reference year. In the reference year, a Generalized Additive

Model (GAM) is computed to match the relative proportion of “minimal”, “light” and “intense” cropland with the 60 NLU land classes (See supporting information, Fig. 7). Pastures in the NLU mixed crop-livestock and pastoral production systems are aggregated into a single pasture category. In PREDICTS, pastures include rangeland, “light” and “intense” pastures. Among the aggregated pasture category of NLU, rangeland areas are defined on the basis of the rangeland map produced by Hurtt et al. (2011). For the remaining pastures, livestock density is defined on the basis of livestock density maps produced by Robinson et al. (2014). In the reference year, a GAM is computed to match the relative proportion of “light” and “intense” pasture with livestock density maps (See supporting information, Fig. 8).

2.3.5 Estimating the baseline

The population follows changes in the Shared Socio-economic Pathway (SSP2) (Riahi et al. 2017). Food demand follows FAO projections (Alexandratos & Bruinsma 2012) with a global mean consumption in 2100 of 2585 kcal/cap/day of vegetable products and 615 kcal/cap/day of animal products. International trade parameters are kept constant. The forest, which is exogenous in the model, follows current trends described in Hurtt et al. (2011) until 2050 and then stabilizes. Fertilizer prices are computed using the method described in Brunelle et al. (2015) based on energy prices taken from the baseline of IMACLIM-R (Waisman et al. 2012).

2.3.6 Mitigation scenarios to achieve 2°C of global warming in 2100.

We combine 3 mitigation policies in mitigation scenarios to achieve 4.3GtCO_2/year of mitigated emissions in 2100 (the target for the AFOLU sector to achieve 2° of global warming according to an extrapolation of the 2030 results to 2100 Wollenberg et al. (2018)).

To obtain a broad representation of the possible combinations between second-generation biofuel production, dietary change and reforestation, we use a complete factorial design (See Fig. 2.4 in supporting information) which covers second-generation biofuel production of between 0 and 112 EJ, animal product consumption of between FAO trends (Alexandratos & Bruinsma 2012) and a convergence towards 432 kcal/cap/year (See supporting information, Table.S2.1), and pasture reforestation of between 0% and 31% (See supporting information, Table.S2.2). To achieve 4.3 GtCO_2 of mitigated emissions by means of dietary change, we replace the consumption of animal products by plant products in the Agrimonde scenarios called AG1 (Paillard et al. 2011). This leads to a convergence of the overall animal consumption towards 432 kcal/cap/day in all regions. The consumption of ruminant products obtained is 183 kcal/cap/year in 2050 for Brazil, Canada, Europe, USA, FSU, OECD Pacific and Rest of LAM, 91 kcal/cap/year in 2050 for India, Rest of Asia and China, 154 kcal/cap/year for Middle-East and 65 kcal/cap/year for Africa. The rest of animal product consumption (in the 432 kcal/cap/day) is composed of monogastric and aquatic products (See supporting information, Table.S2.1).

The reforestation scenario follows the same philosophy as the natural climate solutions reforestation scenario presented in (Griscom et al. 2017) by reforesting pastures. The figure of 31 % of pastures reforested in the world corresponds to a reduction of 4.3 $GtCO_2$ of mitigated emissions by the AFOLU sector in 2100 (See supporting information, Table.S2.2). In Europe and the USA, second-generation biofuels are produced in the form of grassy crops; in the rest of the world they are woody crops.

2.4 Results

2.4.1 Trade-off between biodiversity and food security

The scatter of points representing the impacts of land-use mitigation scenarios is widely spread over the output space and has concave boundaries, indicating a moderate trade-off between biodiversity and food security for a given climatic objective (Fig. 2.1 and see supporting information for other indicators).

Scenarios with high second-generation biofuel production are located largely within the envelope indicating that second-generation biofuel production is a less effective mitigation option for reconciling biodiversity and food security objectives than scenarios containing more reforestation or dietary change (Fig. 2.1 and see supporting information for other indicators). Moreover, scenarios with low levels of biodiversity (especially low SR) are linked with scenarios including high levels of second-generation biofuel production (Fig. 2.2).

Mitigation scenarios focusing on dietary change or reforestation are at one edge of the envelope, indicating that they are performing well in relation to one indicator but have negative side effects on at least one of the other indicators (Fig. 2.2). The reforestation of large proportions of the worldś pastures is beneficial to biodiversity whichever indicator is chosen, but causes a sharp increase in food prices and food cost, thus threatening food security (Fig. 2.2). On the contrary, scenarios with significant dietary changes have a lower performance in terms of biodiversity but have lower impacts on food prices and food production costs (Fig. 2.2).

Finally, it should be noted that some mitigation scenarios (mainly involving reforestation and dietary change) can improve the protection of biodiversity and food security in 2100 compared to a scenario without mitigation policies (scenarios in the upper left-hand quadrant of the Fig. 2.1).

2.4.2 Portfolios of land-use-based mitigation scenarios reduce the trade-off between biodiversity and food security

On a global scale, mitigation scenarios that spread mitigation efforts between several policies (reforestation, second-generation biofuel production and dietary change) avoid extreme negative side effects. Scenarios with higher levels of biodiversity and food security than the baseline are mainly mixes of reforestation and dietary change associated with low second-generation biofuel production. For example second-generation biofuel production of 10 EJ/year in 2100 (10% of the mitigation effort) associated with reforestation of 11 % of pasture (40% of the mitigation effort) and animal consumption of 150 kcal/cap/day (50% of the mitigation effort) decrease the food price by 13% compared

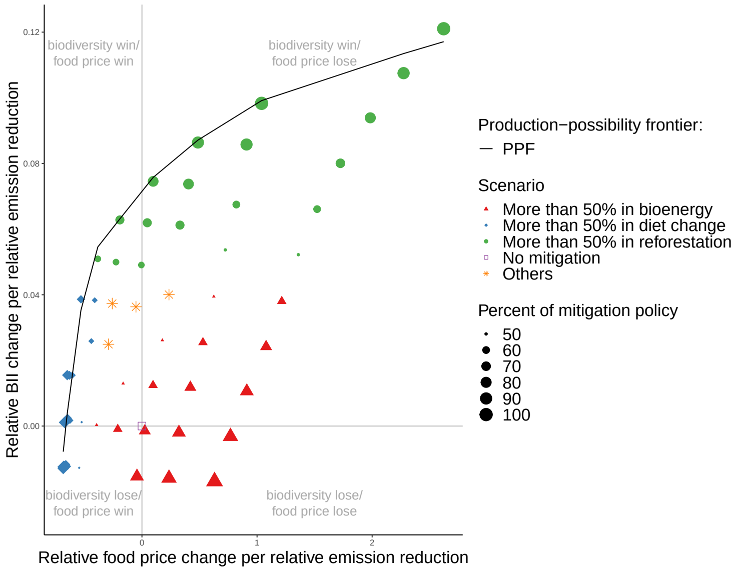

Figure 2.1 – Impacts of mitigation scenarios achieving 4.3 $GtCO_{2_{eq}}$ of mitigated emissions in 2100 based on combinations of second-generation biofuel production, dietary change and reforestation. Outputs are presented as the relative change in BII and food price with respect to the scenario without any mitigation policy (baseline) with respect to the relative change in mitigated emissions. The relative changes in BII and food prices can be deduced from this graph by multiplying the values obtained by the relative change in emissions for each scenario, which is constant at $\frac{\text{Mitigated emissions}}{\text{Baseline emissions}} = \frac{4.3\text{GtCO}_{2_{eq}}}{13.87\text{GtCO}_{2_{eq}}} = 0.3$ The mitigation effort of each policy (second-generation biofuel production, dietary change and reforestation) is expressed in the legend as the percentage of mitigated emissions due to the policy in total mitigated emissions. “Others” in the legend represents scenarios without an option accounting for more than 50% of the mitigation effort.

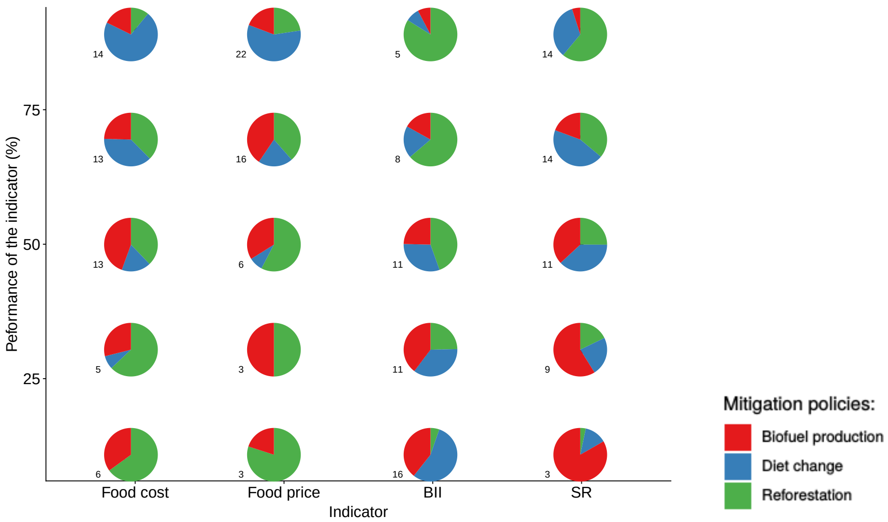

Figure 2.2 – Influence of the distribution of mitigation effort between reforestation, biofuel production and dietary change on biodiversity (SR and BII) and food security (food price and food cost) indicators. Each indicator is linearly rescaled between 0 and 100. We group the mitigation scenarios into quintiles according to their impact on the indicator and calculate for each quintile the average percentage of mitigation achieved by biofuel production, dietary change and reforestation. Because averages are used, it cannot be deduced from this graphic that a mix of mitigation policies is optimal.

to the baseline and increase BII by 1.2% compared to the baseline (Fig. 2.1 and see supplmentary information for other indicators).

2.4.3 Trade-off and synergies between food security and biodiversity conservation in mitigation policies at the regional scale

The trade-offs and synergies between biodiversity conservation and food security protection observed at the global level can be found in most regions of the world. Former Soviet Union countries (FSU), the Rest of Latin AMerica (LAM) and Brazil are exceptions as they present a synergetic relationship between SR and food security indicators under mitigation scenarios (Fig. 2.8 and Fig. 2.10 in supporting information). In this case, dietary change is the optimal policy whichever indicator is considered.

However, regional contexts affect the influence of mitigation strategies on the protection of biodiversity and food security. Canada and the Middle-East are subject to limited changes in their biodiversity indicators(Fig. 2.3). Due to the small area of agricultural land in these regions (Hurtt et al. 2011), their average regional levels of biodiversity are mainly influenced by the state of their natural areas and not by agricultural land-use changes (Fig. 2.3). To reduce malnutrition in Africa, the dietary change mitigation scenario consists of increasing consumption of animal products, unlike other regions (See Table. 2.1). This particular dietary change scenario explains the high levels of biodiversity in this region with significant dietary change (Fig. 2.3). Finally, in India, any reduction or increase in pressure on land and the agricultural system through a constraining mitigation policy significantly influences biodiversity and food security (Fig. 2.3).

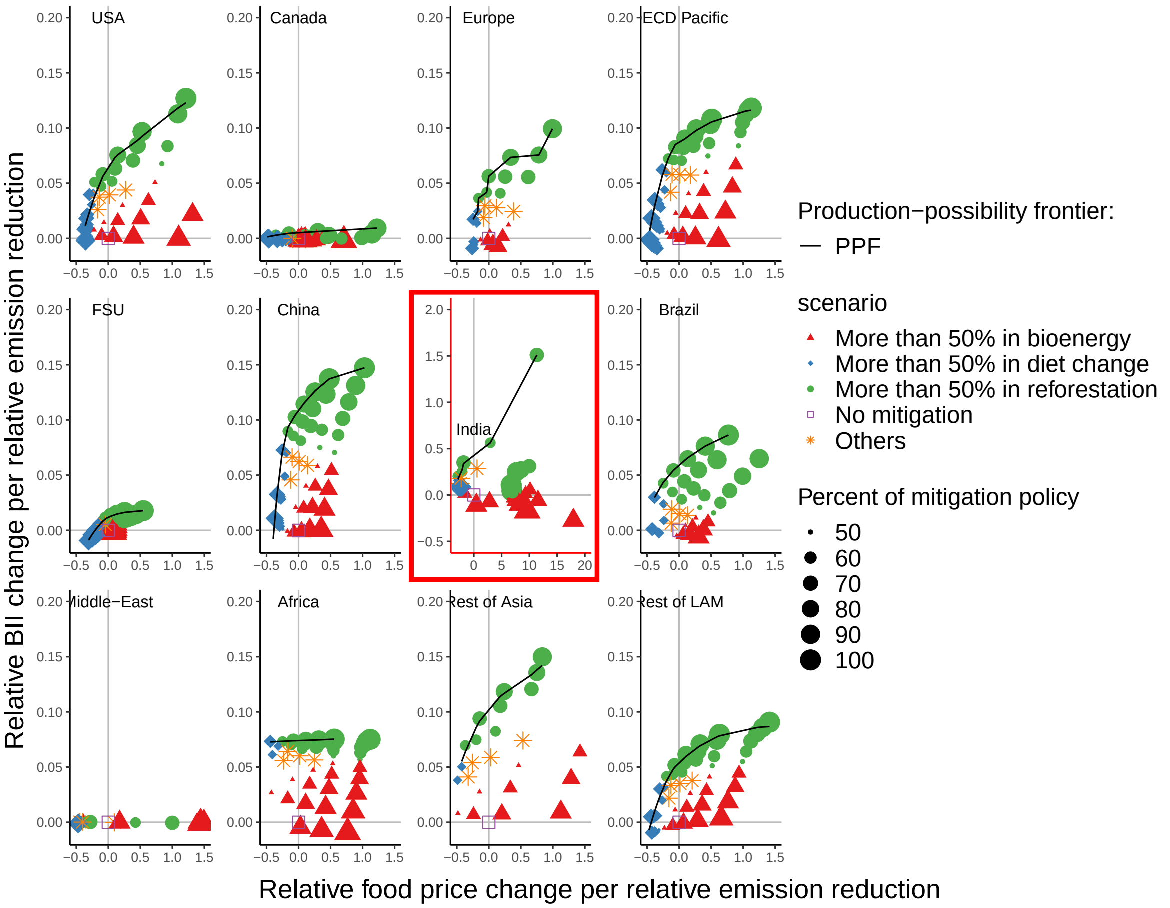

Figure 2.3 – Relative change in BII and food price with respect to relative change in GHG emission reduction at the regional level. This ratio takes into account the different emission changes within a region from one mitigation scenario to another and the unequal distribution of mitigation efforts between regions within a scenario. A relative change in BII of 0.2 therefore indicates that a 10% reduction in regional emissions means a 2% reduction in biodiversity. To compare the regions with each other, a common range is chosen for the axes of each region. An unzoom is provided for India with extreme BII and food indicator change for mitigation scenarios (red rectangle). These indicators are also privided at the global scale in Fig. 2.1. Similar graphs for other biodiversity and food security indicators are provided in supporting information.

2.5 Discussion

2.5.1 Impacts of mitigation scenarios on biodiversity conservation objectives

The major contribution of this study is to represent not only the impact of mitigation policies on habitats of high ecological value such as "biodiversity hot spots" (Obersteiner et al. 2016) or forests (Humpenöder et al. 2018), but also to represent the impact of agricultural intensification and land use changes within the agricultural sector (conversion of the pastoral system into a mixed path-and-crop system). For example, the inclusion of the impact of agricultural intensification on biodiversity in this study mitigates the BII increase resulting from a reforestation scenario by taking into account the impact of the intensification resulting from this forest scenario (Stevanović et al. 2017). Also, the reduction in extent of the crop-pasture mix system in favour of the pastoral system in scenarios of significant dietary change has consequences for biodiversity, as evidenced by the reduction in BII (Fig. 2.1).

Another major interest of this framework is to study the impacts of different land-use-based mitigation scenarios on different biodiversity values: (i) the "naturalness" of ecosystems through the BII and (ii) the "extirpation risk" through the BII and SR according to the classification described by Karp et al. (2015). By making assumptions about the ecological functions provided by new individuals in non-primary ecosystems, the BII also makes it possible to estimate the risks of loss of ecosystem services previously provided by the replaced biodiversity (Newbold et al. 2016). Combined with the extinction risk studied by Obersteiner et al. (2016) through global biodiversity hotspots, reforestation scenarios are beneficial to these three indicators, second-generation biofuel is detrimental to these three indicators and decreasing pressure on land through dietary change has a beneficial effect on SR and biodiversity hotspot preservation but decreases BII due to an increase in the area of pasture.

In addition, the inclusion of the impacts of these policies on biodiversity is a first step towards a deeper integration of biodiversity into the socio-ecological system used in environmental assessment of mitigation options. The crucial role of biodiversity in food production is well established and its integration can significantly change the relationship between biodiversity protection and food security (FAO 2019).

2.5.2 Trade-off and synergies between food security and biodiversity conservation under mitigation scenarios

A portfolio of mitigation strategies reduces side-effects on biodiversity and food security compared to siloed strategies and allows several SDGs to be achieved simultaneously (Bertram et al. 2018, Humpenöder et al. 2018, Minx et al. 2018, Obersteiner et al. 2016). For example reforestation of 22% of pasture (70% of the mitigation effort) and a dietary change of 90 kcal/cap/day (30% of the mitigation effort) is the best scenario to minimize the worst criteria among biodiversity, food security and mitigation in the agricultural sector at the global scale. The portfolio effect is explained in this scenario by the complementarity of mitigation policies. The synergy is particularly strong between dietary change and reforestation strategies, as this combination allows for land to be spared through a reduction in overall food production and using that land both for storing carbon and preserving biodiversity (Herrero et al. 2016, Stevanović et al. 2017, Ewers et al. 2018). On the other hand, the increase in second-generation biofuel production reduces the positive synergies between food security and biodiversity conservation even with an optimistic assumption about the quantity of emissions reduced per unit of second-generation biofuel produced compared to Searchinger et al. (2018).

2.5.3 Regional impacts of mitigation policies on biodiversity and food security

In this study, mitigation effort is allocated between regions according to reforestation potential, biofuel prices and the difference between local diet and a reference diet without taking into account the equitablity or mitigation cost of this distribution of the mitigation effort. The relationships between biodiversity and food security established in this study could change when these allocation criteria are taken into account. Moreover, the potential for mitigation of emissions, food insecurity and biodiversity loss in the AFOLU sector, although very high (Tubiello et al. 2015, Heck, Hoff, Wirsenius, Meyer & Kreft 2018, Tilman et al. 2017), may not be exploited due to equitability of the allocation of effort or high mitigation costs (van den Berg et al. 2019, Markel et al. 2018, Tilman et al. 2017).

In this study, we show the importance of taking into account the regional context, which strongly nuances the trade-offs between biodiversity protection and food security protection on a global scale. This study should therefore be complemented by other mitigation scenarios that take into account the regional context more specifically, such as soil carbon sequestration (Lal 2004) in regions with degraded soils such as southern Europe, some parts of Asia and Africa, or increased Nitrogen Use Efficiency (NUE) (Zhang et al. 2015, Bodirsky et al. 2014) in regions with low NUE such as China or India.

2.5.4 Scenarios in the policy agenda

In this study, we show the importance of going back and forth between exploratory and target-seeking scenarios to include new objectives as we have done here with biodiversity. In the literature, climate scenarios are currently at the target-seeking scenario stage according to the framework proposed by Pichs-Madruga et al. (2016) while global biodiversity impact scenarios are still exploratory scenarios. Here we do seek to quantify exploratory scenarios without sticking to a cost-efficiency criterion that would lead to choosing the scenario with the lowest implementation cost. This approach allows the assessment of a wider variety of combinations of mitigation policies than optimized mitigation scenarios and does not make implicit assumptions about preferences between biodiversity and food security. For example, the RCP2.6 scenario proposed in Vuuren et al. (2011) implies that an important part of the mitigation effort (equivalent to 181 Ej) is assumed by second-generation biofuel production. The rest of the mitigation effort is shared between dietary change, reforestation and a carbon tax on agricultural emissions. This cost-optimal approach leads to relatively low food prices at the expense of low SR levels (See Fig. 2.3). The negative effect on biodiversity is mainly due to the significant production of second-generation biofuel (Hill et al. 2018, Jantz et al. 2015).

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2.6 Supporting information

A detailled description of the modelling framework is provided in chapter.

2.6.1 Indicators

We use four indicators to represent impacts of mitigation policies on biodiversity and food security:

— Global food price ($/Mkcal): The price of food is used here as an indicator of the extent to which global food demand is satisfied by production. This indicator is calculated by taking the output-weighted average of regional prices. There is no price equalization across regions in NLU because trade rigidities constrain the regional supply.

— Crop production cost per unit of food energy produced: food production costs include (i) fertilizer and pesticides which are substitutable to land according to relative prices and (ii) labor and capital (excluding fertilizer and pesticides) which are complementary inputs for each hectare of land.

— Species richness: We focus on species richness because of its wide use and easy interpretation. Local species richness is calculated by projecting a model linking the intensity and the different land-uses onto a world map ($0.5° \times 0.5°$ grid cell) with this indicator (see Table. 3.13 for a presentation of the coefficients of the model). The species richness model is based on between-site comparisons of ecological assemblage composition collated from the literature as part of the PREDICTS project (Hudson et al. (2014)). Random effects in PREDICTSś models accounted for study-level differences in response variables and sampling methods, and for the within-study spatial arrangement of sites.

— Biodiversity intactness index (BII): As defined in Newbold et al. (2015), the BII-abundance indicator results from the multiplication of abundance by the change in composition due to change in land uses and the change in intensity of these uses. It allows us to take into account the effects of human activities on the replacement of original species by newcomers (Dornelas et al. 2014)

2.6.2 Mitigation scenarios

Definition of the mitigation effort

The mitigation effort provided by a mitigation policy (reforestation, dietary change or second-generation biofuel production) is the proportion of emissions mitigated by that policy. The sum of the mitigation efforts of the 3 policies in a scenario is therefore 100% by addition. These mitigation policies interact with each other within the food system when implemented simultaneously. The attribution of given mitigated emissions to one specific mitigation policy is therefore not straightforward. For example, second-generation biofuel production increases the pressure on the food system through an increase in the area under cultivation and an increase in yield (Brunelle et al. 2015). The simultaneous deployment of a pasture reforestation policy also increases the pressure on the agricultural system, which is also reflected in an increase in yields. The attribution of emissions to different mitigation policies (reforestation, dietary change or second-generation biofuel production) is therefore carried out ex-post.

First, we calculate the emission mitigation factor per unit of forest area introduced ($\mathrm{EA}_{f,0}$), per unit of second-generation biofuel energy produced ($\mathrm{EA}_{b,0}$) and per unit of substituted annual product ($\mathrm{EA}_{d,0}$) for the reforestation scenarios of 31% pasture (Forest_0), 112 EJ second-generation biofuel production ($\mathrm{Biofuel}_0$) and change in diet (-301 kcal/cap/day global average of animal products (Diet_0). These 3 scenarios make it possible to achieve 4.3 GtCO_2 of attenuated emissions in 2100 with reforestation, second-generation biofuel production and dietary change respectively.

Then, for mitigation scenarios mixing the 3 policies (involving a dietary change of Diet_i, second-generation biofuel production of $\mathrm{Biofuel}_i$ and reforestation of Forest_i), we apply these mitigation factors to each of the policies to calculate theoretical mitigated emissions without interaction between the policies:

$$
\begin{aligned}
\mathrm{E}_{\mathrm{T}ot,theoretical} =& \mathrm{EA}_{f,0} \times \mathrm{F}orest_i && (2.1)\\
&+ \mathrm{EA}_{b,0} \times \mathrm{B}iofuel_i && (2.2)\\
&+ \mathrm{EA}_{d,0} \times \mathrm{D}iet_i && (2.3)\\
& && (2.4)
\end{aligned}
$$

Because of the interactions between these mitigation policies, $\mathrm{E}_{\mathrm{T}otal,theoretical}$ is different to the emissions mitigated by the policy mix scenario calculated by the NLU $\mathrm{E}_{\mathrm{T}otal,\mathrm{NLU}}$. In the policy mix scenario, mitigated emissions result from mitigation efforts related to reforestation ($\mathrm{E}ffort_{\mathrm{F}orest}$), second-generation biofuel production ($\mathrm{E}ffort_{\mathrm{B}iofuel}$) and diet change ($\mathrm{E}ffort_{\mathrm{D}iet}$) as follows:

$$\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}} = \frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}}(\mathrm{EA}_{f,0} \times \mathrm{F}orest_i \tag{2.5}$$

$$+\,\mathrm{EA}_{b,0} \times \mathrm{B}iofuel_i \tag{2.6}$$

$$+\,\mathrm{EA}_{d,0} \times \mathrm{D}iet_i) \tag{2.7}$$

We deduce the efforts related to reforestation ($\mathrm{E}ffort_{\mathrm{F}orest}$), second-generation biofuel production ($\mathrm{E}ffort_{\mathrm{B}iofuel}$) and dietary change ($\mathrm{E}ffort_{\mathrm{D}iet}$):

$$\mathrm{E}ffort_{\mathrm{F}orest} = \frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}}\mathrm{EA}_{f,0} \times \mathrm{F}orest_i \times \frac{1}{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}} \tag{2.8}$$

$$= \frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}} \frac{\frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{F}orest_0} \times \mathrm{F}orest_i}{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}} \tag{2.9}$$

$$= \frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}} \frac{\mathrm{F}orest_i}{\mathrm{F}orest_0} \tag{2.10}$$

$$\mathrm{E}ffort_{\mathrm{B}iofuel} = \frac{\frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}}\mathrm{EA}_{b,0} \times \mathrm{B}iofuel_i}{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}} \tag{2.11}$$

$$= \frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}} \frac{\mathrm{B}iofuel_i}{\mathrm{B}iofuel_0} \tag{2.12}$$

$$\mathrm{E}ffort_{\mathrm{D}iet} = \frac{\frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}}\mathrm{EA}_{d,0} \times \mathrm{D}iet_i}{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}} \tag{2.13}$$

$$= \frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}} \frac{\mathrm{D}iet_i}{\mathrm{D}iet_0} \tag{2.14}$$

Through this formalization, we hypothesize that mitigation policies mitigate emissions linearly according to the mitigation factors $\mathrm{EA}_{f,0}$,$\mathrm{EA}_{d,0}$ and $\mathrm{EA}_{b,0}$ for reforestation, dietary change and second-generation biofuel production respectively. This assumption is corrected by the ratio $\frac{\mathrm{E}_{\mathrm{T}ot,\mathrm{NLU}}}{\mathrm{E}_{\mathrm{T}ot,theoretical}}$ which changes in the different scenarios to obtain 4.3 $GtCO_2$ of mitigated emissions in 2100. Finally only scenarios that mitigate 4.3 $GtCO_2$ (±5%) are retained.

Complete factorial experiment

The scenario sampling plan defines which scenarios will be simulated. The type of sampling is important to avoid biased sampling. For this reason we chose to sample using a complete factorial plan that avoids scenario sampling bias.

A complete factorial plan consists of sampling scenarios defined by several variables (here reforestation effort, dietary change effort and second-generation biofuel production effort) on a regular basis throughout the set of values taken by these variables. In this plan, the efforts of each of three mitigation policies therefore take values between 0% and 100% in 10% steps with the constraint that the sum of hte efforts must be equal to 100%.

In the following sections, we define how scenarios are built when 100% of the mitigation effort is provided by a single mitigation policy (reforestation, dietary change or second-generation biofuel production).

Dietary change scenario

The mitigation scenario composed exclusively of a dietary change (called here DC) is inspired by the Agrimonde scenario called AG1 (Paillard et al. 2011) which aims to describe a sustainable diet. We modified the plant, ruminant and monogastric demand of AG1 to reach the 4.3Gt$CO_{2,eq}$ mitigated emissions target by substituting plant food calories (low emission intensive product) for ruminant calories (intensive emissions product). This substitution occurs in the same proportion in all regions unless a lower limit of ruminant consumption of 65 kcal/cap/day is reached (as in Africa in the DC scenario). In that case, ruminant calorie substitution continues in other regions (excluding Africa) until 4.3 Gt$CO_{2,eq}$ of mitigated emissions are achieved. DC scenario regional diets are presented in Table. 2.1.

Table 2.1 – Regional diet in 2050 (kcal/cap/day)

	Baseline				DC[1]			
Regions	Plant Food	Ruminant*	Monogastric	Aquatic[2]	Plant Food	Ruminant[3]	Monogastric[3]	Aquatic[2,3]
Africa	2586	111	27	-	2564	65	350	21
Brazil	2466	382	331	-	2568	183	253	42
Canada	2543	516	389	-	2568	183	200	49
China	2682	161	334	-	2568	91	253	88
Europe	2543	516	389	-	2568	183	200	49
FSU	2543	516	389	-	2568	183	212	37
India	2517	230	64	-	2568	91	253	88
Middle-East	2837	274	74	-	2568	154	207	40
OECD Pacific	2543	516	389	-	2568	183	200	49
Rest of Asia	2682	161	334	-	2568	91	253	88
Rest of LAM	2466	382	331	-	2568	183	207	42
USA	2543	516	389	-	2568	183	200	49

[1] DC is a diet based on AG1 and modified to achieve 4.3GtCO_2/year in 2100
[2] Aquatic products are not computed by NLU
[3] Sum of aquatic, ruminant and mongastric products is 432kcal/cap/day in all regions

In mitigation scenarios composed of a change in diet mixed with reforestation and production of second generation biofuel, we take intermediate diets between the DC and the FAO diet used in the baseline (Alexandratos & Bruinsma 2012). In these intermediate diets, the consumption of monogastric and aquatic products is set to those of the DC diet and the consumption of ruminant and plant products are linear interpolations between the respective consumptions of DC and FAO.

The diets in the scenarios change between 2020 and 2050. Between 2001 and 2020, actual trends are used (Alexandratos & Bruinsma 2012) and between 2050 and 2100, the diets are kept constant.

Second-generation biofuel scenario

Ligno-cellulosic biofuels are produced in NLU from dedicated energy crops (woody or grassy crops). Dedicated energy crops correspond to short rotation coppice such as eucalyptus, willow or poplar and grasses such as miscanthus or switchgrass. The increase in second-generation biofuel production is linear between 2005 and 2100.

A global yield of 230GJ/ha in 2020, rising to 340GJ/ha (or 72Mkcal/ha) in 2050, is assumed for dedicated energy crops based on our literature review cross-checked with experts' views. This value is then distributed regionally based on the land distribution of potential yield used in NLU (see Souty et al. (2012)).

Energy crops are allocated homogeneously over the different categories of land quality. They expand over agricultural areas without affecting forested land. In so doing they increase the scarcity of agricultural land and spur intensification of crop and livestock production. In the scenario with only biofuel production to mitigate 4.3 $GtCO_2$, 112 EJ are produced worlwide.

Emissions from biofuel fertilization and from conversion of pasture to cropland are computed based respectively on emissions from crop fertilization as described in Tier 1 of IPCC (2006*a*) and emissions from land-use change as described in Le Quéré et al. (2009). With a global yield of 230 GJ/ha, a NUE of 0.5, a fertilization rate of 93 kgN/ha and an emission factor of 0.03 $kgCO_{2,eq}$/kgN, we deduce an emission factor of 6 g CO_2/MJ due to biofuel fertilization. Emissions saved due to the use of biofuel instead of fossil fuel are also computed. First, we convert primary energy included in grassy and woody crops into energy included in biofuel after refining with a coefficient of 0.481 MJ/MJ. We made the assumption that biofuel is used in the transport sector instead of a mix of diesel (50%) and ethanol (50%) with an emission factor of 87.85 gCO_2/MJ (Hoogwijk et al. 2009). Finally we removed emissions produced during refining (0.6 gCO2/MJ) and transport to the refinery (0.6 gCO_2/MJ) (Hoogwijk et al. 2009). The final emission coefficient is 41 gCO2/MJ of saved emissions per MJ of biofuel minus 6 gCO_2/MJ due to biofuel fertilization. By computing the difference betwenn fossil fuel emissions of 86 gCO_2/MJ (Hoogwijk et al. 2009) and emissions from second-generation biofuel production between 26 and 65 gCO_2/MJ Jungbluth et al. (2008), our estimation of saved emissions due to the production of second-generation biofuel instead of fossil fuel (35 gCO2/MJ) is in the middle of the range 21-60 gCO_2/MJ. By taking into account uncertainty around this coefficient, more pessimistic assumptions about the mitigation potential of second-generation biofuel would lead to worse impacts on biodiversity and food prices in NLU due to the requirement to produce a higher amount of biofuel in order to mitigate 4.3 $GtCO_{2,eq}$ and vice-versa. Use of carbon capture and storage, or use of co-production in bioelectricity production could improve the mitigation potential of second-generation biofuel (Whitaker et al. 2010) and reduce its negative impacts on biodiversity and food prices.

Forest scenario

The forest scenario used as a baseline is the continuation of current trends until 2050 and a stabilization of forest areas after 2050. The alternative scenario is inspired by the reforestation scenario in the Natural Climate Solution presented in (Griscom et al. 2017). In this scenario, forest lands expand at the expense of pastures to reach the climate target. The distribution of the reforested area between regions is therefore proportional to the area of pasture present in each region.

[!h]

Table 2.2 – Regional reforestation rate in 2020 and 2050. A negative reforestation rate indicates deforestation.

	Baseline		Reforestation	
Regions	Reforestation rate (%)	Forest change (Mha)	Reforestation rate (%)	Forest change (Mha)
Africa	-0.032	-11.042	0.213	71.409
Brazil	-0.029	-13.471	0.021	9.393
Canada	-0.001	-0.526	0.002	1.318
China	0.086	13.824	0.290	44.767
Europe	0.029	3.642	0.101	12.244
FSU	0.005	4.215	0.052	43.241
India	0.044	1.284	0.085	2.388
Middle-East[1]	0.000	0.000	0.000	0.000
OECD Pacific	-0.022	-1.353	0.464	27.650
Rest of Asia	-0.016	-5.616	0.041	14.094
Rest of LAM	-0.027	-9.586	0.082	28.343
USA	0.011	2.973	0.106	27.167

[1] In (Hurtt et al. 2011), there is no forest in the Middle-East in the reference year

The area of forest follows historical trends between 2001 and 2020. The increase in forest area at the expense of pasture occurs between 2020 and 2100.

2.6.3 Scenario sampling plan

In this section, we present a set of policies allowing us to reach 2° by making the necessary mitigations in the AFOLU sector. The experimental design follows a complete factorial design to address a wide range of adequate mitigation scenarios.

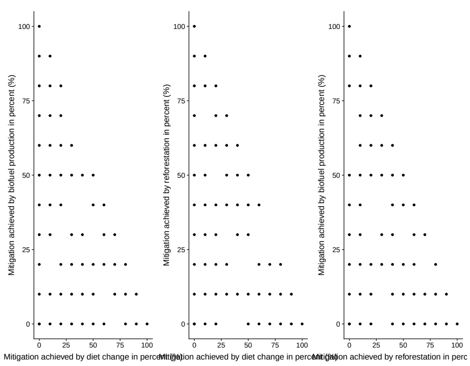


Figure 2.4 – Complete factorial design to address a wide range of mitigation scenarios

The holes in the complete factorial design correspond to scenarios that do not mitigate 4.3 $GtCO_2$ with a 5% error.

2.6.4 Results

Biodiversity indicators and food indicators relations at global scale

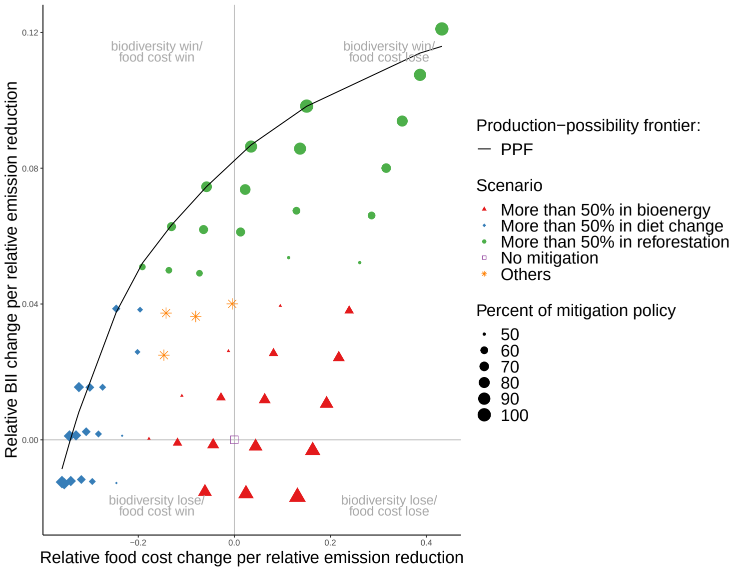


Figure 2.5 – Impacts of mitigation scenarios reaching 4.3 $GtCO_{2_{eq}}$ in 2100 of mitigated emissions based on combinations of second-generation biofuel production, dietary change and reforestation on global BII average and global food production cost. BII and food production cost are presented as a relative difference to the scenario without any mitigation policy (baseline). The mitigation effort of each policy (second-generation biofuel production, dietary change and reforestation) is expressed in the legend as a percentage of the overall mitigation effort.

Relationships between biodiversity indicators and food indicators at the regional scale

Mitigation scenarios

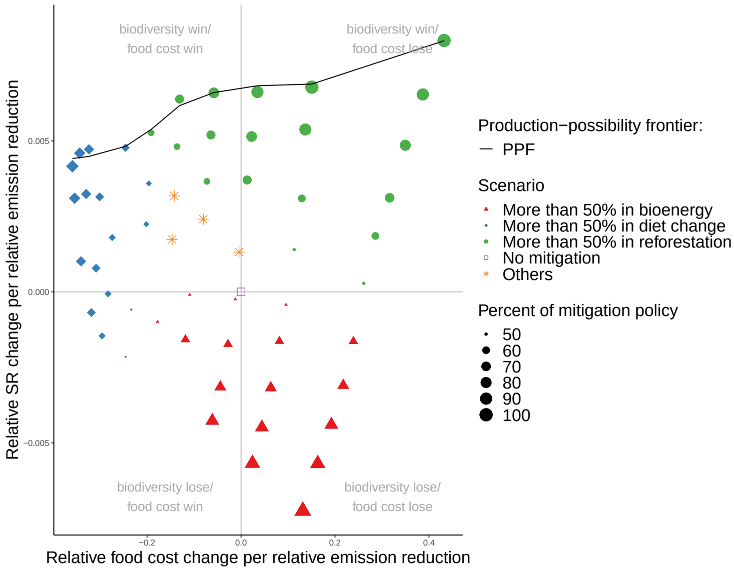


Figure 2.6 – Impacts of mitigation scenarios reaching 4.3 $GtCO_{2_{eq}}$ in 2100 of mitigated emissions based on combinations of second-generation biofuel production, dietary change and reforestation on global SR average and global food production cost. SR and food production cost are presented as a relative difference to the scenario without any mitigation policy (baseline). The mitigation effort of each policy (second-generation biofuel production, dietary change and reforestation) is expressed in the legend as a percentage of the overall mitigation effort.

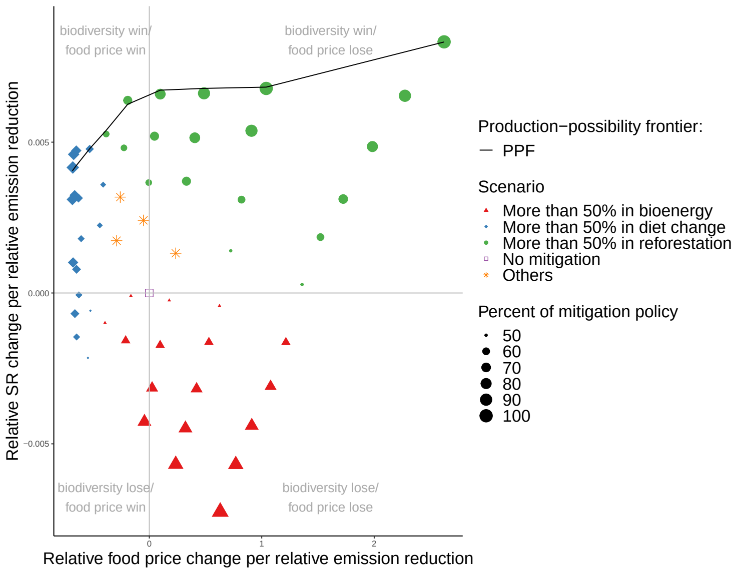


Figure 2.7 – Impacts of mitigation scenarios reaching 4.3 $GtCO_{2_{eq}}$ in 2100 of mitigated emissions based on combinations of second-generation biofuel production, dietary change and reforestation on global SR average and global food price. SR and global food price are presented as a relative difference to the scenario without any mitigation policy (baseline). The mitigation effort of each policy (second-generation biofuel production, dietary change and reforestation) is expressed in the legend as a percentage of the overall mitigation effort.

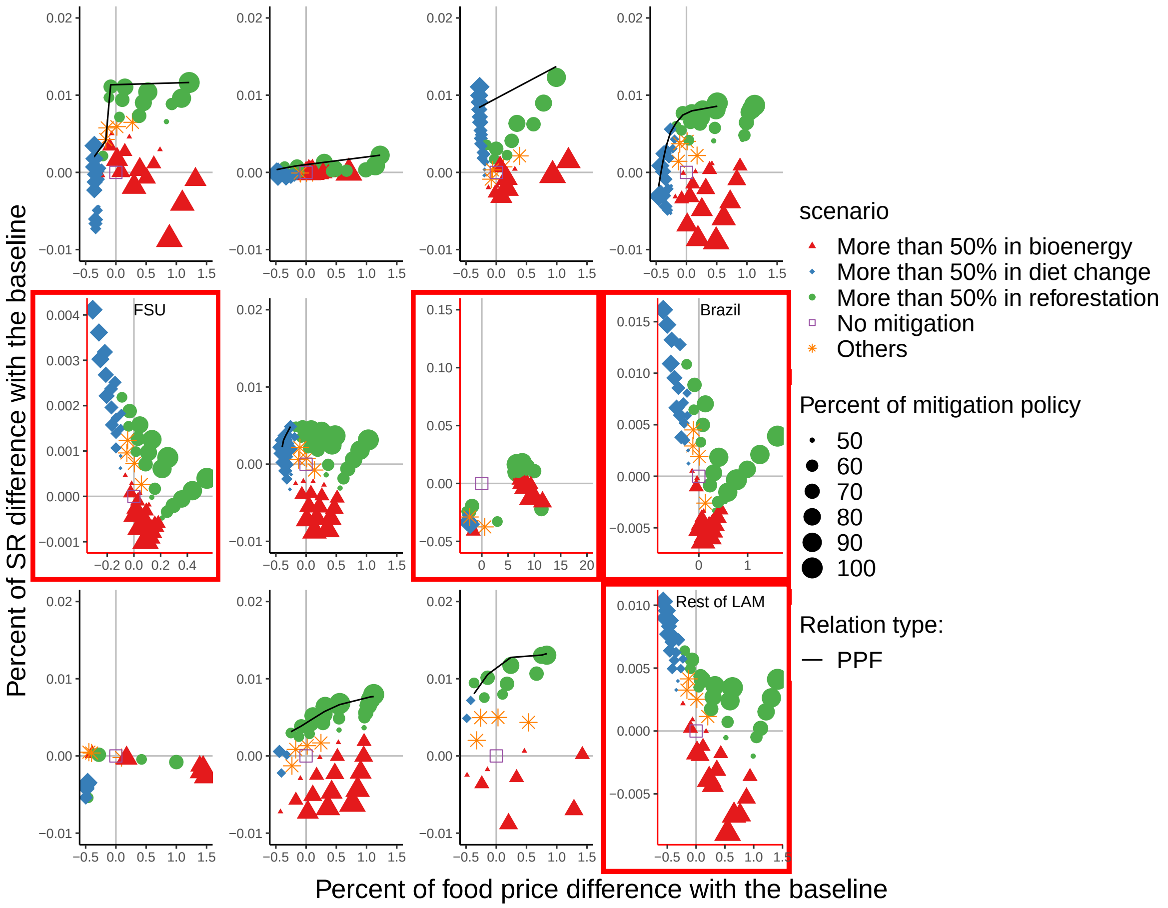


Figure 2.8 – Relative change in SR and food price with respect to relative change in GHG emission reduction at the regional level. This ratio takes into account the different emission changes within a region from one mitigation scenario to another and the unequal distribution of mitigation efforts between regions within a scenario. A relative change in the SR of 0.2 therefore indicates that a 10% reduction in regional emissions means a 2% reduction in biodiversity. To compare the regions with each other, a common range is chosen for the axes of each region. An unzoom is provided for FSU, India, Brazil and Rest of LAM with extreme BII and food indicator change for mitigation scenarios (red rectangles).

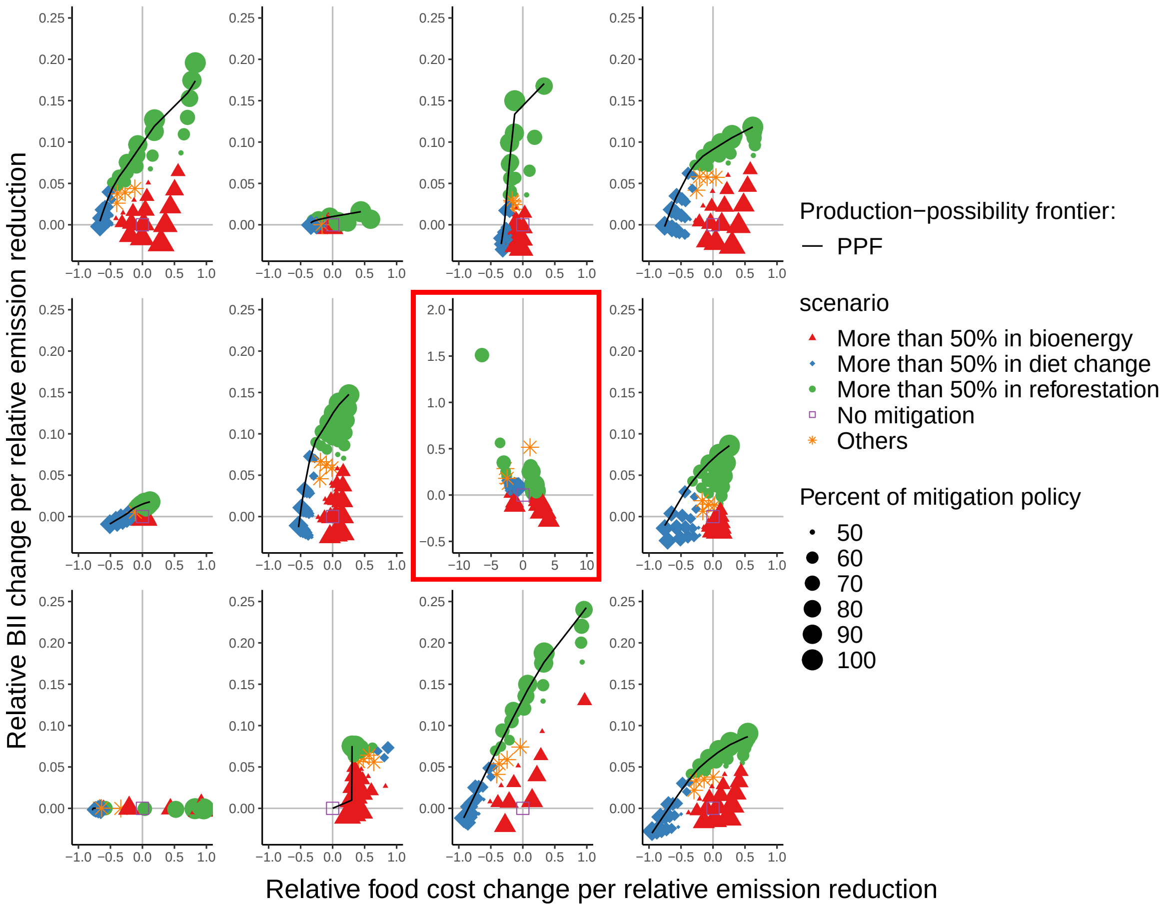


Figure 2.9 – Relative change in BII and food cost with respect to relative change in GHG emission reduction at the regional level. This ratio allows takes into account the different emission changes within a region from one mitigation scenario to another and the unequal distribution of mitigation efforts between regions within a scenario. A relative change in the BII of 0.2 therefore indicates that a 10% reduction in regional emissions means a 2% reduction in biodiversity. To compare the regions with each other, a common range is chosen for the axes of each region. An unzoom is provided for India with extreme BII and food indicator change for mitigation scenarios (red rectangle).

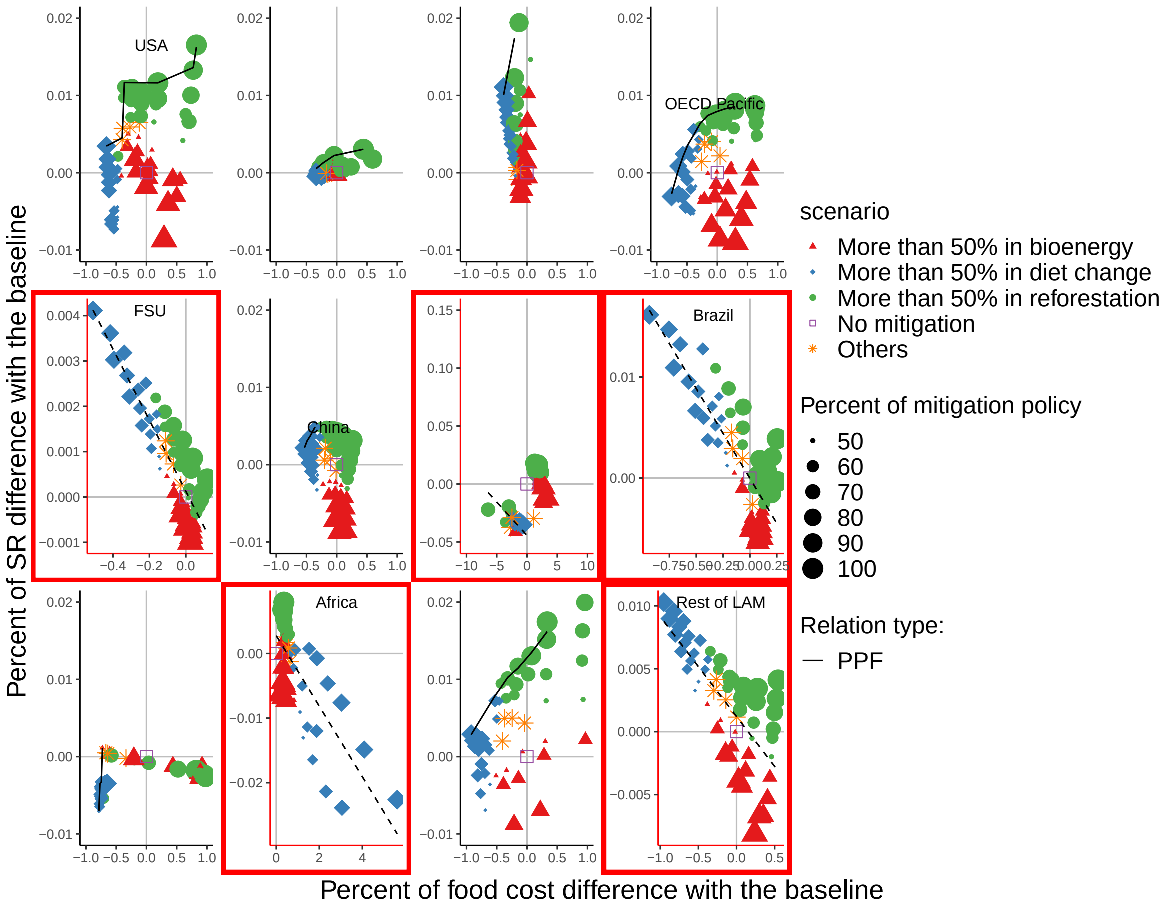


Figure 2.10 – Relative change in SR and food production cost with respect to relative change in GHG emission reduction at the regional level. This ratio takes into account the different emission changes within a region from one mitigation scenario to another and the unequal distribution of mitigation efforts between regions within a scenario. A relative change in the SR of 0.2 therefore indicates that a 10% reduction in regional emissions means a 2% reduction in biodiversity. To compare the regions with each other, a common range is chosen for the axes of each region. An unzoom is provided for FSU, India, Brazil and Rest of LAM with extreme BII and food indicator change for mitigation scenarios (red rectangles).

Table 2.3 – Global food security and biodiversity indicators for the sampled mitigation scenarios in 2100. Indicators are rescaled between 0 and 100 for each indicator. 0 means a high price, a high cost, low BII and a low SR. On the contrary, 100 means a low price, a low production cost, a high SR and high BII. Scenarios are described by the mitigation effort (in %) of second-generation biofuel production, reforestation of pasture and dietary change.

Scenario			Food Cost ($/Mkcal)	Food Price ($/Mkcal)	BII	SR
Biofuel*	Forest*	Diet*				
40	40	20	4197	85.2	0.831	2.54
30	40	30	4098	78.1	0.830	2.54
60	10	30	4049	74.2	0.820	2.53
0	100	0	4766	143.9	0.851	2.54
80	20	0	4453	101.8	0.823	2.53
50	40	10	4328	94.7	0.830	2.53
90	10	0	4415	98.3	0.820	2.53
30	0	70	3788	63.1	0.817	2.53
100	0	0	4374	94.9	0.816	2.53
10	90	0	4707	135.3	0.848	2.54
20	10	70	3801	63.5	0.821	2.53
30	30	40	4012	72.2	0.827	2.54
20	0	80	3759	62.7	0.817	2.54
0	50	50	3953	70.0	0.833	2.54
80	10	10	4261	87.3	0.820	2.53
10	10	80	3773	63.1	0.821	2.54
50	20	30	4061	75.4	0.824	2.53
50	30	20	4187	83.8	0.827	2.53
20	80	0	4658	128.2	0.844	2.54
20	50	30	4108	79.3	0.833	2.54
0	80	20	4248	91.4	0.842	2.54
90	0	10	4234	85.2	0.816	2.53
10	60	30	4119	80.5	0.836	2.54
10	70	20	4232	89.3	0.839	2.54
60	20	20	4167	81.8	0.824	2.53
30	10	60	3834	64.0	0.821	2.53
70	10	20	4145	80.0	0.820	2.53
80	0	20	4123	78.3	0.817	2.53
0	60	40	4032	74.7	0.836	2.54
0	70	30	4127	81.8	0.839	2.54
10	20	70	3811	63.9	0.824	2.54
40	50	10	4350	97.2	0.834	2.54
0	10	90	3756	62.8	0.821	2.54
10	0	90	3741	62.6	0.817	2.54
0	40	60	3882	66.3	0.830	2.54
20	60	20	4220	87.5	0.836	2.54
20	20	60	3845	64.5	0.824	2.54
50	50	0	4543	112.8	0.834	2.53
0	90	10	4399	105.0	0.845	2.54
20	30	50	3940	68.6	0.827	2.54
60	40	0	4514	109.3	0.830	2.53
30	70	0	4615	121.8	0.841	2.54
70	30	0	4486	105.9	0.827	2.53
10	50	40	4025	73.8	0.833	2.54
30	60	10	4371	99.6	0.838	2.54
70	20	10	4285	89.7	0.823	2.53
40	0	60	3818	63.5	0.817	2.53
60	30	10	4309	92.4	0.827	2.53
10	40	50	3947	69.3	0.830	2.54
50	10	40	3971	69.7	0.821	2.53
0	20	80	3781	63.4	0.824	2.54
0	0	100	3735	62.6	0.817	2.54
40	10	50	3898	66.5	0.821	2.53
20	40	40	4018	73.0	0.830	2.54
50	0	50	3883	66.0	0.817	2.53
10	80	10	4381	101.7	0.842	2.54
40	60	0	4575	116.8	0.837	2.54
0	0	0	4203	79.4	0.820	2.53

Chapter 3

Robust strategies for the AFOLU sector to stay inside planetary boundaries

This article is still in progress.

Keywords: Planetary boundaries | land-use | Biodiversity | AFOLU | Robust decision Making | Scenario discovery

Contents

3.1 Highlights

- Des sratégies ambitieuses sont à mettre en place dans le secteur AFOLU si la société veut rester au sein des planetary boundaries.
- Dietary change, reforestation and improvement of nitrogen use efficiency are robust strategies to stay within planetary boundaries
- The mix of robust strategies to implement in the AFOLU sector depends on the thresholds of the PBs: a higher threeshold for the PB related to the nitrogen cycle needs higher nitrogen use efficiency and a higher threeshold for the PB related to forest area and BII needs higher reforestation rate.

3.2 Introduction

Pressure on global land use is doomed to increase in the coming decades. With the increase in global income, diets may shift to more animal-rich diets (Popp et al. 2010). Combined with an increase in population, this increase in food demand, particularly for animal products, may increase pressure on the environment (Popp et al. 2010, Stehfest et al. 2009, Marques et al. 2019). Population and economic growth increase also the risk of extinction through agriculture, forestry or infrastructure development that degrades and fragments natural habitats (Tilman et al. 2017). Moreover, the AFOLU sector is likely to be highly solicited for compliance with the Paris Climate Agreements (Richards et al. 2015). Currently responsible for a quarter of emissions (Tubiello et al. 2015), the agriculture, forestry and other land use (AFOLU) sector has a significant potential to reduce emissions through technological options or structural change for relatively low mitigation costs (Frank et al. 2018). These additional pressures increase the uncertainty surrounding the sustainability of possible future land-use systems (Popp et al. 2017).

Literature on the assessment of land-use based mitigation policies usually divide the uncertainty surrounding the drivers of the land-use system into two categories: (i) the socio-economic context and (ii) the adoption of environmental policies. The uncertainty related to the socio-economic context was addressed by the construction of 5 model-based scenarios, collectively known as shared-socio economic pathways (SSP) (O'Neill et al. 2017), describing socio-economic conditions with specific assumptions for the AFOLU sector (Popp et al. 2017). In these predefined socio-economic contexts, different combinations of environmental policy in the AFOLU sector are studied (Obersteiner et al. 2016, Humpenöder et al. 2018, Heck, Gerten, Lucht & Popp 2018, Bertram et al. 2018). The setup of the SSP is particularly suitable to study climate mitigation policies, and intercompare models on that subject, by having all the other regulations considered as context. In that framework, an emphasis is put on assumptions on climate change with the Representative Concentration Pathways (RCP) that complete the SSP and can be studied independently with climate models (van Vuuren et al. 2011).

These studies assume that socio-economic conditions are contextual components in which environmental policies are applied. Within the SSP framework, the boundary between policy and context is somewhat flexible, for instance some assumptions on land-based mitigation policies can be part of context even when different RCP are tested (Popp et al. 2017). More fundamentally, the split between policies and context is arbitrary. Land use regulations, trade liberalization, technological development in agriculture, dietary choices or demographics could all be considered as policy levers rather than contextual assumptions.

Using narratives and grouping assumptions to build consistent scenarios allows to have a common reference for subsequent studies and to be able to refer to a limited set of well defined SSP scenarios. However fixed assumptions associations also reduce the combinations that can be considered and evaluated. To take a random example, there is no definitive evidence that strong regulations to avoid environmental tradeoffs, in SSP1, cannot be associated with animal products based resource intensive food consumption from SSP3. The SSP framework therefore limits drastically the possibility to study and evaluate trade-offs between environmental policies.

Although initially framed for climate policies evaluation and greenhouse gases emissions quantification, diverse dimensions and objectives can be studied within the SSP/RCP framework. Two types of studies are readily achieved in this framework, studying the effect of different climate policies on other indicators and determining costs and risks to reach diverse climate targets. However, the use of fixed assumptions on some drivers considered as context limits the usefulness of this framework to target issues such as biodiversity protection or nutritional adequacy of diets or explicitely quantify the risk, when a strategy is selected, to fail to reach a target due to an unexpected context.

To fill this gap, we propose here to apply the methodological framework of the robust decision making (RDM) used in situations of deep uncertainty (Bryant & Lempert 2010). Deep uncertainty refer to situations "where parties to the decision do not know or do not agree on the system model relating actions to consequences or the prior probability distributions for input parameters to these system models" (Lempert & Collins 2007). Here, we do not focus on a limited number of a priori specified alternatives as in previous studies which made a distinction between environmental policies and the SSPs. We define a land-use based strategy, in a broad sense, as the future evolution of a compound (diet, trade, population, forest area...) of the AFOLU sector. These land-use based strategies are considered as possible levers for action, though with uncertain amplitude. The RDM also allows to pursue a diversity of objectives with equal treatement. Here we use the planetary boundaries (PBs), a set of threshold on diverse environmental indicators (Steffen et al. 2015). The purpose of this study is therefore to define land-use based strategies that allow to remain robustly within PBs.

First, we sample a set of scenarios covering a wide diversity of possible futures for the AFOLU sector. We then assess the environmental impacts of these scenarios using NLU coupled with the PREDICTS model (Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle in prep.). Then, using the "Scenario Discovery" method (Bryant & Lempert 2010), we select robust land-use based strategies that remain within the PBs. Finally, we look at the robustness of these scenarios to different thresholds for the PBs and to different sustainability indicators.

3.3 Method

3.3.1 Overview of the modelling process

The purpose of this study is to select the levers to stay within PBs. The levers are defined following 5 steps: (i) selection of components of the AFOLU sector to describe possible futures for the AFOLU sector, (ii) definition of values taken by these components of the AFOLU sector, (iii) the simulation of scenarios with NLU/PREDICTs models, (iv) a "Scenario discovery" analysis to select land-use based strategies which stay inside the gloabl PBs and (v) a sensitivity analysis of this scenario selection to different PBs and different sustainability indicators.

3.3.2 Possible futures for the AFOLU sector

To describe some possible futures for the AFOLU sector, we describe different variants in the different components of the AFOLU sector. To ensure that we cover a sufficient diversity of possible

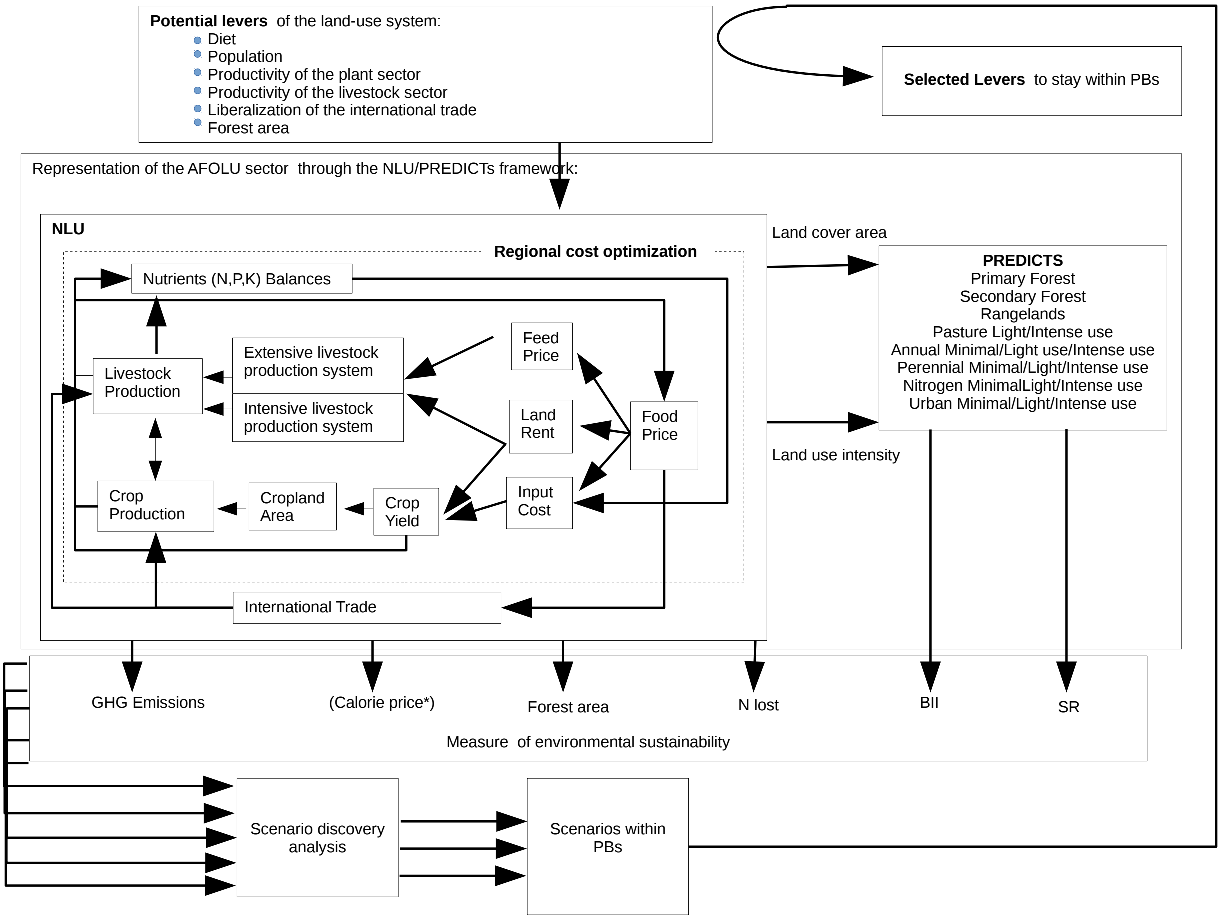


Figure 3.1 – Overview of the modelling steps.

futures, we use the socio-economic components of the AFOLU sector defined in Popp et al. (2017), namely population, food plant demand, animal demand, forest protection, plant sector productivity, animal sector productivity and trade openness (Table. 3.1). Although they are important factors in the AFOLU sector, we choose to leave urban sprawl, increased irrigated areas and waste reduction outside the scope of this study because their modelling (in the context of this study) is slightly redundant with other components of the AFOLU sector (forest area, NUE or diet change respectively) and because of the too large number of scenarios generated. We also leave aside bioenergy production scenarios that are not compatible with the forest, biodiversity and nitrogen cycle PBs (Heck, Gerten, Lucht & Popp 2018) and whose associated technologies to efficiently convert biomass to hydrogen and sequester the carbon produced are highly uncertain.

3.3.3 Sampling of scenarios

Because of the large number of possible futures for the AFOLU sector, we have to sample a limited number of scenarios. We decide to carry out a complete factorial design in order to have a wide coverage of the possible scenarios by taking 3 values for each variable in its possible range of values. For each variable, the three variants are called "Low", "Medium" and "High" (Table. 3.1).

Table 3.1 – Global range of selected components of the AFOLU sector in 2030.

	Low	Medium	High
Population (Mia of head)	8.10	8.37	8.61
Human plant demand (Mkcal/cap/year)	920	917	914
Human ruminant demand (Mkcal/cap/year)	107	84	60
Forest area (Gha)	3568	3653	3730
NUE	0.34	0.38	0.38
Livestock conversion factor (kcal/kcal)	13	10	-
Total plant imports (Tkcal)	2961	3305	3645

In NLU, population trends are described for each of the 12 major regions of the world (See Fig. 4 in Supplementary data). It corresponds to the population evolution described by Kc & Lutz (2017) and transposed to NLU regions. It varies globally between 8.6 and 10.1 billion inhabitants (Table. 3.1). The regional distribution is presented in Table. 3.6 of supplementary information.

Plant and animal food demand are described by three different scenarios: (i) a scenario based on Alexandratos & Bruinsma (2012) which will be the most intensive scenario in animal calorie, (ii) a scenario that will achieve an emission reduction of 4.3 $GtCO_2$ in 2100 described in Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle (in prep.) which will be the least intensive scenario in animal products and (iii) an intermediate scenario that sets a food consumption equal to the mean of the two previous scenarios (See table. 3.7). The 3 diet scenarios explored correspond to global ruminant consumption of 107, 84 and 60 Mkcal/cap/year in 2030 (Table. 3.1).

The forest area in 2030 is set at 3568 Mha, 3653 Mha and 3730 Mha worldwide (Table. 3.1). With a forest area in 2001 of 3541 Mha, this set of scenarios represents 3 reforestation scenarios (+27 Mha, +112 Mha and + 189 Mha of forest in 2030 compared to 2001). To minimize the impact of this reforestation on food security, reforestation is distributed according to the distribution of pasture around the world (Table. 3.8).

In this study, we define plant sector productivity as the yield achieved per unit of input. In NLU, we represent it with the slope at the origin of the crop production function (concave function linking yield to the use of nitrogen, phosphate and potash fertilizers). The slope of this function in 2050 can be set constant at its 2001 value, or follow the scenario of increased efficiency in input use described by FAO (Bruinsma 2003). This slope can also be related to the Nitrogen Use Efficiency (NUE) (See supplementary information for a presentation of the equations) whose regional values are presented in Table. 3.9. The two NUE scenarios explored correspond to global NUE of 0.34 and 0.38 (Table. 3.1).

In NLU, livestock farming is composed of two production systems: the pastoral system where livestock are only fed with grass from pastures with low yields and the mixed crop-pasture system where livestock are raised on intensive pastures located in mosaic with crop production. In this case, livestock feed is a mixture of grass, grain and fodder (Souty et al. 2012). To represent an intensification of livestock farming, we change the efficiency of systems to convert ingested vegetal calories into animal calories either by setting it at its 2001 value (10.4 $\text{kcal}_{veg}/\text{kcal}_{anim}$), or by increasing it according to the scenario proposed by Bouwman et al. (2005) until it reaches 12.9 $\text{kcal}_{veg}/\text{kcal}_{anim}$ (ref). The change in production from one system to another is calculated by NLU in an endogenous way according to the land value and the price of the feed (Souty et al. 2012). The 2 productivity scenarios in the animal sector explored in this study have an global average of conversion rate of vegetal calories into animal calories of 13 and 10 $\text{kcal}_{\text{veg}}/\text{kcal}_{\text{anim}}$ (Table. 3.1).

In this study, we consider different variants of trade openness (i.e. share of internationally traded quantities on total production) of plant food products by modifying the elasticity of traded quantities to the growth in plant food production: (i) +5% between 2018 and 2050 (increased openness), (ii) -5% between 2018 and 2050 (reduced openness) and (iii) no change. These scenarios correspond respectively to 2961, 3305 and 3645 Tkcal of plant imports in 2030 (Table. 3.1).

3.3.4 The modelisation frame-work: coupling of NLU and the PREDICTS models

To evaluate position of the possible futures for the AFOLU sector relative to the 4 PBs, we use the coupling of NLU and PREDICT models presented in section. . We also use the nitrogen balances on cropland presented in section. and the computation of GHG emissions presented in section. . For pastures, we use nitrogen lost from a reduced nitrogen balance (See section. 3.8.8 in supplementary information).

A precise description of NLU model is provided in Souty et al. (2012), Brunelle et al. (2015) and Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle (in prep.). Equations of NLU are recalled in the section. 3.8.8 of supplementary information.

A detailed description of the PREDICTS project is provided in Purvis et al. (2018).

3.3.5 “Scenario discovery” cluster analysis

To select the scenarios which remain robustly within the PBs, we use the "Scenario discovery" cluster analysis. This method aims at characterizing the combinations of uncertain input parameter values or "drivers" most predictive to stay within the PBs (Bryant & Lempert 2010). In contrast to exploratory approaches or optimization approaches such as the possibility-production frontier, we explicitly take into account with this method the uncertainty around socio-economic conditions of the AFOLU sector and the adoption of other environmental policies into the selection of the robust strategy to stay within PBs. For example, when an environmental protection policy is taken, we do not know under which socio-economic conditions it will be implemented. For example, the environmental benefits of deploying a NUE improvement policy could be completely offset by an increase in the consumption of animal prodcuts.

To define AFOLU sector that stay robustly inside PBs, we select a set of scenarios (also called a "box") using the Patient Rule Induction Method (PRIM) cluster analysis (Bryant & Lempert 2010). The "box" is defined by the boundaries that limit the space of inputs which define the land-uses remaining in the PBs. For example a "box" is an AFOLU sector with a population between 8.37 and 8.61 Gheads, which eat in average 84 Mkcal/cap/year of animal product per year, with a high reforestation rate and with a high liberalization of trade. To measure the quality of the "box", two indicators measures the ability of the selected AFOLU sectors to stay within PBs: coverage and density (Bryant & Lempert 2010). Coverage measures how completely the scenarios defined by the "box" capture the AFOLU sectors that remain within the PBs (also called the Type I or wrong rejection). Density measures the purity of the "box". It is expressed as the share of scenarios in the "box" that allows to remain within the PBs (called the Type II or wrong acceptance).

The objective of the PRIM algorithm is to minimize type I and type 2 uncertainties to define the robust strategies in the AFOLU sector that stay within the PBs. To do so, the PRIM algorithm is an iterative process, which peels away thin faces of the input space to generate smaller regions each containing a higher mean coverage and density (Bryant & Lempert 2010). The density increases with the number of leverage as less and less land-use based strategies are out the PBs in the selected "box". Conversely, coverage decreases with a reduction in the size of the box because PRIM leaves out land-use based strategies which stay within PBs (Fig. 3.3).

The indicators used to define PBs are based on the latest study defining PBs (Steffen et al. 2015). The PBs "define a safe operating space for humanity based on the intrinsic biophysical processes that regulate the stability of the Earth System" (Steffen et al. 2015). With NLU/PREDICT models, we can address four PBs: the remaining forest area, nitrogen fluxes, climate change and biosphere integrity. A detail of PB computation is available in Section 3.8.7 in supplementary information. The PBs are summarized at a global scale in Table. 3.2 and at regional scales in Table 3.12.

To perform this "Scenario Discovery" cluster analysis, we use the python package "Exploratory Modeling Workbench" (Kwakkel 2017).

3.3.6 Sensitivity analysis

Sensitivity analysis to the uncertainty surrounding PBs

PBs are associated with broad uncertainties (Steffen et al. 2015). In the following, we detail and quantify the different sources of uncertainty for each PB.

A major source of uncertainty is the inclusion or not of a forest type classified as "other forest" in the potential vegetation types without anthropogenic influence (Ramankutty et al. 2010). This type of forest includes non-continuous forests and is at the boundary with another nearby vegetation type called woodland. We include 40% of the forest in this category in the calculation of the PB as realized in Steffen et al. (2015). For the sensitivity analysis, we include between 0% and 100% of this category. The use of this range varies the threshold on the quantity of forest to be maintained between 3.4 Gha and 5.1 Gha.

For nitrogen lost in surface run-off waters, the main source of uncertainty on the critical concentration of nitrogen in the environment is the nitrogen concentration threshold in surface water, which can vary between 1.0 and 2.5 (De Vries et al. 2013). The PB associated with the amount of nitrogen lost by agriculture is therefore estimated at between 49 and 79 $TgN.yr^{-1}$.

For the calculation of the boundary boundary associated with climate change, previous estimates of emissions mitigated by the AFOLU sector are between 0.92 and 1.37 $GtCO_{2,eq}$ of non-CO_2 emissions in 2030 (Wollenberg et al. 2018).

Finally, the PB associated with the integrity of the biosphere is estimated for Biodiversity Intactness Index (BII) between 0.3 and 0.9 (Steffen et al. 2015).

Due to this high uncertainty, we carry out a sensitivity analysis around the calculated PBs. Because of the interactions between PBs, we undertake a global sensitivity analysis following a Monte-Carlo experiment. This experiment consists in uniformly taking 1 million values in the uncertainty range of each PB. We therefore obtain a sample composed of 1 million combinations of PBs. For each combination of PBs, we use the PRIM algorithm to select the robust combination of strategy that allow to stay within the PB and have the highest density. Finally, we compute for each decile of the uncertainty range of each boundary, the probability to find dietary change, reforestation, crop farming productivity, livestock farming productivity or liberalisation of trade among the strategies selected by the PRIM algorithm to stay within PBs.

Sensitivity analysis to a supplementary indicator of sustainability to the PBs: adding the food price

At the heart of the AFOLU sector's problems, the "eradication of hunger in the world" (SDG 2) can be jeopardized with other sustainable development objectives such as forest protection (Stevanović et al. 2017), access to energy through the development of bioenergy (Humpenöder et al. 2018) or with non-CO_2 emissions mitigation (Hasegawa et al. 2018). To describe the relations between the SDG2 and the four previous PBs, we add the food price as an additional indicator of the sustainability of the AFOLU sector.

The purpose of this sensitivity analysis is to estimate the influence of the addition of different global food price objectives on the choice of a robust strategy to stay within PBs and maintain a certain level of food security. As an indicator of food security, we use a price index defined as the ratio between the food price divided by the food price in 2001.

Unlike other global limits, this objective does not have a threshold beyond which food insecurity increases sharply. We use not only one threshold for this indicator but a range of price limits between the maximum food price of the first decile and the minimum food price of the last decile of the food price range obtained through the different simulations.

Table 3.2 – Global PBs and the threeshod for the food price.

	This study		Steffen et al.2015 (Steffen et al. 2015)	
PBs	PB value	Range	PB value	Range
Nitrogen run-off (TgN)	62	50-79	62	62-82
Climate Change	1 ($GtCO_{2,eq}$)[1]	0.92-1.15[1]	396.5 ppm	350-450 ppm
Forest area (Gha)	3.7	3.4-5.1	4.8	3.4-4.8
BII	-	0.3-0.9	0.9	0.3-0.9
Food price index[2]	1.5	1.5-1.9	-	-

[1] non-CO_2 mitigated emissions in the AFOLU sector
[2] Food price in 2030 divided by food price in 2001

3.4 Results

3.4.1 Relationship between PBs

First, we study the relationships between the outputs that determine the environmental sustainability of the AFOLU sector to establish synergies and trade-offs between these different criteria (Fig. 3.2). Synergies between environmental indicators show that it is possible to achieve several environmental objectives at the same time. Biodiversity conservation (BII) and forest maintenance, or non-CO_2 emissions and nitrogen lost in surface run-off waters, present synergies. Part of the synergy between BII and forest comes from the positive impact of forest on BII in PREDICTS models. Part of the synergy between non-CO_2 emissions and reactive nitrogen lost in the medium is due to the presence in the nitrogen lost in the medium of reactive nitrogen that volatilizes into the air after leaching. This last synergy is not fully robust to the scenarios because indirect N_2O emissions from the plant sector represent only 0.1% of N_2O emissions from the plant sector (Prudhomme, Brunelle, Dumas & Le Moing in prep.).

The forest has no clear relationship with non-CO_2 agricultural emissions and lost nitrogen.

A trade-off between (i) biodiversity conservation (BII) and (ii) reducing non-CO_2 emissions or reducing nitrogen losses to the environment seems to be emerging. This relationship, which may seem counter-intuitive, is in fact evidence of two elements: (i) the absence of direct climate impact on biodiversity in this framework and (ii) the low impact of agricultural intensification on BII compared to the impact of forest reduction (see PREDICTS model equations in supplementary information). This last effect confirms the beneficial impact of land-sparing in freeing up areas that are suitable for biodiversity whatever the future land-use based strategy chosen (Phalan 2018).

In line with literature, the relationships between biosphere integrity, nitrogen lost and remaining forest area are quite strong (Mace et al. 2014).

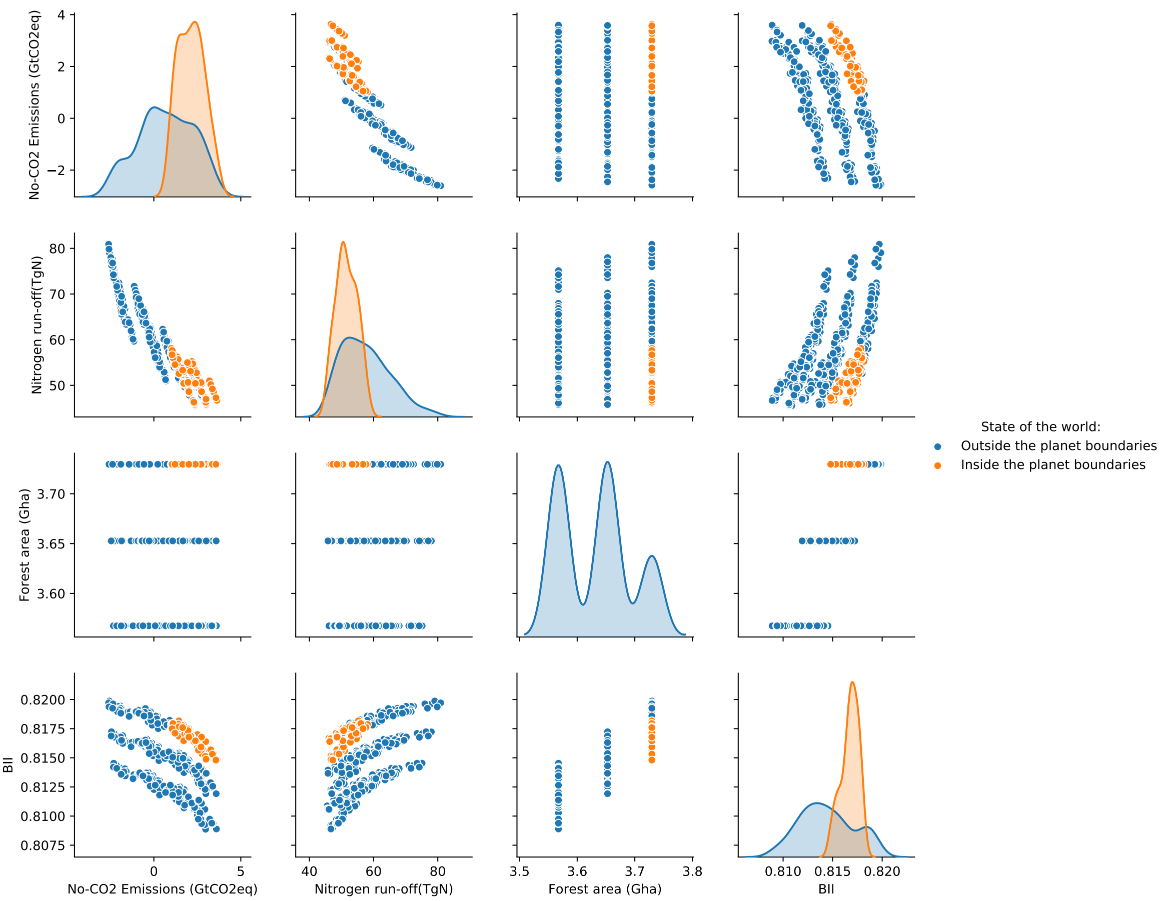
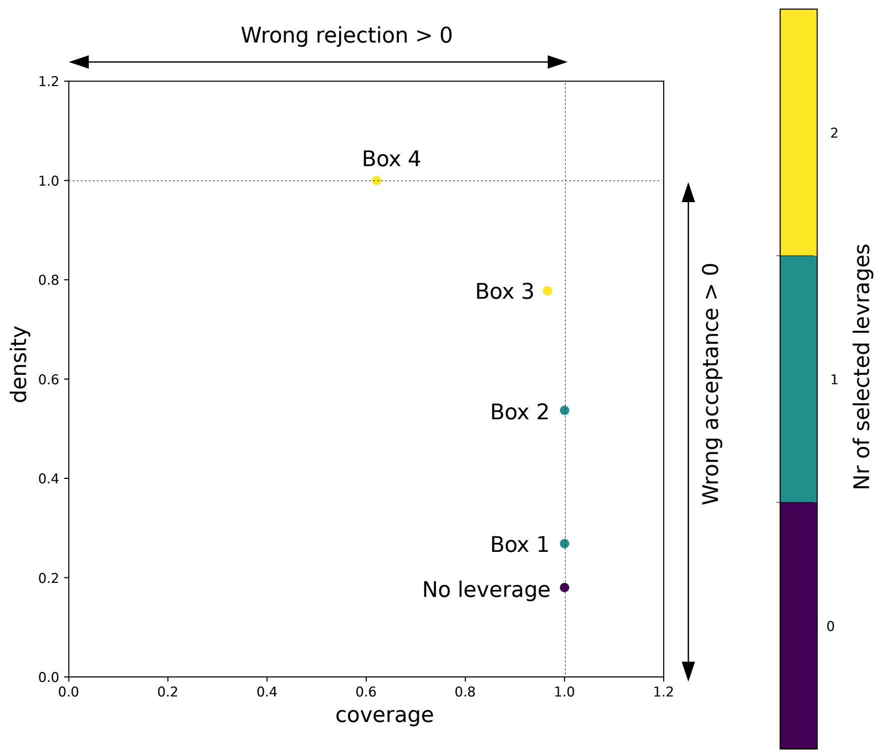


Figure 3.2 – Relations between environmental indicators resulting from scenarios explored in 2030. Graphs on the diagonal of the figureth are density plots of the different PBs.

3.4.2 Robust strategies in the AFOLU sector to stay within PBs

First, we represent the trade-off between type I uncertainty (measured by density) and type II uncertainty (measured by coverage) through the concave shape of the peeling trajectory (Fig. 3.3). To ensure to have no wrong-acceptance (density of 1 in the Fig. 3.3), the box has to have wrong-rejection (coverage of 0.62 in the Fig. 3.3). The scenarios that stay within the PBs are therefore relatively close scenarios and are located in a relatively small input region, as confirmed in the Fig. 3.2. No selection of leverage has a coverage of 1 since it covers all the ouput space, but the density is 0.18. This means that 18% of explored scenarios are inside the PBs. This results highlights the difficulty to find strategy for the AFOLU sector to stay within PBs.

The AFOLU sectors strategies that maximize the density contains a reforestation of more than 15% of the pastures and a reduction in the consumption of animal products in the diet with a global demand for ruminant of less than 41 Tkcal.



Box	Selected leverages	Lower bound	Upper bound	Coverage	Density
Box 1	Forest	0.05	0.2	1	0.26
Box 2	Forest	0.15	0.2	1	0.54
Box 3	Forest	0.15	0.2	0.97	0.78
	Diet	0.25	1.0	0.97	0.78
Box 4	Forest	0.15	0.2	0.62	1
	Diet	0.75	1.0	0.62	1

Figure 3.3 – Trade-off between coverage and density of strategies to stay inside PBs in the minimization of uncertainty in 2030

3.4.3 Sensitivity to the uncertainty surrounding the PBs

We then assess the sensitivity of our results to the uncertainty surrounding the PBs values (Steffen et al. 2015) thanks to a Monte-Carlo experiment. Only 16% of the scenarios are able to remain within the planetary boundaries. This reflects the difficulty of remaining in the safe operating space in 2030 and the relations between PBs that doesn't allow the AFOLU sector to reach some combinations (See Fig. 3.2).

Despite the uncertainty surrounding PBs, a major change in diet, a significant reforestation (reforestation of more than 15% of pastures) and a significant increase of NUE are selected respectively 90%, 60% and 58% in the robust strategy to stay within PBs in table. 3.3.

Table 3.3 – Probability of selection of a strategy as a lever to stay inside PBs and average thresholds associated with it.

	DietScen	Forest	NUE	PopScen	RumProd	TradeScen	Total
Probability of selection	0.96	0.6	0.6	0.06	0	0	1.0
Average minimum value	0.75	0.1	0.5	1.0	-	-	-
Average maximum value	1.0	0.2	1.5	2.5	-	-	-

This selection of robust strategies varies according to the thresholds chosen for the different PBs. A high threshold for "biosphere integrity" leads to the selection of a larger number of leverages (a reduction in animal consumption, reforestation of pastures, an increase in the NUE, a reduction in population, an increase in productivity in the ruminant sector in Fig. 3.4) reflecting the difficulty of remaining within this PB.

A low threshold of nitrogen lost in surface waters in PBs leads to an increased proportion of NUE increase and population growth control strategies (Fig. 3.4). For extremely low nitrogen losses in the environment, controlling population growth reduces nitrogen losses due to consumption and the NUE reduces losses in crop production. Reforestation reduces nitrogen losses in surface waters by concentrating crops on good quality land, allowing more efficient use of nitrogen.

A threshold of non-CO_2 emissions mitigated by the AFOLU sector beyond 1.3 $GtCO_{2,eq}$ leads to an increase in the proportion of NUE increase strategies at the expense of reforestation strategies (Fig. 3.4). Reforestation has an ambiguous effect on non-CO_2 emissions from the AFOLU sector, while the increase of NUE has not.

A lower threshold for the remaining forest leads to lower proportions of NUE increase and population control (Fig. 3.4).

3.4.4 PBs at regional scale

In some regions (e.g. Europe), none of the robust strategies tested in this study allows to stay within regional PBs due strong historical environmental change resulting from nitrogen pollution or deforestatrion (Fig. 3.5). China needs to implement many mitigation policies to stay inside PBs (Fig. 3.5). Finally, countries with large areas such as the USA (Fig. 3.5) or the Former Soviet Union (FSU in Fig. 3.7) have patterns similar to those of the world scale, namely the frequent selection of diet

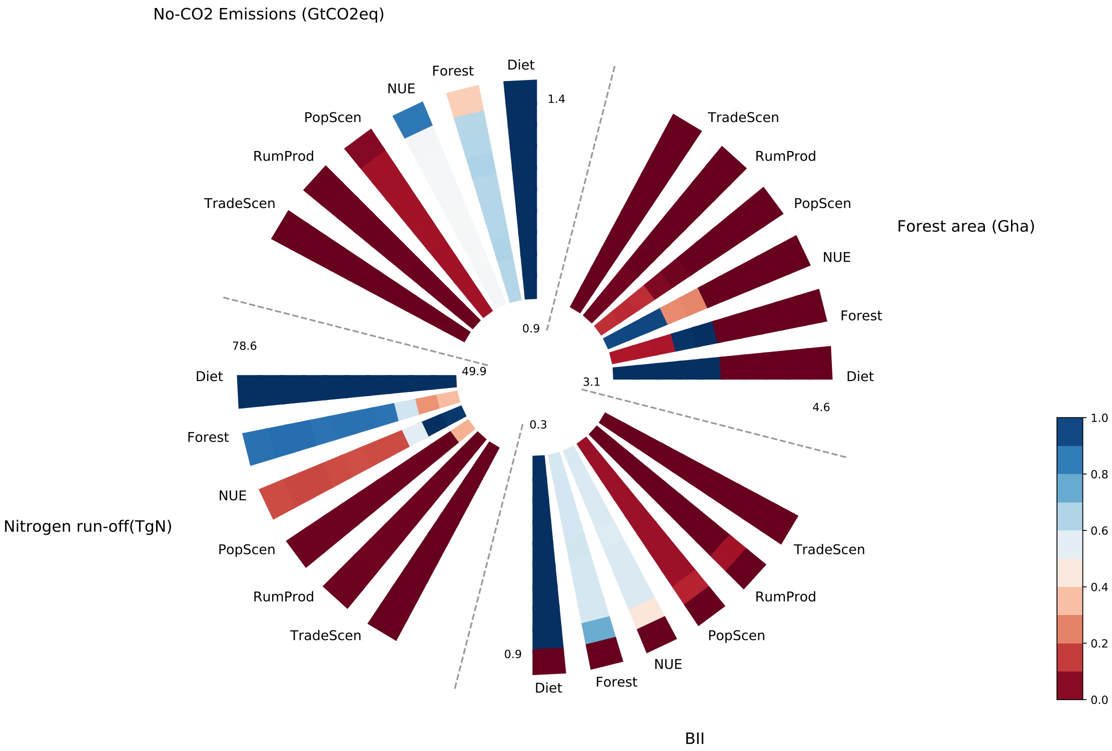
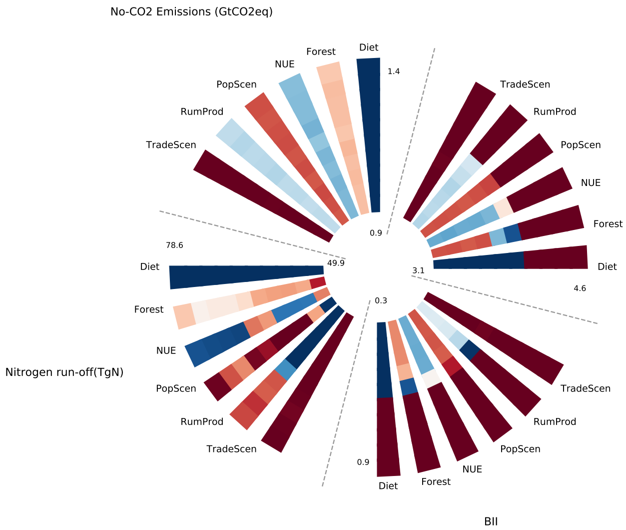
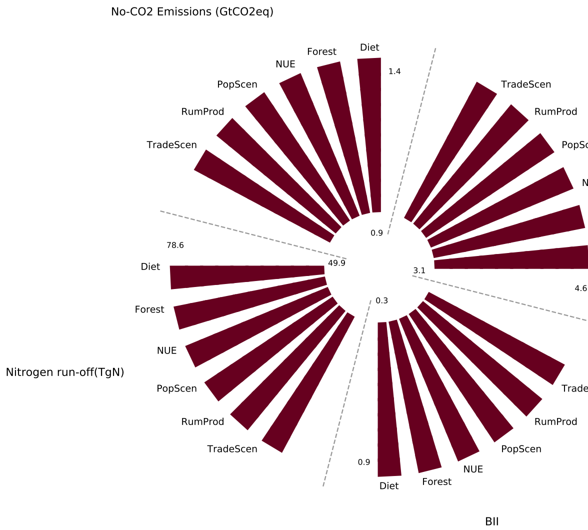


Figure 3.4 – Sensitivity of the selection of strategies to stay inside the safe operating space to different PBs en 2030. A high probability of selection of one strategy is colored in blue and a low probability of selection of a strategy is colored in red.

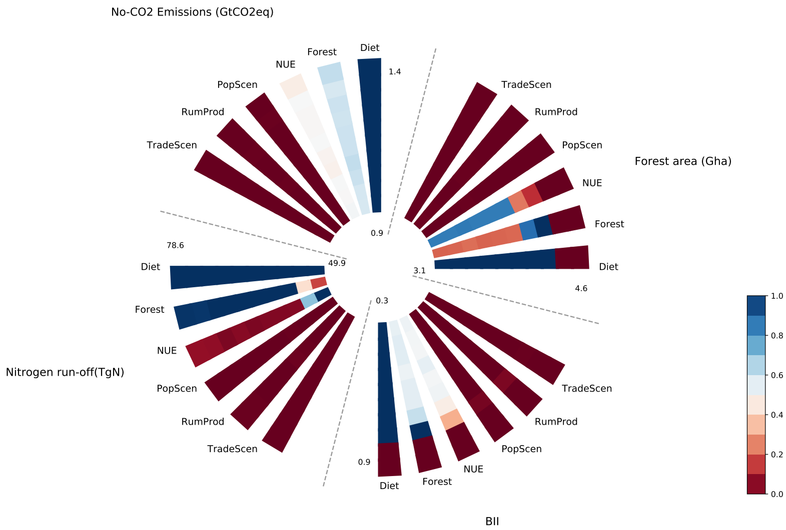
change, a strong reforestation of pastures and an increase in the NUE to remain robustly within the PBs.



(a) China



(b) Europe



(c) USA

Figure 3.5 – Probability to select a strategy (Diet change, reforestation, increase of NUE, Population control, increase of ruminant productivity or liberalization of trade) as a lever to stay within PBs in the strategies which allow to stay inside PBs. PBs are set at different level in a uncertain zone following a Monte-Carlo experiment. An high threshold (outside the circle) for the remaining forest, the BII and the non-CO_2 mitigated emissions and a low threshold (inside the circle) for run-off nitrogen correspond to stringent PBs.

3.5 Sensitivity analysis

3.5.1 Sensitivity analysis to a supplementary indicator of sustainability: adding the food price

The "Food price index" presents trade-offs with other environmental sustainability indicators (Fig. 3.6). A price increase is associated with an increase in BII, thus representing a trade-off between BII and food security, an increase in nitrogen losses, representing a trade-off between food security and reduced nitrogen losses in surface water, and an increase in mitigated emissions representing a trade-off between non-CO_2 emissions reduction and food security. All these findings are consistent with the related literature (Obersteiner et al. 2016, Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle in prep., Humpenöder et al. 2018, Hasegawa et al. 2018)

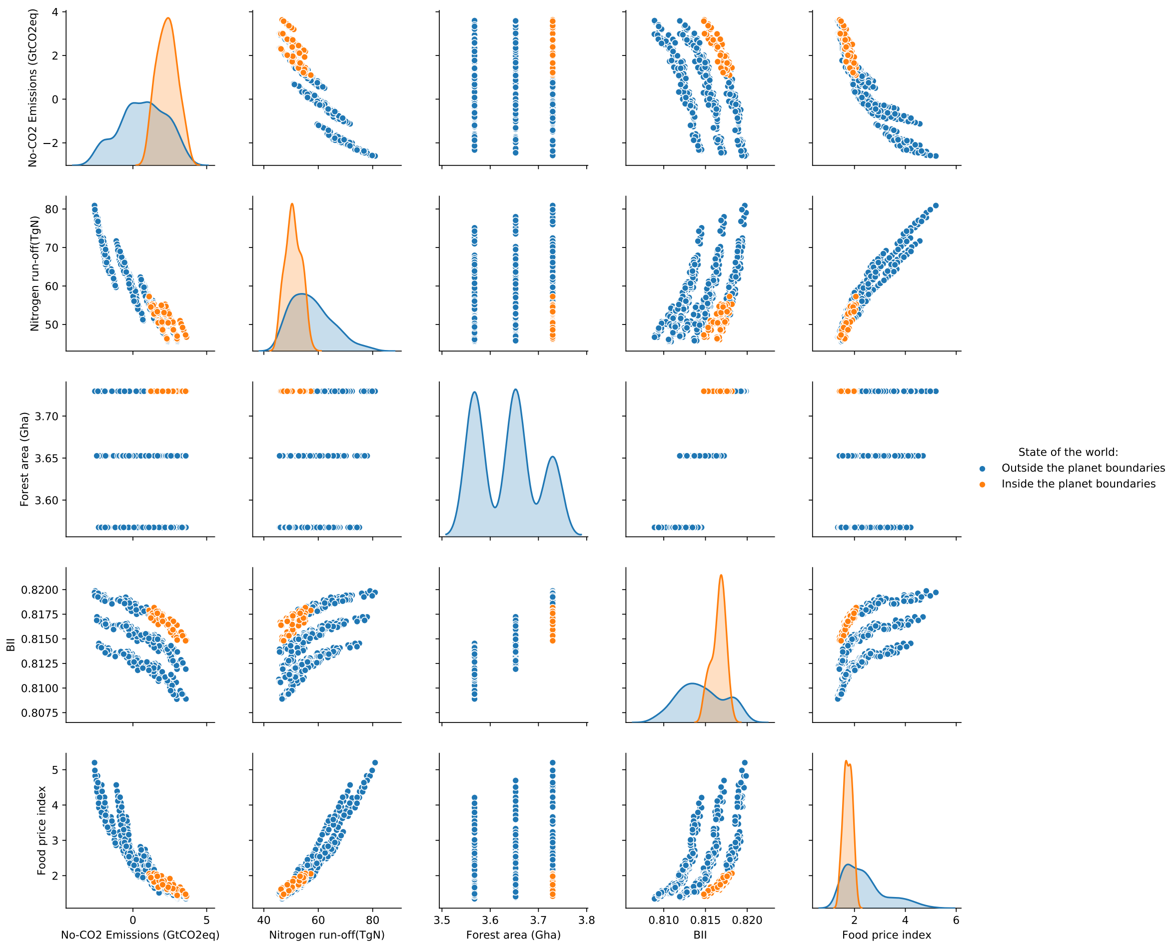


Figure 3.6 – Relations between PBs and food security resulting from scenarios explored in 2030. Graphs on the diagonal of the figureth are density plots of the different sustainability indicators.

The inclusion of a food security indicator as a measure of sustainability does not change the probability of selecting dietary change, reforestation and NUE to stay within the PBs (Table. 3.4).

Table 3.4 – Probability of selection of a strategy as a lever to stay inside PBs and under a food price index of 2 and average thresholds associated with it.

	DietScen	Forest	NUE	PopScen	RumProd	TradeScen	Total
Probability of selection	0.96	0.6	0.6	0.06	0	0	1.0
Average minimum value	0.75	0.1	0.5	1.0	-	-	-
Average maximum value	1.0	0.2	1.5	2.5	-	-	-

3.6 Comparison with other studies

One of the main conclusions of this study is the difficulty of remaining within the PBs in 2030, in particular maintaining the integrity of the biosphere and wide areas of forest. To verify that this conclusion is not due to a bias in scenario choices towards unsustainable scenarios, we compare the ranges of synthetic nitrogen applied to crops (lost nitrogen not being available in some studies), GHG emissions, BII and remaining forest in 2030 with those from other similar studies (Table. 3.5). This reforestation is especially ambitious in forest area scenarios of this study and the impacts on food prices are a little severe in this study due to the ambitious forest scenarios which put pressure on the food system (Stevanović et al. 2017) and inelastic demand in NLU. Due to the diversity of biodiversity indicators and GHG emissions in the studies, we cannot really compare the non-CO2 mitigated emissions and BII of this study with indicators other studies.

Moreover, this bias towards sustainable scenario choices (diet more sustainable than the FAO baseline, reforestation scenarios) can influence the results. In particular, when using the PRIM algorithm to select the box, future AFOLU sectors with deforestation combined with other strategies cannot be selected.

Table 3.5 – Comparison with environmental indicators in 2030 of other studies.

SDG	This study	Obersteiner et al. (2016)	Humpenöder et al. (2018)
Climatic change	7.15-13.39 (Non-CO_2 emissions in $GtCO_{2,eq}$eq/yr)	(-1.94)-2.35 (Non CO_2 and CO_2 $GtCO_{2,eq}$eq/yr)	42-168 (cumsum of non-CO_2 in $GtCO_{2,eq}$eq between 2030 and 2010)
Biosphere integrity	0.81-0.9 (BII)	3-258 (Mha)	-
Forest area	10-172 (Mha compared to 2010)	(-60)-93 (Mha compared to 2010)	(-160)-25 (Mha compared to 2010)
Nitrogen cycle	143-273 (Losses TgNr/yr)	183-232 (Losses TgNr/yr)	117-219 (Fert TgNr/yr)
Food price	1.68-4.41 (index 2010 price)	0.84-1.16 (index 2010 price)	1.04-1.13 (index 2010 price)

3.7 Discussion

This study is consistent with the available literature by showing the difficulty of maintaining an AFOLU within the four PBs (Steffen et al. 2015, Newbold et al. 2016, De Vries et al. 2013). In particular, strategies to remain within PB related to the integrity of the biosphere (whose threshold has currently been crossed) imply significant modifications of the AFOLU sector which are beyond the scope of this study. Because PB for biodiversity integrity is crossed until 2030 regardless of land use, quantifying the impacts of overshooting this threshold seems necessary (Newbold et al. 2016).

However, increasing the nitrogen use efficiency, reforesting pastures and shifting diets will allow us to stay within the other PBs in 2030 regardless of the thresholds used to define PBs within the limits of the scenarios in this study (Fig. 3.4). Uncertainty surrounding socio-economic conditions, the implementation of other environmental strategies or PB thresholds does not prevent our societies from taking robust decisions to stay within the PBs. In line with the literature, the selection of a strategy to increase the NUE with a change in diet reduces pressure on land as explained in Obersteiner et al. (2016) and thus reduces the impact of the AFOLU sector on nitrogen pollution of surface waters, GHG emissions from the sector and food prices (Humpenöder et al. 2018). Unlike these studies, we stress here the importance of explicitly integrating environmental protection measures through reforestation into land-use strategies. Intact forests have exceptional value for biodiversity beyond biodiversity hotspots (Watson et al. 2018) and preventing soil degradation (Miles & Kapos 2008). Obersteiner et al. (2016) found that a dietary change witout measures dedicated to its protection leads to changes in forest area of -38.2 Mha in 2030 compared to 2010. In Steffen et al. (2015), 62% of the forest is actually remaining. In Obersteiner et al. (2016), forest areas decrease, our study suggest instead that reforestation is an important strategy to avoid crossing the forest PB.

Another salient result of this study is the possibility of returning to the PB associated with the global nitrogen cycle, when this threshold is currently crossed (Steffen et al. 2015, De Vries et al. 2013) by implementing strategies to increase NUE, change diet and reforestation. However, this result is not robust when considering regional thresholds for nitrogen losses, since Rest of LAM, Rest of Asia, Africa, Middle-East, India, Brazil and Canada require the deployment of strategies that go beyond the scenarios studied in this study.

This study also specifies the relationships between the different PBs linked through agricultural intensification or land use. In particular, the trade-off between reducing nitrogen losses and increasing biodiversity on a global scale (Fig. 3.2) depends on the rebound effects resulting from intensification (Meyfroidt et al. 2018) and the impact of intensification on biodiversity (Phalan 2018). In NLU, a lower consumption of animal products in 2030 compared to the baseline results in a lower development of the intensive system in favour of the pastoral system (Brunelle et al. 2018). In PREDICTS models, these extensive patches have high levels of SR but low levels of composition similarity (See coefficients of the PREDICTS models in section. and Purvis et al. (2018) for a broad description of the modelling framework). However, this modeling framework does not take into account relationships that do not pass through land-use such as the influence of climate on species distribution (Thuiller et al. 2014), biodiversity on agricultural production (FAO 2019) and the water cycle not represented in this study on agricultural production and nitrate concentration in surface waters.

In addition, a major liberalisation of international trade, to relocate agricultural production in temperate regions and promote carbon storage and biodiversity protection in tropical regions (West et al. 2010, Tilman et al. 2017), is being confronted in our study with PBs on the nitrogen cycle. In this study, the hypothesis of an increasingly inefficient use of resources with an increase in yield (See eq. 2 in NLU's description in supplementary information) leads to a sharp increase in nitrogen losses in regions with high levels of intensification. This trend observed in some regions such as China, can be offset by the use of more efficient technologies in nitrogen use as is currently done in the United States or Europe (Zhang et al. 2015). This improvement in efficiency in nitrogen management is not yet represented by this framework.

The selection of strategies to stay within the safe operating space of the planet through a robust decision making technique allows us to discuss the description of future socio-economic futures "under the green growth paradigm" (SSP1 in van Vuuren et al. (2017)). In this study, we show that trade liberalization or low population growth (van Vuuren et al. 2017, Popp et al. 2017) does not necessarily imply sustainable development levels, and vice versa. However, the robust selection of diet change strategy to stay within the PBs highlight the importance for decision-makers not to consider this aspect as context but to address it in environmental policies. Understand how to influence food consumption choices towards sustainable diets through a meat tax (Bonnet et al. 2016) or the influence of climate on behaviour (Beckage et al. 2018) appears to be an important issue for current research.

3.8 Supplementary informations

3.8.1 Regional population scenarios

The regional population follows population evolution described by Kc & Lutz (2017) and transposed to NLU regions (Fig. 4).

Table 3.6 – Range of regional population (Mia head) in 2030.

Regions	Low	Medium	High
USA	0.3631	0.3605	0.3368
Canada	0.0419	0.0414	0.0379
Europe	0.6560	0.6544	0.6314
OECD Pacific	0.2111	0.2083	0.1985
FSU	0.2752	0.2806	0.2846
China	1.3747	1.3960	1.4145
India	1.4601	1.5299	1.6050
Brazil	0.2172	0.2242	0.2319
Middle-East	0.2953	0.3092	0.3219
Africa	1.4535	1.5382	1.6327
Rest of Asia	1.2921	1.3458	1.4076
Rest of LAM	0.4580	0.4777	0.5091

3.8.2 Regional diet scenarios

Plant and animal food demand are described by the scenario proposed by FAO Alexandratos & Bruinsma (2012) which will be the most intensive scenario in animal calorie, a scenario that will achieve an emission reduction of 4.3 $GtCO_2$ in 2100 described in Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle (in prep.) which will be the least intensive scenario in animal products and an intermediate scenario that sets a food consumption between the two previous scenarios (See table. 3.7).

Table 3.7 – Regional diet in 2030 (kcal/cap/day)

	Baseline		DC[1]		DC - Baseline	
Regions	Plant Food*	Animal Food*	Plant Food*	Animal Food*	Plant Food[2]*	Animal Food[2]*
Africa	2586	221	2564	436	-22	215
Brazil	2466	952	2686	343	220	-609
Canada	2544	1394	2686	314	142	-1080
China	2682	758	2594	406	-88	-352
Europe	2544	1264	2686	314	142	-923
FSU	2543	905	2686	314	143	-591
India	2517	424	2594	406	87	-18
Middle-East	2838	517	2686	314	-181	-203
OECD Pacific	2544	905	2686	314	142	-591
Rest of Asia	2682	541	2686	406	-88	-227
Rest of LAM	2466	794	2686	406	220	-388
USA	2544	1338	2686	314	324	-1024

[1] the built of DC is described in Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle (in prep.)
[2] Difference of consumption in diet change scenario
compared to baseline
* In kcal/cap/day

3.8.3 Regional forest scenarios

Table 3.8 – Range of regional forest area (Mha) in 2030.

Regions	Low	Medium	High
USA	266.9665	273.6732	280.0170
Canada	554.2264	554.7080	555.1887
Europe	129.0436	131.4190	133.6982
OECD Pacific	61.0349	70.4074	77.5305
FSU	866.5668	877.1424	887.4286
China	170.9614	179.8984	187.9048
India	30.2381	30.5460	30.8469
Brazil	452.9123	459.0018	464.9084
Middle-East	0.0000	0.0000	0.0000
Africa	338.4215	362.9023	384.0400
Rest of Asia	346.7231	352.0383	357.1726
Rest of LAM	350.4145	360.9477	370.8088

3.8.4 Regional plant sector productivity

The regional plant sector productivity is defined in NLU by the initial slope of the production function. This slope can also be related to the Nitrogen Use Efficiency (NUE) (See supplementary information for a presentation of the equations) whose regional values are presented in the Table. 3.9.

Table 3.9 – Range of regional NUE in 2030

Regions	Low	High
USA	0.0177	0.8298
Canada	0.0154	0.8343
Europe	0.0119	0.8177
OECD Pacific	0.0130	0.7898
FSU	0.0447	1.1989
China	0.0076	0.5550
India	0.0033	0.2108
Brazil	0.0443	0.6232
Middle-East	0.0000	0.3192
Africa	0.0137	0.6352
Rest of Asia	0.0123	0.4083
Rest of LAM	0.0249	0.6550

3.8.5 Regional livestock sector productivity

To represent an intensification of livestock farming, we change the efficiency of systems to convert ingested vegetal calories into animal calories either by setting it at its 2001 value (10.4 $kcal_{veg}/kcal_{anim}$), or by increasing it according to the scenario proposed by Bouwman et al. (2005) until it reaches 12.9 $kcal_{veg}/kcal_{anim}$. In Table. 3.10, we present the regional conversion coefficient of vegetale alories into animal calories.

Table 3.10 – Range of regional conversion of plant product to animal product ($kcal_{veg}/kcal_{anim}$) in 2030.

Regions	Low	High
USA	8.0604	7.7211
Canada	9.2861	8.4002
Europe	6.5210	5.6539
OECD Pacific	8.6368	6.0563
FSU	8.8257	8.1435
China	12.4976	10.8784
India	15.9083	12.0645
Brazil	23.0806	23.0806
Middle-East	10.0098	8.2658
Africa	24.4435	18.2852
Rest of Asia	14.9424	12.4271
Rest of LAM	19.9935	15.8649

3.8.6 International trade scenarios

In Table. 3.11, we present the imports of the different NLU regions in the different scenarios.

Table 3.11 – Range of regional plant imports (Tkcal) in 2030.

Regions	Low	Medium	High
USA	105.1858	124.5109	144.1464
Canada	64.1198	74.3051	84.3414
Europe	407.1421	467.1169	526.9992
OECD Pacific	243.1458	285.2099	328.1755
FSU	64.7089	74.8361	85.0303
China	87.9730	101.3466	114.8243
India	351.6776	377.5340	401.8639
Brazil	71.1348	83.9025	96.8661
Middle-East	342.0749	357.7841	372.9089
Africa	453.3650	505.2649	555.4108
Rest of Asia	542.8512	590.4782	636.9114
Rest of LAM	227.5210	262.5716	297.5108

3.8.7 Calculation of PBs

Global PBs

The first PB is the remaining forest area. This variable was chosen as the three major forest biomes-tropical, temperate and boreal-play play a stronger role in land surface-climate coupling than other biomes (Snyder et al. 2004, Bonan 2008). Forest degradation in a region above a certain threshold can lead to regional climate changes that would threaten the sustainability of the remaining forest. For example, tropical forests have significant feedbacks to climate via changes in evapotranspiration when they are converted to non-forested systems.

The area of forest that must be maintained to ensure the stability of the local climate and forest system depends on the type of forest concerned. 85% of the tropical forest, 50% of the temperate forest, 85% of the boreal forest and 40% of the pastures qualified as "other pastures" (in the classification of Ramankutty et al. (2010)) must be maintained to avoid massive conversion of the rest of the forest (Steffen et al. 2015). We recalculate this boundary from a map describing the distribution of potential vegetation without anthropogenic influence (Ramankutty et al. 2010). The preservation of 2.1 Gha of tropical forest, 1 Gha of temperate forest, 0.77 Gha of boreal forest and 0.66 million "other forests" respectively allows to remain within the safe operating space. This represents 3.7 Gha of forest worldwide.

The second PBs is the human-induced nitrogen cycle. An excessive flow of nitrogen unbalances the nitrogen cycle through an increase in N_2O emissions into the atmosphere and eutrophication of aquatic environments by streaming the reactive nitrogen lost from agricultural environments. In this PBs, we are only interested in leached nitrogen because the N_2O emitted into the atmosphere is taken into account in the PBs concerning climate change. It is calculated by NLU as the nitrogen input into the agricultural system remaining after harvest, after sequestration of nitrogen in the residues for the next crop and volatilization to the atmosphere.

To determine the flow of anthropogenic nitrogen to the environment compatible with an ecological balance in terrestrial aquatic environments, De Vries et al. (2013) determine a risk indicator between 0.5 and 0.67 which describes how much the critical limit has been exceeded. This risk indicator divides the critical limit of NO_3 concentration in run-off surface water (1.0-2.5 mg.L^{-1}) by the concentration of nitrogen in run-off surface water in 2001 computed with the IMAGE model (Bouwman et al. 2009). We use the same method and deduce our own critical limit based on this risk indicator and our own nitrogen in run-off surface water at global scale ($N_{\text{run-off,PB}}$=RI × $N_{\text{run-off,NLU,2001}}$). We deduce a $N_{\text{run-off,PB}}$ (see section. 3.8.8 for details on the computation of nitrogen run-off) included in range 49-78 TgN.yr^{-1}. In this study, we use the lower value (49 TgN.yr^{-1}) as a threshold to maintain a sustainable nitrogen cycle.

The third PB is the atmospheric CO_2 concentration that the authors set between 350 and 550 ppm. This concentrations have been computed in the IPCC's reports (IPCC 2007). CO_2 concentrations above these levels could severely destabilize the climate system, and in particular cause an increase in extreme events. (IPCC 2012). To stay below the 450 ppm threshold, Wollenberg et al. (2018) estimate that the AFOLU sector must reduce its emissions by about 1 $GtCO_{2,eq}$ of non-CO_2 emissions in 2030 compared to the baseline.

The fourth PB studied is the integrity of the biosphere through the Biodiversity Intactness Index (BII). This indicator is used in Steffen et al. (2015) to estimate damages from human activities on large-scale ecological processes. It aims to quantify changes in individual abundance and changes in the composition of species communities (Purvis et al. 2018). This PB seems more complicated to establish on a global scale because it is still subject to a lot of uncertainty (Steffen et al. 2015) and seems to make sense on a much more local scale (Newbold et al. 2016). We therefore choose not to set a threshold on BII in the first instance and to perform a sensitivity analysis at different levels of BII.

Regional PBs

For environmental eutrophication, the BII or the remaining forest area are more relevant at the regional scale than at the global one (Steffen et al. 2015). We therefore apply the calculation of these PBs at the regional level to establish the regional AFOLU sectors that remain within the regional PBs (Fig. 3.12). We still take into account the mechanisms at the global level through international trade to take into account tele-decoupling, which plays a major role in the environmental impact of the land-use (Henders et al. 2015, Kastner et al. 2014, Marques et al. 2019, Oita et al. 2016). The PB on climate change is a global PB due to the global dynamic of the climate. To reach a 450ppm CO_2 concentration, a global emission budget can be computed and distributed among regions. The distribution of the mitigation effort in the AFOLU sector is outside the scope of this study. The AFOLU sector stay within the climate change PB on the condition that the global mitigation effort of the AFOLU sector is 1 $GtCO_{2,eq}$. Similary to the global level, threshold for BII is crossed in most region of the xorld. We perform a sensitivity analysis on this PB to undertand its influence on the selection of robust strategies in the AFOLU sector.

3.8.8 NLU and PREDICTS models

A description of the modelling framework is provided in the Method part.

Table 3.12 – Regional PBs computed in this study.

Regions	Forest area (Mha)	N_{losses} (TgN)
USA	141.3	3.5
Canada	210.6	0.5
Europe	151.3	5.3
OECD Pacific	19.9	0.7
FSU	304.9	0.7
China	86.3	11.5
India	139.1	9.9
Brazil	366.8	1.8
Middle-East	1.2	0.7
Africa	356.5	1.0
Rest of Asia	279.1	4.8
Rest of LAM	326.0	1.8

Computation of nitrogen run-off

Nitrogen that flows into surface waters represents a share (0.26 estimated from Bouwman et al. (2005) as the share of leaching nitrogen in "balance nitrogen" in 2000) of nitrogen lost from plant crops, pastures and household consumption. We use the nitrogen balances described in Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle (in prep.) to determine the nitrogen lost from crops. For pastures, we use nitrogen lost from a reduced nitrogen balance:

$$\mathrm{N}_{deposition} + \mathrm{N}_{excreted} + \mathrm{N}_{synthetic} + \mathrm{N}_{\mathrm{BNF}} = \mathrm{N}_{losses} + \mathrm{N}_{grazing} \tag{3.1}$$

with $N_{deposition}$ nitrogen from deposition on pasture,$N_{excreted}$ nitrogen excreted by ruminant on pasture, $N_{synthetic}$ synthetic nitrogen, N_{BNF} biologically fixed nitrogen, N_{losses} lost nitrogen, $N_{grazing}$ grazed nitrogen, $N_{volatilization}$ volatilized nitrogen.

A deposition rate per hectare $N_{deposition,rate}$ is computed in reference year based on deposition rate computed in Prudhomme, De Palma, Dumas, Gonzales, Levrel, Leadley & Brunelle (in prep.). This coefficient is then applied to the pasture area calculated by NLU each year to estimate the nitrogen deposited on the pastures. Nitrogen fertilization and biologically fixed nitrogen (BNF) are computed in the same way based on fertilizer consumption computed from respectively FAOSTAT (2011) and Herridge et al. (2008). An excreted manure coefficient per ruminant calorie produced is computed at the reference year and applied to the ruminant production each year to compute the amount of excreted manure ($N_{excreted}$). A grazed nitrogen coefficient per grass produced on pasture is computed at the reference year and applied to the grazed grass each year to compute the amount of grazed nitrogen ($N_{grazing}$). We then applied the computed NUE in the reference year for the other years to compute lost nitrogen from pasture:

$$\text{NUE} = \frac{\text{N}_{grazing}}{\text{N}_{deposition} + \text{N}_{excreted} + \text{N}_{synthetic} + \text{N}_{\text{BNF}}} \tag{3.2}$$

$$\text{N}_{losses} = (1 - \text{NUE}) \times (\text{N}_{deposition} + \text{N}_{excreted} + \text{N}_{synthetic} + \text{N}_{\text{BNF}}) \tag{3.3}$$

Table. 3.13 presents nitrogen balance of pasture worldwide in NLU.

Table 3.13 – Nitrogen balance of pasture in 2001 (TgN.yr^{-1}

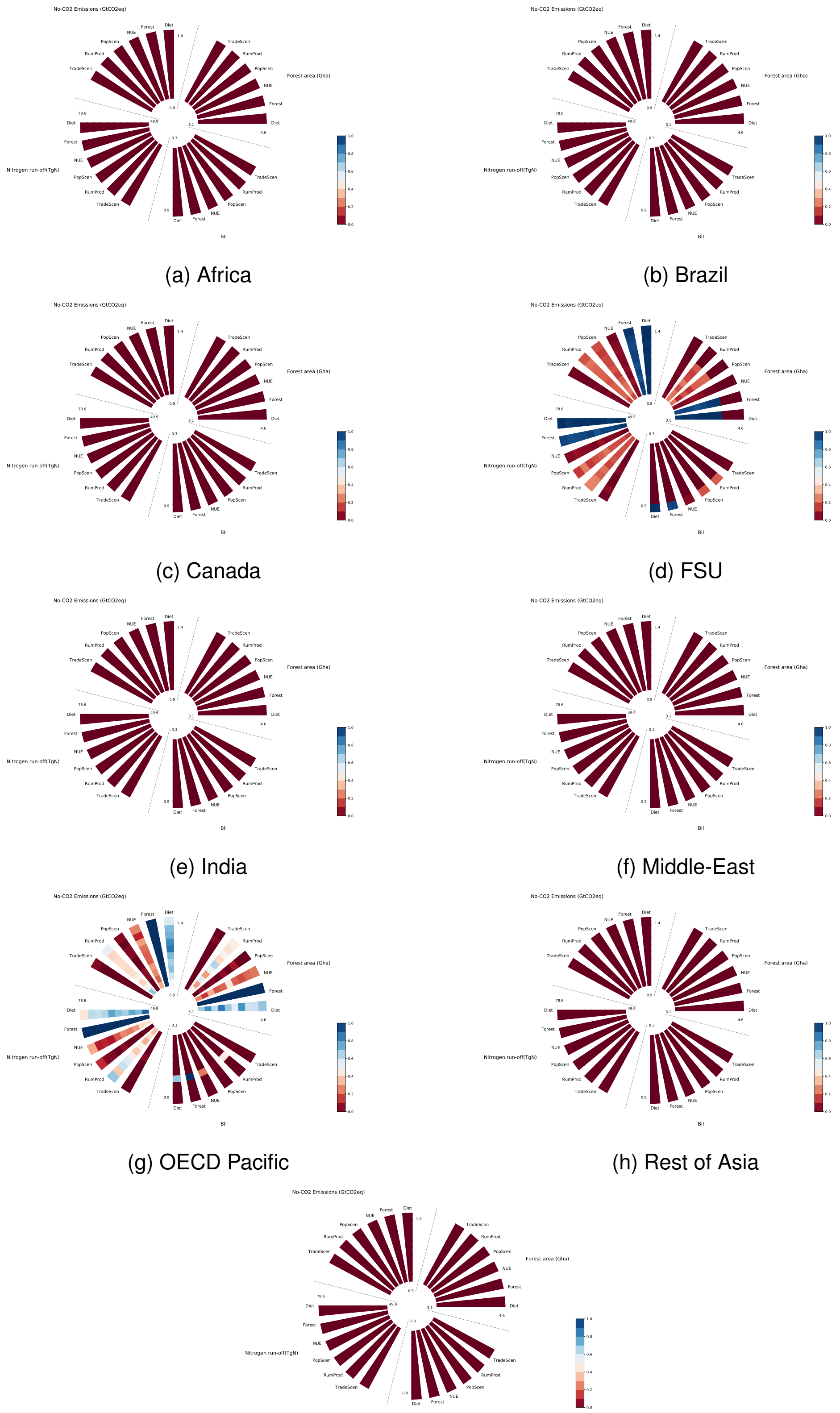
Components	NLU
Deposition	31
Synthetic Fertilization	7.2
Excreted manure	41
BFN	12.1
Grazed Nitrogen	60
Lost Nitrogen	31
NUE	0.65

Among the nitrogen lost, the nitrogen lost from household consumption waste represents 15% (similar rate at Bodirsky et al. (2014)) of the nitrogen consumed. This consumed nitrogen is the product of the regional food demand defined in the food demand scenarios described in the Table. 3.7 with the nitrogen content of this feed calculated by GLOBAGRI (Ranganathan et al. 2016) model and is summarized in the following Table:

Table 3.14 – Regional nitrogen content in diet in 2001

	Plant Food			Ruminant Food			Monogastric Food			Total
Regions	Energy demand[1]	Nitrogen content[2]	Nitrogen demand[3]	Energy demand[1]	Nitrogen content[2]	Nitrogen demand[3]	Energy demand[1]	Nitrogen content[2]	Nitrogen demand[3]	Nitrogen demand[3]
USA	328.5	16.2	5313.0	78.6	68.6	5392.5	59.7	79.5	4750.1	15455
Canada	32.8	17.4	572.1	8.6	68.3	587.0	5.7	76.5	437.8	1597
Europe	583.9	18.6	10888.0	145.4	57.6	8371.2	100.5	65.8	6609.4	25868
OECD Pacific	173.1	18.6	3213.1	19.8	63.5	1256.0	22.5	76.8	1725.2	6194
FSU	254.5	19.6	4978.6	43.6	54.1	2360.9	19.1	73.2	1399.8	8739
China	1165.8	25.4	29563.8	53.7	63.8	3424.0	188.6	44.8	8442.5	41431
India	826.2	17.5	14469.6	63.1	44.2	2788.1	4.9	73.6	358.0	17614
Brazil	160.0	15.3	2442.1	25.8	69.8	1803.3	18.6	61.3	1137.8	5383
Middle-East	164.2	19.7	3233.1	14.5	52.8	764.4	7.5	94.3	702.5	4700
Africa	778.5	19.1	14873.4	40.4	57.1	2308.1	12.3	76.5	942.3	18123
Rest of Asia	727.7	15.8	11532.3	38.7	52.4	2029.3	27.2	53.9	1467.5	15028
Rest of LAM	298.1	15.9	4736.3	42.7	65.6	2797.5	26.0	66.1	1721.4	9254

[1] In Tkcal
[2] In gN/kcal
[3] In TgN



(a) Africa

(b) Brazil

(c) Canada

(d) FSU

(e) India

(f) Middle-East

(g) OECD Pacific

(h) Rest of Asia

(i) Rest of Latin America

Figure 3.7 – Regional robust strategies for different level of PB

General conclusion

In this conclusion, we will firstly summarize the elements that answer the main question of this thesis: how does the combination of environmental protection strategies in the AFOLU sector influence the global sustainability of this sector. In this first section, we will also explain how our results depend on our modelling assumptions by comparing them with other studies. In the perspectives section, we will discuss the impacts of mitigation strategies that could not be addressed because of the choice of modelling framework and suggest ways either to improve the modelling framework used in this thesis, or to use other modelling frameworks to further the analysis.

Main findings

The main contribution of this thesis is to take into account in the same modelling framework (i) the impact of different land-covers (forest, pasture and cropland) and different land-use intensity of pasture and cropland on biodiversity, (ii) the computation of nitrogen balances, (iii) the computation of GHG emissions by source according to the IPCC method (IPCC 2006*b*) ; (iv) an agricultural production model from the plant and animal sectors (Souty et al. 2012). With the integration of these different elements, we study the impacts of environmental protection strategies in the AFOLU sector on four sustainable development goals: SDG 2 (Zero Hunger), SDG 6 (Water and sanitation), SDG 13 (Climate change), SDG 15 (Biodiversity, forests, desertification). This section will specify the contribution of each chapter in quantifying the impacts of environmental protection strategies in the AFOLU sector on the SDGs and put them in perspective with other studies.

Impact of a diet change in Europe on GHG emissions of the AFOLU sector

In chapter. 1, we assess the impact of dietary change strategies and of a combination of dietary change with reforestation on GHG emissions from the AFOLU sector (indicator used here for the SDG13). The specificity of this study compared to the existing literature is that we disaggregate the total GHG emission reduction due to dietary change (212 $MtCO_{2,eq}$) with respect to the different sources of emission.

The main emission reduction was achieved in the livestock farming sector (65%) through a reduction in emissions due to enteric fermentation and manure management (38%), and feed production (27%). The second source of reduction is the reduced fertilization (31%), mainly linked to economic choices regarding production allocation and intensification levels. In this case, the main part of the emission reduction is exported out of Europe, as Europe re-imports emissions in the plant food sector by reducing its domestic land needs (mainly pastures) and improving its trade balance. In the NLU, the reduction in emissions per tonne of substituted ruminant dry matter (0.07 $MtCO_2$,eq/ktDM) is slightly higher than other studies (0.038 in Stehfest et al. (2009), 0.03 in Popp et al. (2010) and 0.025 in Westhoek et al. (2015)), except Ranganathan et al. (2016). The reduction in emissions due to a reduction in fertilization appears to be slightly higher in the NLU due to a higher use of synthetic fertilizer within the NLU (Table. 1.6 in chapter. 1).

The sources of emission reduction are different when dietary change is combined with a reforestation strategy. In this combination of strategies, most of the emission reduction are generated through carbon storage in forests (53% of the emission reduction or 134 MtCO2), and non-CO2 emissions from the agricultural sector decrease compared to the baseline (- 119 MtCO2) but significantly less than in a dietary change alone (-211 tCO2). The combination of forest scenario and diet change show interactions.

Impact of combinations of diet change, reforestation of pasture and biofuel production on biodiversity and food price

In chapter. 2, we decide to examine in more detail the impact of the combination of a forest scenario and a diet change scenario on food prices and biodiversity. We also added second-generation biofuel production in mitigation scenarios. In line with the literature, we find that second generation bioenergy production is done at the expense of both SDG 2 (food security) (Stevanović et al. 2017, Havlík et al. 2011, Hasegawa et al. 2018, Humpenöder et al. 2018) and SDG 13 (Terrestrial biodiversity) (Hill et al. 2018, Obersteiner et al. 2016, Heck, Gerten, Lucht & Popp 2018). The modelling framework chosen here takes into account both the effects of second-generation biofuel cultivation on biodiversity, but also the indirect effects through the intensification of the rest of the agricultural system due to biofuel production. In this study, we estimate that the production of 112 EJ of biofuel (SSP2 and 4.3 $GtCO_2$ mitigated in the AFOLU sector) reduces the BII by 18%. In Heck, Gerten, Lucht & Popp (2018), the production of 173 EJ of biofuel (SSP1,RCP2.6) results in a 30% loss of BII in 2050. These results cannot be compared directly because of the difference of biofuel production and in the SSP, but it seems that impacts of biofuel production are increasingly negative with the biofuel production.

Also in line with the literature, a combination of a dietary change with a forest protection scenario increases both the level of biodiversity and food security. This biodiversity increase is smaller a this combined scenario (dietary change+reforestation) than in a scenario of protected areas alone (Visconti et al. 2015). Similarly, the food price is higher in a dietary change combined with a reforestation scenario than in a dietary change alone (Stevanović et al. 2017).

The reduction of the availability of agricultural land in land protection strategies leads to an intensification of cropland (Stevanović et al. 2017, Koch et al. 2019). Unlike Visconti et al. (2015), we take into account the impact on biodiversity of this intensification of agriculture, which is due to the reduction in the available area for agriculture. We also take into account the impact on biodiversity of larger pastoral system in the dietary change compared to the baseline. On the other hand, the forest scenario has the disadvantage of distributing reforestation according to the presence of pasture (Griscom et al. 2017), without taking into account the impact of specific geographical features such as the presence of biodiversity hotspots (Obersteiner et al. 2016) or protected areas (Visconti et al. 2015). Due to the absence of an explicit link between the tension in the land market and the forest area in the modelling framework (Richards et al. 2014), we cannot take forest degradation into account in this thesis. In that sense, biodiversity assessment might be optimistic.

Robust strategies in the AFOLU sector to stay within planetary boundaries

In chapter. 3, we select a combination of NUE increase, forest protection and dietary change as a pathway for the AFOLU sector to stay robustly within the planetary boundaries (except for the biosphere integrity which requires change outside the scenarios studied to be respected). This strategy is also robust to the uncertainty surrounding planetary boundaries thresholds since an increase in the NUE, a dietary change and reforestation are selected respectively in 60%, 96% and 60% of strategies to stay within planetary boundaries (Table. 3.3 in chapter. 3). An increase of NUE is a less selected leverage to stay within PBs when the threshold of forest is especially high. Reforestation is a less selected leverage to stay within PBs when the threshold on nitrogen loss or non-CO_2 emissions is particularly high. This last case may occur in emissions trajectories with low emissions in the first part of the century and to avoid the use of negative emissions in the second half of the century to stay below 1.5° of climate warming (Tanaka & O'Neill 2018).

The selection of forest protection in a land-use based strategy for a sustainable land-use future differs from similar studies such as Humpenöder et al. (2018) and Obersteiner et al. (2016). Obersteiner et al. (2016) overlooks the impact on biodiversity of strategies including reforestation by considering only biodiversity hotspots. Humpenöder et al. (2018) does not directly quantify impacts of maintaining forest on global biodiversity because authors use forest area as a proxy for biodiversity. Moreover authors focus on the large-scale deployment of second-generation biofuels leaving few space for sustainable reforestation. In order to respect planetary boundaries, the forest area to maintain is set between 75-54% of original forest cover (Steffen et al. 2015) which represents 3100-4600 Mha using Ramankutty et al. (2010) to compute the original forest cover. This corresponds to a forest area change of -900 to +600 Mha compared to the current 4000 Mha of forest (FAO 2010). The forest protection scenarios in Humpenöder et al. (2018) and Obersteiner et al. (2016) are rather in the lower range of these thresholds with reforestation scenarios between +130 and +0 Mha respectively compared to 2010. The use of lower reforestation scenarios than in chapter. 3 is partly due to the combination with bioenergy production scenarios (Humpenöder et al. 2018). These studies underestimate the risks of degradation of this ecosystem by not taking into account forest protection measures. This forest degradation can lead to non-linear change threatening the maintenance of degraded forests (Bonan 2008). These non-linear changes are underlying the PB related to remaining forest. In the chapter. 3, we take it into account by using a threshold for this PB.

A limit of our framework is that our forest scenarios may overestimate the impact of maintaining these forests on biodiversity because we don't take into account the degradation of other forests due to increased pressure on the agricultural system and the lack of representation of the impacts of forest degradation on biodiversity (De Palma et al. 2018, Leclere et al. 2018, Hill et al. 2018).

The almost systematic selection of dietary change (96% of strategies) as a strategy to stay within the planetary boundary highlights the need to understand how to influence food consumption toward sustainable diets (Bonnet et al. 2016, Beckage et al. 2018) and include it in models not only as an element of context (Popp et al. 2017) but as environmental policies. The positive impact of dietary change on environmental indicators must be reduced by taking into account the associated rebound effect. A reduction in demand leads to a reduction in the food price compared to the baseline. The rebound effect corresponds to environmental impacts associated to the transfer of consumption to other sectors due to increased savings (Grabs 2015).

Perspectives

We present here future research paths on the impact of land-use based strategy on sustainability indicators in the AFOLU sector.

Missing land-use based mitigation strategies

In the AFOLU sector, we can distinguish between emissions from land-use changes, which are mainly CO_2 emissions (4.8 Gt CO_2 in 2012) and agricultural emissions, which are mainly non-CO_2 emissions (5.4 Gt) (Tubiello et al. 2015). In Frank et al. (2018), the authors decompose the mitigation potentials of no-CO_2 emissions from the agricultural sector into 3 types of options:

— Technical options: Technical options reduce agricultural emissions using technologies like anaerobic digesters, feed supplements, nitrogen inhibitors, nitrogen optimal application, improved cropping practices, improved rice management... In chapter. 3, we test different increase in the conversion rate of grass by ruminants and for increasing nitrogen use efficiency (NUE).

— Structural changes: They correspond to changes within the agricultural sector such as transition towards high intensity management systems or relocation of production across regions through international trade. This structural change is represented endogenously in the model and is therefore integrated into the different mitigation strategies evaluated in this thesis. Strategies for liberalizing international trade are also assessed in chapter. 3.

— Consumers' change: It reduces consumption of GHG-intensive products and waste. Dietary change scenarios are assessed in chapters. 1 with legume production in Europe, 2 and 3.

An important source of emissions is nitrogen fertilization of crops linked with a doubling use of synthetic fertilizer since 1960 (Galloway et al. 2008). In the thesis, a detailed nitrogen balance is used but change in NUE and fertilization control is modelled simply, without taking into account processes other than the decreasing marginal efficiency of fertiliser application. The efficient use of nitrogen inputs is crucial both for reducing emissions from fertilization and for limiting runoff in surface water.In the thesis, a detailed nitrogen balance is used but change in NUE and fertilization control is modelled simply, without taking into account processes other than the decreasing marginal efficiency of fertiliser application. The NUE is calculated from the balance elements established in chapter. 1. Taking into account NUE improvement scenarios would make it possible to change the production function so that it reproduces the NUE changes observed on a global scale (Zhang et al. 2015). In particular, a Kuznet nitrogen curve appears to be emerging to describe the improvement in nitrogen use efficiency with increased yield. A major challenge in representing efficient use of nitrogen, is taking into account how nitrogen fertilization interacts with other input uses and different agricultural practices.

Integrating other SDGs

In this thesis, we limit our analysis of sustainability indicators to nitrogen runoff in surface waters (chapter. 3), non-CO_2 emissions (chapter. 1, 2, 3) and CO_2 emissions (chapter. 1, 2), food prices (chapter. 2 and 3) and biodiversity (chapter. 2 and 3).

We could also include a water cycle to study the SDG.6 (water and sanitation) because of the important freshwater withdrawals of the AFOLU sector (70% of the freshwater withdrawals on earth in Millennium Ecosystem Assessment Board (2005)) for irrigation. The AFOLU sector is indeed in competition with other sectors for access to water (Neverre & Dumas 2015). Including a water cycle seems also relevant to study the large-scale deployment of bioenergy and limitation by available water (Bonsch et al. 2016) and for adaptation to climate change in dry areas.

We could also adress the SDG.7 (energy) by describing the link between biofuel and the energy sector (Bauer et al. 2018).

Adding biodiversity feedbacks to the AFOLU sector

The next step in understanding the link between biodiversity and the AFOLU sector is to integrate feedbacks from biodiversity to agriculture and forestry (Foley et al. 2005). Biodiversity losses cause the reduction of several ecosystem services such as pollination, pest control, nutrient cycling and erosion control (Cardinale et al. 2012), that impact agriculture and forestry production.

Another possibility is to represent the link between biodiversity and the production of ecosystem services in the forest sector (Morin et al. 2014) as well as in the agricultural sector (Lafuite et al. 2018). The degradation of this link will therefore leads to a decrease in the services currently provided. Thus, Newbold et al. (2016) estimated that the degradation of biodiversity through current human pressure on land-use threatens 71.4% of the world's population.

Limits of the use of this modelling framework

The quantification of the sustainability of land-use based mitigation strategies led to the identification of issues that cannot be addressed in the modeling framework used in this thesis.

The first limitation of the modelling framework used is the lack of consideration of the dynamic aspects of biodiversity degradation, climate change and land-use change. The decrease in biodiversity not only refers to a decrease in species, genes, traits, ecosystems but also an ecological debt[1] (Tilman et al. 1994). An intertemporal approach is then necessary to take into account intergenerational equity and the sustainability of environmental protection measures (Lafuite & Loreau 2017). Another dynamic aspect of land-use impacts on ecosystems is the influence of rent on investment. In a case of land access restrictions, increase in land price leads to an increase in land rent. This supplementary rent is reinvested by land-owners in intensification (Koch et al. 2019). Taking into account the dynamic of land rent could inform where intensification occurs. The increase in agricultural rent also indirectly affects deforestation by being responsible for one-third of the amazon deforestation between 2002 and 2011 (Richards et al. 2014). Finally, the disconnection between consumption and remote production leads to a delay in the perception of the impacts of consumption modes. As presented by Beckage et al. (2018), the perception of the impact of climate change leads to a change in behavior. In this case, consumers can shift towards more sustainable diets. The timing of these changes would allow us to quantify the incentive that must be presented to consumers to influence these choices (Gitz et al. 2006). The last dynamic consideration to further develop concerns the optimal level of mitigation with uncertain future damages due to climate change. It is represented in integrated models such as the Dynamic Integrated model of Climate and the Economy (DICE) (Nordhaus 1992), or the RESPONSE model (Espagne et al. 2012). Combining these different dynamics would improve understanding of transitions to more sustainable human-nature systems and would remove the obstacles to integrated models for integrating biodiversity into environmental policies, investing in an agro-ecological transition and better account for changes in food consumption.

A second limitation of the modelling framework is the lack of integration of different scales. The biodiversity assessment changes through different spatial scales (Chase et al. 2018). In particular, landscape design effects on species increase biodiversity at the landscape scale while intensifying at the plot scale (Pereira & Daily 2006). The consideration of these landscape effects can be taken into account in global biodiversity models in cSAR-iDiv (Martins & Pereira 2017) and cSAR-IIASA-ETH (Chaudhary et al. 2015) models. The representation of land uses at these scales requires the representation of different land-use allocation processes. For this purpose, land-use models at a local scale (Verburg et al. 2002) can be combined with global-scale land-use models as was done in the IMAGE integrated modelling framework (Bouwman et al. 2006). The challenge is then to describe the relationships between the different scales when coupling these models (Verburg & Overmars 2009). This consideration of scale effects also improves the estimation of biofuel emissions associated with transport and land-use changes associated with their introduction (Daioglou et al. 2017).

1. Ecological debt is the extinction of species with a temporal delay following a modification of trophic networks in the ecosystem

Annexes

3.9 Tableaux annexes

Table 3.15 – Comparison of land-use models

	NLU	AROPAJ	GCAM	CLUE-S	Econometrics models
Reference	Souty et al. (2012) and Brunelle et al. (2015)	Jayet et al. (2018)	Calvin et al. (2019)	Verburg et al. (2002)	Chakir & Le Gallo (2013)
Objectives/Uses	Biofuel production, Mitigation strategies, Climate scenarios, Diet change	CAP, environmental taxes	Linkages between energy, water, land, climate, and economic systems	Local, regional assessment	Climate scenarios, Prediction of land-uses in France
Theory	Micro-economic,Integrated	Micro-economic,Integrated	Macro-economic,Integrated	Geography	Spatial Econometric
Scope	World	Europe	World	Regional	France/Europe
Level of description	12 regions	130 european regions	384 regions	Grid cell	Grid cell
Sector	Forest, Livestock, crop	Livestock, crop	All economy	Forest,Livestock,crop	Forest, Agriculture
Temporal dynamics	Static	Static	Dynamic recursive	Static	Static
Type of modelling	Linear Programming	Mixed Linear programming	CGE	Statistic	Spatial econometrics

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ÉCOLE DOCTORALE
Agriculture, alimentation, biologie, environnement, santé (ABIES)

université PARIS-SACLAY

Titre : Évaluation quantitative de la durabilité de stratégie d'atténuation des émissions de gaz à effet de serre dans le secteur AFOLU à l'échelle mondiale

Mots clés : Biodiversité,Durabilité,AFOLU,Usage des sols,Alimentation,Limites planétaires

Résumé : L'implémentation à large échelle de stratégies d'atténuation des émissions dans le secteur de l'agriculture, la forêt et autres usages des sols (AFOLU) pose des questions sur la durabilité de ces stratégies. Par exemple, les bio-fuels de seconde génération menacent la biodiversité et la reforestation d'espaces agricoles augmente le prix de l'alimentation. De plus, ces stratégies d'atténuation des émissions dépendent fortement des conditions socio-économiques décrivant le reste du système alimentaire (libéralisation du commerce agricole, développement économique, augmentation de la population...). Dans cette thèse, nous cherchons à préciser les impacts sur la biodiversité, l'alimentation et les émissions de gaz à effet de serre de différentes stratégies d'atténuation à large échelle dans le secteur AFOLU au regard de différentes situations socio-économiques. Pour cela, nous utilisons la modélisation prospective qui nous permet de simuler des scénarios décrivant l'évolution de l'usage des sols à l'échelle mondiale à l'horizon 2030, 2050 et 2100. Le couplage du modèle d'usage des sols Nexus Land-Use (NLU) avec le modèle de biodiversité Projecting Responses of Ecological Diversity In Changing Terrestrial Systems (PREDICTS) permet d'étudier l'impact de ces stratégies d'atténuation sur différentes composantes de la biodiversité. Le calcul de bilan d'azote permet quant à lui de préciser le lien entre l'intensification et sont impact environnemental.
Dans la première partie du manuscrit de thèse, nous testons des scénarios d'augmentation de la production de légumineuses en Europe en évaluant les effets sur les émissions de gaz à effet de serre du secteur AFOLU.
Dans la seconde partie, nous étudions les compromis et les synergies entre conservation de la biodiversité et maintien de la sécurité alimentaire pour différents scénarios d'atténuation.
Dans la troisième partie, nous comparons différentes évolutions de l'usage des sols à l'échelle mondiale en identifiant les scénarios qui permettent de ne pas dépasser les limites de la planète au regard d'indicateurs renseignant le cycle de l'azote, l'intégrité de la biosphère, les émissions de CO2 du secteur AFOLU et la conservation des forêts.

Title : Quantitative assessment of the sustainability of greenhouse gas mitigation strategies in the AFOLU sector at the global scale.

Keywords : Biodiversity,Sustainability,AFOLU,Land-use,Food,Planetary boundaries

Abstract : The large-scale implementation of emission reduction strategies in the agriculture, forestry and other land uses (AFOLU) sector raises questions about their sustainability. For example, second-generation bio-fuels threaten biodiversity and the reforestation of agricultural land increases food prices. In addition, these emission reduction strategies are highly dependent on socio-economic conditions describing the rest of the food system (agricultural trade liberalization, economic development, population growth, etc.). For example, an increase in food demand, due to population growth and economic development, can increase pressures on the food system, leading to ecosystem degradation and increased greenhouse gas emissions.
In this thesis, we seek to clarify the impacts on biodiversity, food and greenhouse gas emission of large-scale mitigation strategies in the AFOLU sector under different socio-economic conditions. To do this, we used prospective modeling to simulate various global land uses in 2030, 2050 and 2100 under different scenarios. More specifically, to study the impact of different mitigation strategies on biodiversity indicators, we coupled the Nexus Land-Use (NLU) model with the Projecting Responses of Ecological Diversity In Changing Terrestrial Systems (PREDICTS) biodiversity model. A nitrogen balance is also built to specify the link between intensification and environmental impact.
In the first chapter, we assessed the impact of scenarios of increased legume production in Europe on greenhouse gas emissions in the AFOLU sector.
In the second chapter, we analyzed the trade-offs and synergies between biodiversity and food security for different combinations of mitigation scenarios.
In the third chapter, we identified global land-use scenarios that ensure to stay within planetary boundaries in terms of nitrogen cycle, biosphere integrity, non-CO2 emissions from the AFOLU sector and forest conservation.

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