Régulation des pollutions azotées d'origine agricole

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Ecole Doctorale ABIES

Doctorat ParisTech

THÈSE

pour obtenir le grade de docteur délivré par

L'Institut des Sciences et Industries du Vivant et de l'Environnement

(AgroParisTech)

Spécialité : Sciences économiques

présentée et soutenue publiquement par

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le 10 avril 2012

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Résumé

Les travaux présentés dans cette thèse ont eu pour double objectif de définir les principaux déterminants permettant une politique optimale de régulation des pollutions azotées, puis, à l'aide de modèles appliqués, de quantifier et d'évaluer les impacts de ces politiques. Nous avons d'abord défini un cadre permettant de mieux comprendre comment la réallocation des terres induite par une politique influe sur le niveau des pollutions azotées. Une solution fondée sur une taxation des intrants couplée à une subvention en faveur des cultures pérennes paraît alors appropriée. L'évaluation de cette politique permet de démontrer qu'elle pourrait être coût-efficace pour les pollutions dues aux nitrates. Nous avons également approfondi l'étude de la régulation de la pollution des aquifères due aux nitrates. Ce type de pollution suggère des traitements spécifiques au problème posé par la forte inertie du milieu physique considéré en raison du temps de latence entre l'application d'engrais et la pollution de l'aquifère. Ce délai a un impact important sur la caractérisation d'une politique optimale. Outre le stock de pollution optimale de long terme qui augmente avec le délai, au cours du temps un sentier de taxe plus élevée est nécessaire pour atteindre ce stock. La valeur du dommage marginale est ensuite estimée dans le cas d' un aquifère du bassin de la seine.

Abstract

The work presented in this thesis had two purposes : to identify the key determinants to regulate the nitrogen pollution, and to test the ability of applied models to assess the impacts of the regulation policies. We first define a framework for understanding the role of land-use in the optimal management of nitrogen pollution. Regulation design based on an input taxation coupled to a subsidy for perennial crops seems appropriate. This policy is then evaluated and we show that it could be cost-effective in the case of nitrate pollution. We also developed the study of aquifer pollution from nitrates. This pollution problem leads to develop taking into account of the high inertia of the physical environment considered in reason of the delay between the application of fertilizer and pollution of the aquifer. This delay has a significant impact on optimal policy. Besides the pollution stock optimal long-term increases over time, a path of stronger tax is necessary to achieve the steady-state. The marginal damage value is then estimated for an aquifer associated to the Seine river basin.

Remerciements

Mes remerciements vont tout d'abord à Pierre-Alain Jayet, pour l'honneur qu'il m'a fait en acceptant d'encadrer ces travaux de thèse. Je le remercie pour sa disponibilité et la liberté qu'il m'a laissé pour mener mes recherches. J'espère que nous continuerons à travailler ensemble encore longtemps.

Cette thèse est également redevable à la Région Ile-de-France à travers l'allocation doctorale financée par le projet EcoSeine dans le cadre du DIM R2DS. Elle est également largement redevable au programme PIREN-Seine, qui en plus de son soutien financier, a permis des échanges avec le centre Géosciences de l'école des Mines et le laboratoire Sisyphe de l'Université Paris VI.

Je remercie Katheline Schubert, Mabel Tidball, Florence Habets et Werner Hediger d'avoir accepté de participer au jury de cette thèse. J'espère que ce travail est digne de leur intérêt.

Je remercie l'équipe de Géosciences et l'UMR Sysyphe, notamment Florence Habets et Pascal Viennot, pour leur collaboration et leur disponibilité, en nous permettent d'utiliser leur modèle hydrologique, MODCOU.

Je dois également beaucoup à Vincent Martinet qui, au delà de notre collaboration, m'a transmis son expérience de chercheur et à Stéphane De Cara pour les réflexions, les questions et les conseils qu'il m'a prodigués.

Je tiens particulièrement à remercier toutes les personnes qui par leur nombreuses discussions ou relectures ont largement contribué à rendre cette thèse meilleure. Je pense en particulier à Stéphane De Cara, Maïa David, Vincent Martinet, Suzette Tanis-Plant ainsi que Pierre Humblot, David Leclère et Bruno Vermont.

Je remercie également l'équipe de L'UMR Economie Publique, ses doctorants, chercheurs et enseignants, et les différents membres du personnel administratif et technique qui s'y sont succédés. Mes pensées vont particulièrement à Nosra et David qui soutiendront bientôt à leur tour, et Bruno avec qui la cohabitation dans les dix mètres carrés de notre bureau a été agrémentée de francs moments de rigolade.

Je retiendrai également, l'ambiance inoubliable qui a régné dans ce laboratoire durant ces trois années, ainsi que les pauses cafés donnant naissance à de mémorables discussions, plus ou moins métaphysiques, et propices à de nombreuses péripéties et catastrophes !

Enfin, je tiens également à remercier parents et amis pour leur soutien durant ces trois ans

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Introduction générale

L'agriculture en France depuis 1945.

Depuis 1945, le monde agricole en France a vu la diminution considérable de la petite paysannerie de subsistance et l'avènement des fermes modernes gérées par les paysans-agriculteurs, cultivateurs, éleveurs que l'administration française nomme les exploitants agricoles. En effet, de 10 millions d'actifs agricoles en 1945, la France ne compte, aujourd'hui, guère plus d'un million d'exploitants agricoles représentant à peine 4% de la population active. Cette diminution est directement liée à la révolution verte et à la mécanisation, à l'emploi d'engrais de synthèse dans l'agriculture, qui ont conduit à une hausse importante de la productivité et à une concentration des terres au sein de quelques grandes fermes d'exploitations agricoles. Cet effort de concentration des terres après la Seconde Guerre mondiale est très encadré par la législation et a été soutenu par une volonté politique d'aménagement du territoire. Cette concentration est cependant très inégale géographiquement : bien adapté au bassin parisien et aux grandes plaines céréalières,elle est quasiment absente des régions d'arboriculture fruitière comme dans le Sud de la France. Parallèlement à la concentration des terres agricoles, la taille des exploitations a augmenté. 60 % des exploitations françaises de moins de 20 hectares ont disparu entre 1967 et 1997, tandis que le nombre de celles de plus de 50 hectares a quasiment doublé. À partir des années 1990, la surproduction de l'agriculture européenne faisant place à d'autres préoccupations que la productivité : écologiques, environnementales, paysagères. Cependant la concentration des exploitations agricoles continue, celles-ci passant de 1 016 755 au recensement agricole de 1988 à 663 807 au recensement agricole de 2000. La taille moyenne des exploitations est passée de 15,8 ha en 1963, à 39 ha en 1995 jusqu'à atteindre 73.3 ha en 2003. La concentration économique accompagne ce processus. En 1997, 10 % des exploitations européennes réalisaient plus de 65 % des revenus agricoles, les 50 % plus petites n'en réalisant que 5 %.[1] Cette concentration des terres a été aussi accélérée par une volonté d'atteindre l'autosuffisance alimentaire, qui est atteinte dès les années 1970. Ces facteurs combinés à une surface agricole utile importante (environ 1/2 ha par habitant), à une situation géographique et climatique favorable et à l'aide apportée par la Politique Agricole Commune (PAC) expliquent que la France soit devenue le premier pays agricole de l'Union européenne avec 18 % du produit agricole et agro-alimentaire européen. Les principales productions sont les céréales (blé, 1er rang européen et 5e mondial ; maïs, 7e mondial), la betterave à sucre (1er mondial), le vin (1er mondial), le lait (3e mondial) et les produits laitiers, les fruits et légumes, l'élevage (notamment en Bretagne) et les produits carnés (5e mondial pour la viande bovine).[2] Le poids de l'activité agricole (y compris le secteur des industries agro-alimentaires) représente 3,5 % du PIB (2008), soit 66,8 milliards d'euros. L'agriculture tient ainsi une place prépondérante dans la balance commerciale : en 2002, son excédent commercial avoisinait les 9 milliards d'euros, presque autant que les secteurs automobiles et aéronautiques - respectivement 11 et 12 milliards d'euros pour cette même

1. Bimagri HS n°23, janvier 2010
2. 2009, source FAO

année. L'agriculture occupe, en 2006, un peu plus de 32 millions d'hectares sur les 55 millions d'hectares du territoire français métropolitain et employait, en 2007, 3,4 % de la population active totale.

Les problèmes environnementaux. L'agriculture intensive a demandé aux paysans une remise en cause complète de leur mode de vie, afin de produire en quantité et à bas prix. La nouvelle norme de production se caractérise par la mécanisation systématique, l'apport massif d'intrants (engrais et produits phytosanitaires), la France est ainsi devenue le second consommateur mondial de pesticides derrière les USA et le premier consommateur européen d'engrais [3]. Cependant, les liens entre les apports d'engrais, de phosphates et de pesticides d'une part et la pollution de l'autre sont multiples et complexes car faisant intervenir plusieurs polluants et les interactions avec le milieu sont nombreuses.

Les substances nutritives telles que l'azote et le phosphore sont prélevées du sol par les plantes pour leur croissance et ont besoin d'être remplacées. Dans le cas contraire, les sols risquent de s'appauvrir, entrainant une baisse de rendements des cultures. Par le passé, la rotation des cultures et les périodes régulières de jachère ainsi que l'épandage de fumier animal permettaient à la terre de retrouver une partie de sa fertilité. Aujourd'hui, la principale méthode pour restituer au sol ses substances nutritives et accroître les rendements des cultures consiste à appliquer des engrais minéraux, qui sont ainsi devenus la principale source d'azote dans les pays de l'Union Européenne. Celui-ci, particulièrement soluble, facilite son assimilation par les plantes. Les apports fournis par les

3. 2007, source FAO

effluents d'élevage restent néanmoins importants, en particulier dans les régions à forte densité de bétail comme la Bretagne.

Les substances azotées en excédent, c'est à dire l'écart entre les apports et les consommations par les cultures, peuvent constituer une menace pour l'environnement, entraînant une pollution de l'eau, de l'air et du sol. On peut ainsi distinguer les pertes dans l'air sous forme d'ammoniac (NH_3) et de méthane (CH_4) par volatilisation directe après épandage des effluents d'élevage dans les champs, celle dans l'air sous forme de protoxyde d'azote par dénitrification (N_2O), et celles dans les eaux souterraines par lessivage des nitrates (NO_3), sous forme de composés organiques, et dans les eaux de surface, rivières et les lacs, par ruissellement après de fortes précipitations. Les relations entre le bilan des substances nutritives et les pertes diffèrent selon les systèmes agricoles. Ces relations sont affectées par l'intensité de l'utilisation des sols, par les pratiques de gestion des exploitations, par le type de sol et les conditions climatiques. A cela s'ajoute d'autres polluants liés à l'activité agricole comme le phosphore, principalement présent sous forme de phosphate. Celui-ci, moins soluble que l'azote, est en grande partie acheminé par les sédiments dans les eaux d'écoulement et se retrouve fréquemment dans les rivières et les ruisseaux.

Ainsi, l'agriculture est un émetteur prépondérant de N_2O et de CH_4 avec, en 2008, respectivement 84,9 % et 79,1 % des émissions nationales ce qui place ce secteur au premier rang pour ces deux polluants. L'agriculture (fermentation entérique, gestion des déjections et les sols) n'émet pas directement de CO_2 cependant sa contribution au PRG (pouvoir de réchauffement global) atteint 18,5% en 2008. La part de l'agriculture pour le CH_4 et le N_2O est en augmentation depuis 1990, bien que les émissions en masse soient en baisse, ce qui s'explique par la chute des émissions des autres secteurs (le CH_4 des décharges, d'une part, et le N_2O de la chimie, d'autre part). Par contre en 2008 sa contribution aux émissions de Gaz à Effet de Serre (GES) a diminué par rapport à 1990 en raison de l'augmentation des émissions de CO2 lié à la consommation d'énergie, en particulier sur la même période. Les pratiques agricoles contribuent, également, essentiellement aux émissions de NH_3, dont elles sont responsables à 95% même si elle dépendent également d'autres facteurs comme le type d'engrais utilisé, le type de sol (notamment de son pH), les conditions météorologiques et le moment d'application par rapport à la couverture végétale. En France, la volatilisation des déjections animales constitue la principale source agricole d'émission d'ammoniac (plus de 80%). 10% à 20% des émissions de NH_3 proviennent de la volatilisation de l'ammoniac émanant des engrais azotés et des cultures fertilisées. Les émissions dues aux "cultures non fertilisées" sont négligeables. L'acidification, due au NH_3, réduit considérablement la fertilité des sols, essentiellement en affectant leur biologie, en décomposant les matières organiques et en provoquant la perte de substances nutritives. C'est aussi une des principales causes de pluies acides. De plus, l'acidification des sols est un facteur déterminant de la libération de cations tels que le fer, l'aluminium, le calcium, le magnésium ou les métaux lourds (qui sont présents dans le sol en quantités significatives, mais qui sont généralement très peu mobiles). Cela a pour effet de réduire le pouvoir tampon des sols (par la décomposition des minéraux argileux) et de modifier leur capacité à neutraliser l'acidité. Ce phénomène se produit notamment sur les sols dotés d'un faible pouvoir tampon et constitue un problème grave car irréversible. Enfin, l'acidification des sols est étroitement liée à l'acidification de l'eau, qui peut affecter la vie aquatique, les eaux souterraines et l'approvisionnement en eau potable qui y est lié. Le total résiduel des contaminations azotées, dues aux nitrates (NO_3), des ressources et milieux aquatiques et marins est ainsi estimé à environ 806 000 tonnes dont environ 715 000 tonnes proviennent de l'agriculture et de l'élevage, soit 88,7%. Plus précisément, pour 2008, le dernier rapport du Service de l'Observation et des Statistiques (SOeS 18) sur l'état des eaux établit que 60% des eaux de surface se situait au dessus du seuil de bonne qualité de 10 mg/l. Au niveau des eaux souterraines, la lente détérioration par rapport aux nitrates semble se poursuivre au niveau national depuis les années soixante. De façon générale, sur les dix dernières années, on observe des pourcentages de moins en moins importants de points d'eau avec des teneurs inférieures à 10 mg/l (qui passent de 56 à 48 %) et une augmentation de l'ordre de 50 % du nombre de ceux dont la concentration en nitrates dépasse 50 mg/l (qui passe de 4 à 6%). Les concentrations excessives d'azote, sous forme de NO_3, et de phosphore dans l'eau peuvent entraîner une eutrophisation des rivières à débit lent, des lacs, des réservoirs et des zones côtières. Ce phénomène se manifeste par une prolifération d'algues verte, une moindre infiltration de la lumière, la raréfaction de l'oxygène dans les eaux de surface, la disparition des invertébrés benthiques [4] et la production de toxines nuisibles aux poissons, au bétail et aux humains. Les eaux souterraines contaminées ne sont ainsi plus potables. Les sols sont également exposés au risque d'eutrophisation lorsque la quantité excessive de substances nutritives entraîne la raréfaction de l'oxygène dans le sol et empêche donc les micro-organismes naturels de fonctionner correctement. Ceci affecte alors la fertilité des sols. Les sols eutrophisés sont, de plus, à l'origine d'émission de N_2O.

4. Organismes vivant dans les milieux aquatiques à la surface ou à l'intérieur des substrats immergés (sédiments, végétaux, etc.). L ?étude des communautés d ?invertébrés benthiques (annélides, insectes, crustacés, mollusques, etc.), qui sont composées d'espèces qui présentent un gradient de sensibilité aux pollutions (espèces sensibles, indifférentes ou tolérantes) permet une évaluation qualitative et quantitative de la pollution des milieux aquatiques

On estime que l'agriculture utilise 95 % des pesticides vendus en France, qui en est le premier utilisateur. Cependant, contrairement à l'azote, on ne connaît pas le bilan distinguant ce qui est fixé dans les productions végétales et le sol et les excédents qui repartent vers les milieux aquatiques. L'enquête de la Direction Générale de la Santé (DGS) de 2005 révélait que 20,6 % des points de captages d'eau potable fournissant 46,7 % de l'eau brute prélevée en 2002 concernaient des eaux de qualité médiocre ou mauvaise vis à vis des pesticides, ce qui nécessite un traitement ou des mesures de potabilisation.

Directives Pour répondre à ces divers problèmes environnementaux, de nombreuses politiques, contraignantes ou non, sont apparues. Cependant, si les diverses pollutions ont pour source les engrais azotés, ces politiques ne sont pas coordonnées et ciblent les polluants un par un et ce sans prendre en compte les différentes interactions existantes.

Plus précisément, concernant le N_2O et le CH_4, si le secteur agricole n'est à ce jour pas concerné par le marché de permis européen (European Emission trading System, ETS 2003), en 2009, la commission européenne s'est fixée comme objectif de réduire les émissions européennes dues aux secteurs non-couverts par l'ETS [5] de 10% en équivalent CO_2 en 2020 par rapport au niveau de 2005. Pour la France, cela se traduit par un objectif de réduction de 3% en équivalent CO_2.

Le NO_3 est lui principalement concerné par la politique communautaire de l'eau. Elle s'appuie aujourd'hui sur une vingtaine de directives européennes. Les plus anciennes ont fixé des obligations de résultats à visée sanitaire : valeurs limites et normes de concentration définissant la qualité des eaux des-

5. Les secteurs non couverts par l'ETS sont le transport, le logement et l'agriculture. Ce dernier représente 25%, en équivalent CO_2 des émissions non-ETS

tinées à la consommation humaine, des eaux souterraines, conchylicoles, de baignade, etc... La seconde génération fixait des obligations de moyens visant la protection des ressources et milieux aquatiques : assainissement urbain et industriel (DERU, 1991), nitrates (1991). La troisième génération, datant des années 2000, vise la restauration du bon état écologique de l'eau et des milieux marins : directives cadres sur l'eau (DCE, 2000) et les milieux marins (DCSMM, 2008). Ces directives laissent chaque état membre (EM) définir de manière participative son état des lieux initial et son bon état futur dans un calendrier commun. Les dérogations et reports de délais pour l'atteinte de ce bon état doivent être annoncés et justifiés par des analyses socio-économiques (coûts disproportionnés). De plus, les programmes de mesure pour passer de l'état initial au bon état doivent être publiés et financés dans le calendrier des directives. La France est ainsi en contentieux pour le non-respect de certaines directives de 1ère génération et le retard à la mise en oeuvre de la DERU (à ce sujet, on peut également citer le rapport de la Cour des comptes, cf Encadré 1). Les EM ont défini leur bon état écologique de l'eau en 2004 et avaient jusqu'à fin 2009 pour communiquer leurs programmes de mesure dans le cadre de laDCE. Ils avaient jusqu'à juillet 2010 pour désigner les autorités compétentes sur la mise en œuvre de la Directive-Cadre Stratégie pour le Milieu Marin (DCSMM du 17 juin 2008) et ont jusqu'à mi 2012 pour définir le bon état écologique de leurs milieux marins et 2014 pour communiquer leurs programmes de mesure. En 2007, le Grenelle sur l'agriculture a ciblé plusieurs mesures sur les pratiques agricoles (restrictions des pesticides, accroissement de la SAU en agriculture biologique, bandes enherbées, maintien du couvert végétal), la biodiversité (zones humides, trames vertes et bleues) et l'eau (protection des aires d'alimentation des captages).

Encadré 1 :Rapport de la Cour des Comptes
" Sur le plan qualitatif, l'activité humaine, industrielle et agricole est à l'origine de pollutions principalement organiques, chimiques (fertilisants, pesticides, métaux etc...) et biologiques (bactéries, virus, etc.) qui finissent par rejoindre les milieux aquatiques. Ces pollutions peuvent être ponctuelles (exemples : rejets domestiques ou industriels, effluents d'élevage...) ou diffuses (ex : épandages de pesticides et d'engrais). Si les premières commencent à être correctement traitées, il n'en va pas de même des secondes. Pour mettre en place la directive-cadre sur l'eau, des états des lieux par bassin ont été réalisés fin 2004 par les agences de l'eau. Pour les cours d'eau, ces bilans tendent à montrer que la pollution par les matières organiques et phosphorées, issues des rejets urbains et industriels, a sensiblement diminué depuis une dizaine d'années, grâce aux investissements réalisés par les collectivités locales et les entreprises, mais qu'elle atteint aujourd'hui un palier. La pollution due aux nitrates, majoritairement d'origine agricole et dépendante des conditions climatiques, reste en revanche élevée en moyenne. Les baisses dans les bassins les plus touchés sont compensées par des hausses ailleurs, contribuant sur certains littoraux aux phénomènes de "marées vertes" . 65 % des masses d'eau superficielles et 61 % des masses d'eau souterraines présentaient un risque (avéré ou potentiel) de ne pas atteindre le bon état en 2015 et les pesticides étaient présents dans les deux tiers des eaux souterraines. Cette situation conduit la cour à "douter de la capacité de la France d'atteindre dès 2015 les objectifs de qualité qu'elle s'est assignée, sauf à ce que des améliorations y soient rapidement apportées. En toute hypothèse, l'enjeu financier est très élevé puisque le respect de cette échéance aura un coût qui a pu être estimé à 24,7 milliards d'euros pour les actions recensées dans les programmes de mesures 2010-2015"

Problématiques En lien avec le thème "modélisation socio-économique et scénarisation prospective, agriculture du bassin de la Seine" inséré dans le volet "territoire Seine dans le changement global" du programme interdisciplinaire de recherche sur l'environnement appliqué au bassin de la Seine ("PIREN-Seine") dans sa phase V (2007-2011), la thèse a pour objectif premier l'étude de la régulation économique des pollutions azotées d'origine agricoles (N_2O, NH_3, NO_3) et son impact sur l'utilisation des terres agricoles.

D'un point de vue économique, ces pollutions diffuses posent un double problème : si elle viennent toutes de l'utilisation d'un même input, les engrais azotés, le dommage marginal associé à chacun de ces polluants est différent rendant difficile une régulation intégrée. D'autre part, les fonctions d'émission de chaque polluant sont fortement hétérogènes suivant la culture et le type de sols considérés. Ainsi, trouver les instruments économiques permettant de réguler chaque polluant de façon coût-efficace est relativement complexe. Cette difficulté s'accroît également par le fait qu'un instrument coût-efficace pour un type de polluant ne l'est pas forcément pour d'autres. De plus, le milieu considéré joue un rôle important. Pour la pollution par les nitrates, par exemple, ce qui définit la régulation optimale pour les eaux de surfaces est différent de ce qui la définit pour les eaux souterraines : en effet, la première est une pollution instantanée alors que la seconde est retardée. Dans le cas de la pollution des eaux souterraines, 10 à 60 ans peuvent s'écouler entre l'application d'engrais et leur arrivés sous forme de nitrates dans les aquifères.

Dans ce contexte, nous nous sommes essentiellement focalisés sur deux points : (i) montrer que l'utilisation des terres est une variable essentielle si l'on veut définir une politique optimale de régulation des pollutions azotées et (ii) définir une politique optimale de régulation de la pollution des nitrates dans les aquifères.

La thèse est organisée en deux parties. Dans la première partie, nous présentons une revue de la littérature ainsi que les principales réponses apportées aux problèmes de pollution diffuse. Puis nous étudions la question de la régulation optimale des pollutions diffuses suivant deux approches différentes, respectivement statique et dynamique, et dans deux cadres d'études différents, respectivement l'impact du changement d'allocation des terres et la pollution des eaux souterraines. Dans la deuxième partie, avec l'aide de modèles appliqués, nous avons évalué ces politiques. Plus précisément, le chapitre 2 montre dans un modèle d'utilisation des terres qu'une politique de second rang comme la taxation des intrants peut amener un effet paradoxal comme l'augmentation de la pollution. Cet effet est dû à la structure des émissions et aux changements d'allocation des terres nécessitant alors la mise en place d'un second instrument comme une subvention aux cultures non polluantes. Ensuite, dans le chapitre 3, nous nous intéressons, à travers un modèle de contrôle optimal avec asymétrie d'information, à la question de la régulation optimale d'un stock de pollution quand celle-ci a lieu plusieurs années après l'émission de polluants.

La partie appliquée suit le même schéma. Après une description des modèles utilisés dans le chapitre 4, dans le chapitre 5 nous étudions l'effet sur les pollutions azotées d'une taxe sur les fertilisants minéraux couplée à une subvention sur les cultures pérennes. Nous proposons des résultats pour la France et par bassin versant. Enfin, dans le chapitre 6, nous faisons le lien entre modèle théorique et modélisation appliquée (ici un modèle agro-économique couplé à un modèle hydrologique) pour déterminer la valeur sociale de la contamination des aquifères par les nitrates . Nous nous servons aussi de cette valeur, afin de déterminer les pertes de bien-être induit par des politiques plus accommodantes.

Première partie

Détermimants et régulations des pollutions diffuses d'origine agricole

Chapitre 1

Revue de littérature

La littérature sur le contrôle de la pollution porte essentiellement sur les performances des instruments basés directement sur les émissions (Russell and Powell, 1999). La principale conclusion est que les émissions sont la meilleure base d'application pour des incitations économiques quand la source des pollutions et les émissions peuvent être, d'une part identifiées à un degré de précision et à des coûts raisonnables et, d'autre part, ne sont pas significativement stochastiques (Beavis and Walker, 1983; Oates, 1995). Dans un modèle néo-classique statique, cela revient par exemple à l'instauration d'une taxe pigouvienne.

Cependant, si ces conditions sont relativement satisfaites dans le cadre de pollutions non-diffuses et plutôt industrielles, elles s'accommodent assez mal avec les pollutions d'origine agricole, principalement diffuses et pouvant avoir un caractère stochastique car dépendantes des conditions météorologiques. Dans le cadre spécifique de régulation des pollutions diffuses, on peut distinguer quatre approches : une portant sur le contrôle d'un "proxy" des émissions, une portant sur la régulation de la concentration ambiante, une portant sur les marchés de permis et enfin, celle portant sur la régulation des inputs.

La première approche a été initiée par Griffin and Bromley (1982). Leur modèle repose sur l'hypothèse que les émissions diffuses sont une fonction non-stochastique des inputs, continument différentiable, celles-ci pouvant alors être mesurées avec précision. Ils supposent que le profit pouvant provenir d'activités alternatives ou substituts est connu. Sous ces hypothèses, ils montrent qu'une taxe ou une norme permet d'atteindre l'optimum de premier rang. En reprenant ce modèle, Shortle and Dunn (1986) introduisent des éléments stochastiques dans la fonction d'émission. Ils soutiennent alors que si les incitations économiques comme une taxe ou une norme ne permettent pas d'atteindre l'optimum de premier rang, en présence d'asymétrie d'information la première solution est néanmoins préférable car elle permet aux agriculteurs d'utiliser pleinement leur expérience et leur connaissance des relations entre les pratiques agricoles et les pertes. Ils mettent également en évidence le rôle joué par la fonction de dommage, dans le cas où les émissions ont un caractère stochastique. La régulation se fait, alors, sur la base des émissions moyennes. Or la solution qui réduit de façon coût-efficace la moyenne des émissions ne réduit pas obligatoirement de façon coût-efficace la moyenne des dommages si ceux-ci ne sont pas linéaires. Horan et al. (1998) montrent que seul le cas où les émissions proviennent d'un seul input ou lorsque la covariance entre les émissions marginales et le dommage marginal est nulle permet de réduire coût-efficacement aussi bien les émissions moyennes que le dommage espéré.

La seconde approche est initiée peu après par Segerson (1988). L'agence en charge du contrôle de la pollution se base sur la concentration ambiante du milieu considéré. L'idée sous-jacente et le principal avantage de cette méthode consistent dans le fait que la pollution ambiante peut-être mesurée avec une meilleure précision et à un coût inférieur. Le système d'incitation est basé sur le principe d'une taxe (respectivement subvention) pour tous les producteurs lorsque la concentration observée est supérieure (respectivement inférieure) à la cible fixée au préalable par l'agence en charge de la régulation. La mise en place d'une taxe ambiante voulant satisfaire un objectif précis nécessite néanmoins beaucoup d'information sur les pollueurs. Pour répondre à cette critique, Segerson and Wu (2006) combinent une approche volontaire de réduction des émissions avec une menace de taxe ambiante si cet objectif n'est pas atteint. Ils montrent que cette approche permet de minimiser les coûts de réduction sans avoir besoin, par ailleurs, de connaître les caractéristiques spécifiques des pollueurs. Cette politique est donc à la fois plus efficace qu'une approche volontaire de réduction sans menace de taxe ambiante et implique moins d'information et des coûts de transaction plus faibles (la taxe étant une menace, elle n'est pas mise en œuvre la plupart du temps).

Bien que ce soit un instrument pouvant être facilement mis en œuvre de façon uniforme, on peut lui trouver quelque limites : un problème évident d'aléa moral puisque les incitations dépendent du niveau de pollution ambiante plutôt que des émissions individuelles. Ainsi, un pollueur peut avoir intérêt à abattre moins qu'à l'optimum si ses actions ne sont pas contrôlées, son émission marginale supplémentaire lui octroyant plus de profits que la taxe liée au dommage marginal supplémentaire. On peut néanmoins objecter que la tendance à la concentration des terres agricoles, et donc le fait qu'un nombre de plus en plus réduit d'agriculteurs soit responsable de la pollution à un endroit donné, limite en partie cette problématique. Néanmoins, Xepapadeas (1991) résout ce problème d'aléa moral en proposant différents contrats que le régulateur pourrait mettre en place avec les pollueurs. Si la pollution ambiante est supérieure à un seuil prédéterminé, un pollueur est choisi de façon aléatoire par le régulateur et doit payer la taxe. Les autres pollueurs reçoivent une fraction de l'amende moins les dommages associés à la pollution. La menace de pénalité suffit à rendre les bénéfices liés à la pollution marginale supplémentaire moins intéressants que l'espérance des pertes associées (taxe due à la hausse de la concentration ambiante plus la probabilité d'être pénalisé). De surcroît, Herriges et al. (1994) montrent que l'espérance des coûts associée à la triche augmente considérablement si les pollueurs sont suffisamment risque-averses. Le succès de la régulation de la concentration peut être aussi affecté par l'incertitude du pollueur sur le comportement des pollueurs voisins. En particulier, si les pollueurs sont fortement hétérogènes, l'erreur que fait un agriculteur en estimant les pollutions des autres agriculteurs à partir de son propre comportement augmente et sa propre réponse à la régulation sera donc également sous-optimale(Cabe and Herriges, 1992). Outre le problème d'aléa moral, le régulateur fait aussi face à un problème d'équité. La taxe ou la subvention dépend du groupe et non des performances individuelles, ce qui peut ainsi limiter l'acceptabilité politique d'une telle mesure, puisque celui qui a fait le plus d'efforts d'abattement peut, éventuellement, se retrouver à devoir payer la taxe (Shortle et al., 1998). Spraggon (2004) souligne également que si l'on considère des agents hétérogènes alors les mécanismes de régulation des concentrations ambiantes par taxes/subventions amènent les firmes de petite taille à dépolluer moins que le niveau optimal et inversement. En prenant en compte un continuum plutôt que deux types de firme, Suter et al. (2009) confirment ces résultats. Poe et al. (2004) étudient les performances de ces mécanismes en prenant en compte la possibilité pour les firmes de coopérer. Ils montrent alors empiriquement que les résultats diffèrent sensiblement de ceux prévus par la théorie dans le cas de non-coopération, exceptés pour les scénarios de taxes/subventions ou de taxes seules. Les mécanismes de taxes/subventions avec pénalités fixes entraînent un sur-abattement et s'éloignent considérablement des résultats non-coopératifs. Cependant, ces mécanismes entraînent un fort déficit budgétaire et les subventions accordées surpassent de loin les bénéfices liés à la réduction de la pollution. En revanche, un mécanisme de taxes/subventions avec pénalité fixe dans un cadre coopératif conduit la taxe à atteindre la cible environnementale avec une fréquence plus élevée.

On peut également noter que si ce mécanisme s'avère, sous certaines conditions, approprié pour la pollution des eaux de surface, il est relativement inopérant pour la pollution des eaux souterraines. En effet, le laps de temps particulièrement élevé (plusieurs dizaines d'années) entre l'application de fertilisant et son arrivée dans l'aquifère perturbe la relation entre la concentration ambiante et les pratiques agricoles. De plus, la structure de l'activité agricole, comme par exemple, le nombre d'hectares par agriculteur, la conversion de terres agricoles à d'autres usages, et les agriculteurs concernés au moment de la régulation peut avoir considérablement changé.

Souvent associé au concept de taxe ambiante, on peut également évoquer le concept des règles de responsabilité (Segerson, 1990). Sous de telles règles, les pollueurs peuvent être poursuivis pour le montant total des dommages causés. Ces règles créent en effet des incitations similaires à celles causées par les mécanismes de taxes ambiantes dans le sens où la décision de poursuivre ou non les pollueurs est due à un niveau ou seuil de pollution constatée. Elles diffèrent néanmoins par quelques aspects. Elles ne sont imposées que si une procédure juridique est initiée par une partie publique ou privée. Un facteur d'incertitude est ainsi ajouté, relatif au fait de savoir si le pollueur sera ou non poursuivi avec succès pour les dommages causés. Par exemple, l'incapacité à identifier précisément les sources de pollution et de prouver la responsabilité du pollueur pourrait rendre la probabilité qu'il soit poursuivi et tenu responsable très faible dans les règles de responsabilité stricte (Braden and Segerson, 1993). Une règle de négligence dans laquelle la responsabilité est basée sur le respect des 'pratiques de gestion acceptées' peut être plus appropriée dans ce cas. Cependant, le principal inconvénient des règles de responsabilité ou de négligence, est le processus de contentieux en responsabilité qui peut être coûteux par rapport à d'autres méthodes de régulation. Ce coût peut décourager les individus de tenter de réclamer des dommages et conduire *in fine* à ce que les pollueurs ne soient plus soumis à régulation. (Shavell, 1997).

La troisième approche, les marchés de permis, a servi de base à l'élaboration de la législation américaine sur l'environnement (U.S EPA, 2001). Elle a été vue comme la meilleure façon de coordonner les efforts de régulation des pollutions diffuses et non-diffuses et la plus à même d'atteindre des objectifs environnementaux à des coûts moins élevés que les approches 'command and control' (Tietenberg, 1990). Cependant, les objections émises sur les deux mécanismes évoqués précédemment restent valables. En effet, en raison du caractères stochastique et non observable des émissions diffuses, le marché de permis ne peut être fondé sur un échange de ces émissions. Une alternative pourrait être l'introduction de permis sur les intrants (par exemples les engrais) corrélés aux émissions, principe de ' l'emission- for- inputs trading system' (EI). Hanley et al. (1997) proposent ainsi un système dans lequel les permis sur les pollutions non-diffuses pourraient être échangés avec des réductions de fertilisant ou une diminution des surfaces allouées à des cultures particulièrement intensives en engrais. Horan et al. (2004) étudient une politique basée sur des subventions à la réduction d'inputs couplée à un marché de permis. Les subventions peuvent être conditionnées ou non à une cible. Ils partent de l'hypothèse que les politiques administrées par différentes agences ne sont pas substituts mais plutôt complémentaires, c'est-à-dire qu'aucune de ces deux politiques ne peut individuellement conduire à l'optimum social. Il ressort que la coordination de ces deux politiques apporte des gains d'efficacité uniquement si la décision d'émettre la dernière unité de pollution est soumise à l'influence simultanée des deux politiques. Cela ne peut se produire que si les subventions sont subordonnées et calibrées de façon à atteindre une cible environnementale précise. Lankoski et al. (2008) développent un modèle de marché de permis avec hétérogénéité spatiale des émissions diffuses, en l'occurrence celles liées aux effluents d'élevage. Les permis portent sur des quotas visant notamment à laisser des terres agricoles en jachère, ce qui est vérifiable à moindre coût. Ils montrent que l'hétérogénéité spatiale diminue considérablement l'attractivité des marchés des permis. Alors que certains agriculteurs ont réalisé des gains substantiels, d'autres bénéficient de gains négligeables et enfin, la plupart subissent des pertes. Par ailleurs les gains des agriculteurs à échanger diminuent au cours du temps, et les pertes augmentent, si les allocations de permis récompensent les réductions de pollution passées. En prenant pour exemple la pollution d'un bassin versant (symbolisé par un plan d'eau approvisionné par une seule rivière), ils montrent que les permis devraient être ajustés à la fois pour les dommages marginaux à des endroits intermédiaires et au niveau du plan d'eau récepteur ainsi que pour la dégradation/rétention du polluant survenant entre le lieu d'émission et sa destination finale.

Enfin, la dernière approche, qui sera utilisée sous différentes formes dans les deux chapitres suivants, sont les instruments directement basés sur la régulation des inputs. Dans le cas d'inputs multiples, les agents concernés vont répondre aux changements de prix par une utilisation accrue des inputs devenus relativement moins coûteux (Shumway, 1995). Un gain de long terme est aussi montré quand on suppose que les agents concernés investissent dans des technologies destinées à accroître l'utilisation des inputs devenus relativement moins coûteux (Hayami and Ruttan, 1985). Griffin and Bromley (1982) montrent que taxer les inputs à l'origine d'une pollution diffuse et subventionner ceux qui la réduise permet d'atteindre l'optimum de premier rang. Les mêmes résultats sont obtenus par Stevens (1988) et Dinar et al. (1989). Shortle and Abler (1994) et Shortle et al. (1998) montrent de plus qu'une telle taxe est optimale, même en présence de fonctions d'émission stochastique et d'asymétrie d'information. Dans le cas où la pollution diffuse provient d'un seul input, par exemple les fertilisants, mais avec des cultures ayant des fonctions d'émission hétérogènes, la régulation optimale est alors un système de taxe/subvention différentié par culture de sorte que, pour chaque culture, les rendements marginaux soient égaux au rapport du prix des fertilisants plus la taxes/subventions sur le prix de vente. En effet, une taxe uniforme revenant à considérer que l'utilisation de fertilisants entraîne des dommages marginaux équivalents sur l'ensemble des exploitations agricoles, une taxe différentiée est indispensable pour prendre en compte l'hétérogénéité des sols et des cultures (Helfand et al., 2003).

Cependant, en pratique, cette option n'est pas applicable, essentiellement à cause des coûts de contrôle nécessaires à sa réalisation. Cela a amené Helfand and House (1995) à s'intéresser à la perte de bien-être due à la mise en place d'une taxe uniforme comparée à une taxe différenciée par culture. Ils montrent alors que cette perte est relativement mineure.C'est un résultat que Shortle et al. (1998) trouve surprenant mais il a été confirmé empiriquement par Martínez and Albiac (2006).

En présence d'hétérogénéité spatiale,Wu and Babcock (2001) s'intéressent à la différence d'efficacité entre la mise une place d'une taxe ou d'une norme uniforme. Si ces deux instruments permettent théoriquement d'atteindre l'optimum en l'absence d'hétérogénéité (Weitzman, 1974) et si la taxe uniforme est normalement supérieure à cette taxe dans les autres cas, ils montrent que la présence de solution en coin peut inverser les conclusions. Plus précisément, si l'efficacité des applications d'engrais est faible, les coûts marginaux de réduction de la pollution sont alors élevés car la plupart des engrais appliqués ne sont pas utilisés par les plantes et se transforment donc directement en perte. En outre, dans ce cas, les bénéfices marginaux sont peu sensibles aux taxes. Ces effets conjoints favorisent l'instauration d'une norme uniforme. En revanche, lorsque l'efficacité des applications d'engrais est élevée, les coûts marginaux d'abattement deviennent alors relativement faibles, et inversement, le profit marginal est plus réactif à l'instauration d'une taxe. Les prix agricoles, ou plutôt le prix relatif de l'engrais azoté par rapport à sa valeur moyenne dans la production, jouent également un rôle dans l'arbitrage taxe versus norme uniforme. En effet si cette valeur est faible, les agriculteurs sont incités à appliquer plus d'azote pour se prémunir contre les années au cours desquelles l'azote stocké dans les sols est perdu (Babcock and Blackmer, 1992). Ce type de comportement conduit à une faible efficacité de l'application d'engrais et favorise la mise en place d'une taxe uniforme. A un prix plus élevé l'application d'engrais est plus efficace. Enfin une forte variabilité des coûts marginaux, due par exemple à l'hétérogénéité de la qualité des terres et aux caractéristiques environnementales des exploitations agricoles, est de nature à favoriser une taxe uniforme étant donné qu'une norme uniforme peut s'avérer dans ce cas à la fois non-contraignante pour une partie des agriculteurs et prohibitive pour d'autres.

Cochard et al. (2005) étudient l'efficience comparée de plusieurs instruments : une taxe sur les inputs, une taxe sur les concentrations ambiantes et le mécanisme standard de taxe/subvention sur les concentrations ambiantes. Avec des données expérimentales, ils montrent que seule la taxe sur les inputs et la taxe ambiante sont efficaces et fiables. Les pollueurs diminuent leur usage d'intrants de façon optimale dans les deux cas. A l'inverse le mécanisme de taxe/subvention ambiante diminue le bien-être social comparé au scénario de statu-quo. Si son efficacité tend néanmoins à augmenter au cours du temps, il reste moins efficace que les deux instruments précédemment cités. Ce mécanisme conduit lui aussi à une diminution des intrants mais la possibilité de collusion qui maximise le montant de subvention perçue diminue le bien-être.

Outre ces quatre approches principales, il existe d'autres méthodes de régulation des inputs comme celle venant de la théorie des assurances. Les assurances visant à maintenir le revenu agricole, mises en place aux Etats-Unis notamment dans l'Iowa, peuvent conduire à des baisses d'intrants et donc de pollution. En effet, les assurances réduisent le risque de l'agriculteur sur son niveau de production et agissent ainsi comme une incitation à utiliser le niveau correct d'intrant. Ces assurances sont d'autant plus efficaces que les agriculteurs sont risques-averses et donc considèrent la sur-utilisation d'engrais comme visant à leur assurer, au minimum, un niveau de production pré-déterminé (Horowitz and Lichtenberg, 1993; Babcock and Hennessy, 1996). Smith and Goodwin

(1996) et Quiggin et al. (1994) montrent, empiriquement, qu'une assurance conduit à la diminution de l'usage d'intrant (fertilisants et pesticides). Wu (1999) s'intéresse à l'effet "extensif" de l'assurance, c'est-à-dire, que l'assurance peut inciter les agriculteurs à modifier la répartition entre les cultures. Cela provient du fait que l'assurance incite les agriculteurs à remettre en culture certaines terres laissées en jachère car le rapport risque/bénéfice n'était pas jugé satisfaisant ou n'incitait pas à se diversifier. Ils montrent alors que l'effet extensif conduit à une hausse des intrants surpassant la réduction d'intrants observés sur les autres parcelles cultivées (effet intensif).

Peu de papiers théoriques se sont intéressés à l'aspect dynamique des pollutions diffuses (Shortle and Horan, 2001). Xepapadeas (1992) montre qu'appliquer un contrat basé sur un mécanisme de régulation de la concentration ambiante est sous-optimal dans un cadre dynamique. Il propose alors des contrats sous la forme de taxe par unité de concentration sur l'écart entre le sentier désiré d'accumulation de polluants et le niveau réellement observé. Athanassoglou (2010) étudie une classe de jeux différentiels de contrôle de la pollution avec des fonctions de profit polynomiales. Il montre alors que le régulateur peut proposer aux agriculteurs une stratégie d'émission respectant un équilibre parfait à la Markov. Cette stratégie étant également parfaite en sous-jeu, i.e. à chaque instant t considéré, aucun agriculteur n'a intérêt à dévier et aucune taxe/subvention n'est nécessaire au cours de la dynamique. Goetz and Zilberman (2000) examinent, en information complète, la gestion socialement optimale, dans le temps et l'espace, de la pollution de l'eau par les engrais agricoles, à la fois chimiques et organiques. Pour cela ils déterminent d'abord l'allocation spatiale socialement optimale de la production agricole puis l'allocation intertemporelle socialement optimale. Ils concluent alors que l'optimum social est atteignable en taxant les engrais minéraux et organiques ainsi que le capital animal. Cependant l'hétérogénéité spatiale, liée à la fois aux caractéristiques agricoles et aux niveaux de contamination des eaux, les amène à proposer un système de zones relativement homogènes, mais pas forcément adjacentes. Si le régulateur possède suffisamment d'information, ils proposent la mise en place de taxes différenciées suivant les zones et, dans le cas contraire, l'instauration de quotas. Les quotas peuvent être échangés à l'intérieur des zones, et entre les zones avec un ratio d'échange variable dépendant de la différence entre la qualité moyenne des eaux des différentes zones. Le ratio se modifie alors au cours du temps suivant l'état relatif des zones d'eau polluées. Outre les inputs comme les engrais, d'autres facteurs, peu modélisés, comme la répartition des cultures ou les rotations influencent le stock de pollution. Goetz et al. (2006), étudient cet "extensive margin effect", dans un modèle dynamique où le choix d'allocation des cultures est une variable de décision mais où la surface totale de terre est non-contrainte. Ils montrent alors empiriquement, que ne pas réguler le choix d'allocation des cultures (en plus de la régulation des inputs) peut conduire à une perte d'efficacité de l'ordre de 20%.

De cette littérature relativement abondante, nous avons néanmoins identifié deux points essentiels qui nous semblent ne pas avoir été entièrement traités. Premièrement, s'il est apparu qu'une régulation optimale devait prendre en compte la notion 'd'extensive margin' et le choix d'allocation de cultures, peu de modèles théoriques considèrent un problème de régulation optimale des pollutions diffuses à l'aide d'une modèle de land-use avec qualité hétérogène des sols. Le chapitre 2 a pour but de montrer, à l'aide de ces modèles apparus récemment essentiellement pour traiter des questions de cultures bioénergétiques, que les politiques de taxation des inputs, même différenciées, ne sont alors plus optimales et peuvent conduire à des résultats non désirés. Nous proposons également des pistes pour se rapprocher de l'optimum de premier rang. Deuxièmement, d'un point de vue dynamique, il ressort que les modélisations proposées sont plus adaptées aux pollutions diffuses comme les eaux de surface. Dans les masses d'eau souterraine, le délai entre l'application de fertilisants et le moment où ceux-ci rejoignent effectivement les eaux souterraines est susceptible de jouer un rôle dans les politiques publiques. Le chapitre 3 étudie un problème dynamique de contrôle optimal quand la pollution est retardée avec et sans asymétrie d'information.

Chapitre 2

Agricultural non-point source pollution control policies : the effect of land-use change

Ce chapitre est issu d'une collaboration avec Vincent Martinet

2.1 Introduction

Non-point source pollution is a major issue in the European Union. In particular, the use of nitrogen fertilizers by agricultural activity is responsible of water pollution by nitrates and contributes to global warming by nitrous oxyde. The European Commission, concerned by these pollutions, set targets to mitigate them. As regards climatic change, total EU greenhouse gas emissions from the sectors currently not covered by the ETS [1], including the agricultural sector, have to be reduced by approximately 10% in 2020, relative to 2005 levels (European Union, 2009). As regard water pollution, the Water Framework Directive (2000/60/EC) deals with water quality conservation by preventing nitrogen pollution by the agricultural sector.

The "nonpoint source" characteristics of agricultural pollution makes first-best emission instruments inappropriate to regulate it (Helfand and House, 1995). Among the second-best policies reviewed in the previous chapter, Helfand et al. (2003) show that a taxation of fertilizer use differentiated by crops may restore the first-best solution. Numerous empirical studies emphasize, however, the relative inefficiency of input taxation for water pollution (Lacroix et al., 2005; Aftab et al., 2010). Even if these studies consider homogeneous taxation (and not a taxation differentiated by crop), Helfand and House (1995) and Martínez and Albiac (2006) show that the welfare loss of implementing a homogeneous taxation instead of a tax differentiated by crop is relatively low. The nature of the taxation (homogeneous versus heterogeneous) is thus not the main reason of their inefficiency.

It appears (and we shall prove that point in this chapter) that the positive

1. European Trading Share

result of Helfand et al. (2003) is in fact a particular case of a more general result taking into account the land use reallocation effects of taxation instruments. A tax on fertilizers differentiated by crops modifies the agricultural intensity and can restore optimality with respect to the intensive margins. The model of Helfand et al. (2003), however, did not account for the optimal sharing of land among the various crops, and did not consider what happens at the extensive margins of the agricultural system. When one accounts for this land use dimension, an additional policy instrument modifying the crop area choice is required to reach the fist-best optimum (Goetz et al., 2006). It is necessary to account for the land-use changes induced by a policy via the change of the relative profitability of the various crops to assess properly its impact on pollution reduction.

More specifically, a regulation based on instruments affecting input price makes it possible to equalize the marginal profit of fertilizer use among crops, but has no reason to be optimal with respect to land use allocation among crops. This is especially true when the emission functions are heterogeneous and depend on the crop type. In particular, as it is not always the most polluting crop which consumes the largest amount of fertilizers (or which is the least profitable), input taxation may induce a land reallocation in its favor. The result of the policy in terms of welfare may then be ambiguous. This is mainly the case of water pollution caused by the NO_3 which strongly depends on crop. However, for other types of pollution, slightly heterogeneous, such as air pollution caused by N_2O, an input based instrument can be close to optimality. When the regulation of several pollutions due to the same input is at stake, a second-best policy like an input taxation may result in the increase of some pollutions (while the others decrease).

Even if taking into account the extensive margins effect limits this paradoxical effect and restores first-best optimality and it is unrealistic to propose a regulation based on a large number of instruments (in fact, an instrument for each crop to regulate fertilizer use, and instruments to control crop area choices). A solution could be to implement a homogeneous taxation on fertilizer (in order to regulate intensive margins) and a subsidy for a non-polluting crop (in order to regulate the extensive margin). We show that this couple of instruments, even if it is not optimal, can improve the efficiency of non-point source regulation in comparison with both heterogeneous and homogeneous taxation on fertilizer use

The further analyses are based on the land-use share model presented in section 2. In section 3, we first consider the first-best policy which provides a benchmark to compare the effects of second-best policies when a land-use model with heterogeneous soil quality is considered. In section 4, we show that usual input-based instruments (homogeneous and heterogeneous input taxations) can lead to a paradoxical effect on pollution. In section 5, we show that only a policy mix can solve this paradox, but that requires the use of too many instruments. In section 6, we discuss about what second-best policy should be adopted depending on the N-pollution considered.

2.2 A land-use share model of agricultural production

In order to compare the described policy options to manage non-point source pollutions from agricultural production, we shall use the following land-use share model. In this section, we consider the land owner maximization problem.

Land use We assume that the land of a given area is heterogeneous in terms of agronomic quality, and that each land quality can be devoted either to crop production (i.e., to various annual agricultural commodities production, such as corn and wheat) or to an extensive, perennial use such as grassland, forestry or perennial crops dedicated to biomass production (e.g., switchgrass or miscanthus). We use i as an annual crop index, with $i = 1$ for corn, $i = 2$ for wheat. We use o as an index for the other, perennial land allocation.

To reflect land heterogeneity, each parcel of land is associated with a land quality index, q, indicating yield potential. The index is normalized to vary between zero and one. Higher q indicates better quality ; thus the best land has $q = 1$, while the worst land has $q = 0$. The total agricultural area is normalized to one. Defining the density function of land quality by $\phi(q)$, the amount of land with $q \leq \hat{q}$ is $\int_0^{\hat{q}} \phi(q)dq$.

The share of land of quality q devoted to crop i is denoted by $\alpha_i(q)$.

Yield The supply of agricultural commodities and the agricultural profit of the area depend on the overall quantity of land devoted to croplands, on the associated land qualities, on the relative share of land allocated to the various possible crops on a given land quality, as well as on input use.

The input use for crop i on a soil of quality q is denoted by $f_i(q)$. Crop yield on a field of quality q is given by function y_i depending on the soil quality q, the fertilizer use f_i, and the share α_i of land of quality q devoted to crop i, i.e., $y_i(q, f_i, \alpha_i)$. This yield function is assumed to be continuous and twice differentiable in the arguments. Moreover, we assume that y_i is concave and increasing in f_i, i.e., $\frac{\partial y_i}{\partial f_i} > 0$ and $\frac{\partial^2 y_i}{\partial f_i^2} < 0$. We also assume that the soil quality has a positive effect on the yield, i.e., $\frac{\partial y_i}{\partial q} > 0$. Last, we assume that the yield of crop i (production per area unit) decreases with the share of land of a given quality devoted to that crop, i.e., $\frac{\partial y_i}{\partial \alpha_i} \leq 0$. This assumption reflects the fact that a crop production system with a large share of a given crop is faced with yield decline due to the importance of the crop in the agronomic rotations, and related soil fertility erosion or diseases emergence. [2]

Optimal crop allocation Let agricultural output and input prices be given and denoted respectively by $p \equiv (p_1, p_2)$ and w. For any given soil quality q, the per area unit net return from crop production is given by the profit function $\pi(q, f, \alpha, \tau)$, where$f \equiv (f_1, f_2)$ and $\alpha \equiv (\alpha_1, \alpha_2)$, defined as follows [3] :

$$\pi(f, \alpha; q) = \sum_{i=1,2} \alpha_i \left(p_i y_i(f_i(q), \alpha_i(q); q) - w f_i(q) \right). \tag{2.1}$$

We assume that land owners compute the potential agricultural rent from land of quality q by determining the optimal allocation of the land between the two

2. This is an usual assumption (Feng and Babcock, 2010). One could assume increasing marginal production costs (with respect to land use share) instead of decreasing marginal yield without changing the results. Such increasing costs would correspond to additional treatments of the crops to avoid diseases.

3. We assume that $f_i(q)$ and $\alpha_i(q)$ are input and crop share decisions of one or more identical producers facing land quality q. Alternatively,they could represent the decisions of a representative producer standing for all producers with a certain land quality.The interpretation of results is the same across either interpretation

crops as well as the optimal input, i.e., by maximizing the agricultural gross margin. Optimal land allocation and fertilizer use are given by the following optimization problem :

$$\begin{aligned} \max_{f,\alpha} \quad & \pi(f,\alpha;q) \qquad (2.2) \\ s.t. \quad & \sum_{i=1,2} \alpha_i(q) \leq 1 \ , \\ & f_i(q) \geq 0, \qquad i=1,2 \ , \\ & \alpha_i(q) \geq 0, \qquad i=1,2. \end{aligned}$$

Assuming interior solutions,[4] the first-order conditions of problem (2.2) are

$$p_i \frac{\partial y_i}{\partial f_i} - w = 0 \ , \qquad i=1,2 \ . \qquad (2.3)$$

$$p_1\left(y_1 + \frac{\partial y_1}{\partial \alpha_1}\alpha_1\right) - wf_1 = p_2\left(y_2 + \frac{\partial y_2}{\partial \alpha_2}\alpha_2\right) - wf_2 \ . \qquad (2.4)$$

We assume that the second-order conditions are satisfied. Optimal input use and cropland sharing are functions of both prices and land quality, and so we denote the optimal solution to (2.3) and (2.4) by $f_i^*(q)$ and $\alpha_i^*(q)$. For each crop, eq. (2.3) means that the marginal revenue from an increase in input use equals its marginal cost. Equation (2.4) implies that the marginal profits of both crops with respect to their relative share of the land of that quality are equal.

4. Assuming an interior solution amounts to consider that there is no "monoculture" production system, i.e., no land where only one crop is produced permanently (e.g., corn or wheat, exclusively). This makes sense from an agronomic point of view, as crops enter rotations, and a given field may be allocated to different crops from year to year, to maintain soil productivity and reduce disease emergence. To encompass this dynamic aspect in our static framework, we assume that the relative share of each crop on a land is given by its degree of recurrence over time.

Note that the optimized net return is higher for better land because better land has higher yield for any given land allocation and input use. Mathematically, this means that, for any $q > q'$, one has

$$\pi(f^*(q), \alpha^*(q); q) > \pi(f^*(q'), \alpha^*(q'); q') \ . \tag{2.5}$$

Perennial crop rent As an alternative to annual crop production, we assume that land can be allocated to a perennial production, such as forestry, grassland, or a perennial energy crop (switchgrass or miscanthus). For the sake of simplicity, we assume that there is no fertilizer use on these perennial productions, i.e., $f_o = 0$.

The per area unit profit of this alternative production on a soil of quality q is given by the function $\pi_o(q)$. Note that this alternative profit can depend on the soil quality.

Land use rent maximization : optimal land use In a given economic context (i.e., for given input and output prices), the land of a given quality will be allocated to the use that provides the higher rent over all alternative uses. We consider that a given plot of land of quality q is allocated to crop production if the maximal agricultural rent $\pi(f^*, \alpha^*; q)$ exceeds (per area unit) the rent from perennial agricultural area land uses $\pi_o(q)$.

This type of land use share model will allocate land over a compact set of quality to a given use. And in particular, all the land of a given quality will be allocated to the same use. Assuming that better lands are allocated to crop production, we denote the quality range of land devoted to common crops by $[Q^*, 1]$ and the quality range of land devoted to perennial crop by $[0, Q^*]$. This means that, for all $q \geq Q^*$, we have :

$$\pi(f^*, \alpha^*; q) \geq \pi_o(q) \ . \tag{2.6}$$

Agricultural supply and agricultural profit Total cropland area equals $\int_{Q^*}^{1} \phi(q)dq$ and the area devoted to perennial crops equals $\int_{0}^{Q^*} \phi(q)dq$. As more land is brought into perennial production (due to the policy instruments), the change in cropland area (resp. perennial crops) is equal to $-\int_{Q^*}^{Q^*+\Delta Q^*} \phi(q)dq$ (resp +), where ΔQ^* is the change of the soil quality threshold between annual and perennial crops. Being a density function, $\phi(q)$ is always nonnegative.

Given land owners decisions on land share, input use, and marginal cropland, one can determine crop supply and agricultural profit. Summing over land of different quality, the total agricultural profit (from both perennial and annual productions) can be written as follows :

$$\Pi(\tau) = \int_0^{Q^*} \pi_o(q)\phi(q)dq + \int_{Q^*}^{1} \pi\Big(f^*(q), \alpha^*(q); q\Big)\phi(q)dq. \tag{2.7}$$

Land owner's profit optimization problem To determine the decentralized optimum, one needs to define the optimal fertilizer use, cropland sharing and land shares that maximize the private agents profit in a given economic context.

The land owners' optimization problem can be stated as follows :

$$\begin{aligned} \max_{f,\alpha,Q} \Pi(t) &= \max \int_0^{Q} \pi_o(q)\phi(q)dq \qquad (2.8) \\ &+ \int_Q^1 \left(\sum_{i=1,2} [p_i y_i(f_i(q), \alpha_i(q); q) - w f_i(q)] \alpha_i(q) \right) \phi(q)dq \ . \end{aligned}$$

This optimization problem defines, in a given context for prices, the private optimum for fertilizer use for any crop and soil quality ($f_i^*(q)$), the optimal land use sharing between perennial and annual crops (Q^*) and the optimal crop allocation on cropland ($\alpha_i^*(q)$).

The optimality conditions of this problem for fertilizer use and crop allocation are the same as eqs. (2.3) and (2.4) above.

The optimality conditions of this problem for land use share between cropland and perennial crop is $\frac{\partial \Pi}{\partial Q} = 0$, which requires

$$\pi_o(Q^*)\phi(Q^*) - \left(\sum_{i=1,2} [p_i y_i(f_i(Q^*), \alpha_i(Q^*), Q^*) - w f_i(Q^*)]\alpha_i(Q^*) \right) \phi(Q^*) = 0$$

or, equivalently if $\phi(Q^*) \neq 0$,

$$\pi_o(Q^*) - \left(\sum_{i=1,2} [p_i y_i(f_i(Q^*), \alpha_i(Q^*), Q^*) - w f_i(Q^*)]\alpha_i(Q^*) \right) = 0. \tag{2.9}$$

Figure 2.1 displays the land-use share as a function of soil quality without instrument policies.

Agricultural pollution The use of fertilizers in agricultural activities is a source of air and water pollutions.

We shall here first consider a generic function for pollutant emissions, $e_i(f_i)$ which depend on level of fertilizer per hectare f_i applied on crop i. We assume that perennial crops do not require fertilizers, and are thus non-polluting (i.e., $e_o(0) = 0$).

Total pollution is the sum of pollutant emitted on each cropland (i.e., for all

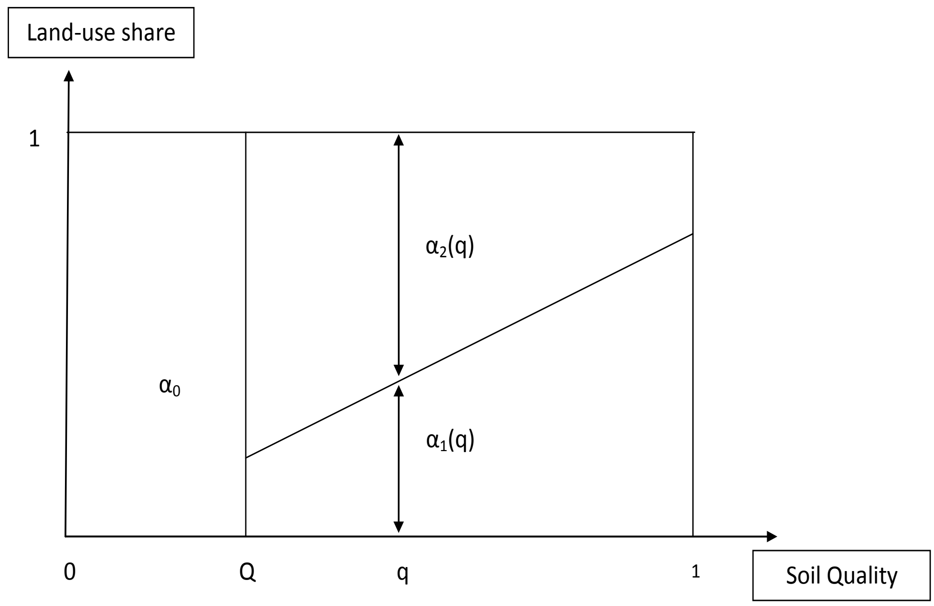


FIGURE 2.1 – Land-use share in function of soil quality

land quality $q \geq Q$), for both crop types. We have

$$E(f, \alpha, Q) = \int_Q^1 \left(\sum_{i=1,2} \alpha_i(q) e_i(f_i(q); q) \right) \phi(q) dq \ . \tag{2.10}$$

The level of emissions is function of fertilizer use and crop area allocation for a given q, as displayed in figure 2.2.

The total level of emissions is illustrated on figure 2.3.

2.3 Non-point source pollution control

In this section, we discuss various policy options to regulate the non-point source pollution problem described above.

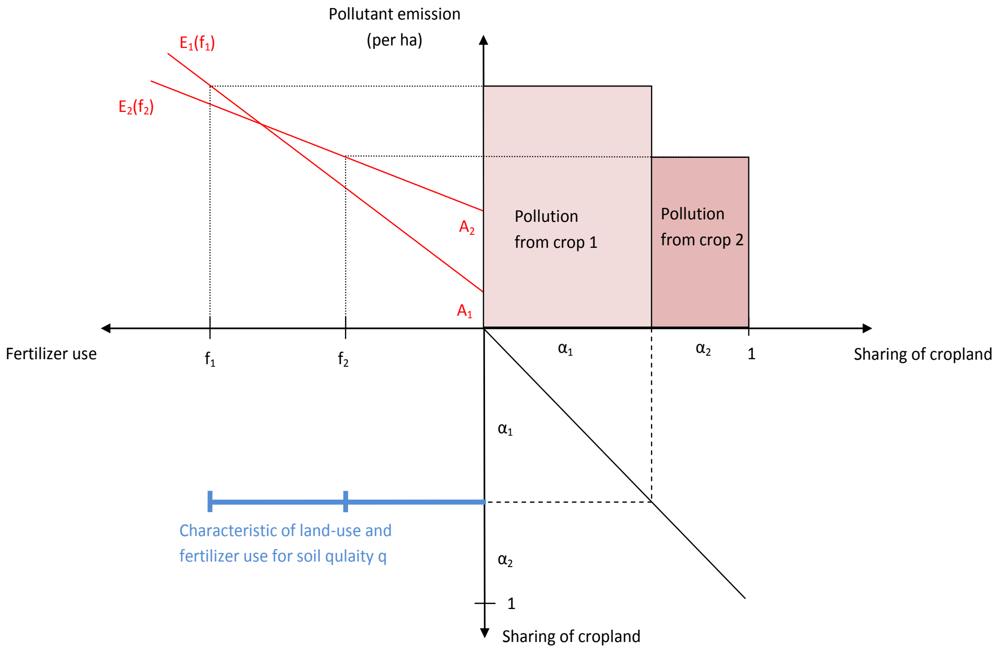


FIGURE 2.2 – Emission level for a given soil quality

2.3.1 First best solution : Optimal taxation of emissions

We first consider the first-best policy. This will provide a benchmark to compare the effects of second-best policies, and emphasize the importance of considering land-use change effects in policy design. To keep the results as general as possible, we do not specify functional forms. We compare the various cases (first best and second best policies) by comparing the associated optimality conditions.

We shall here consider that the social planner is able to define a per unit Pigouvian tax for pollution emission, and that the tax level is set optimally to equalize the marginal private profit to the marginal social cost of pollution. At this stage, we do not describe the social planner problem and the way the optimal tax is defined. We shall consider this point in section 2.6, along with

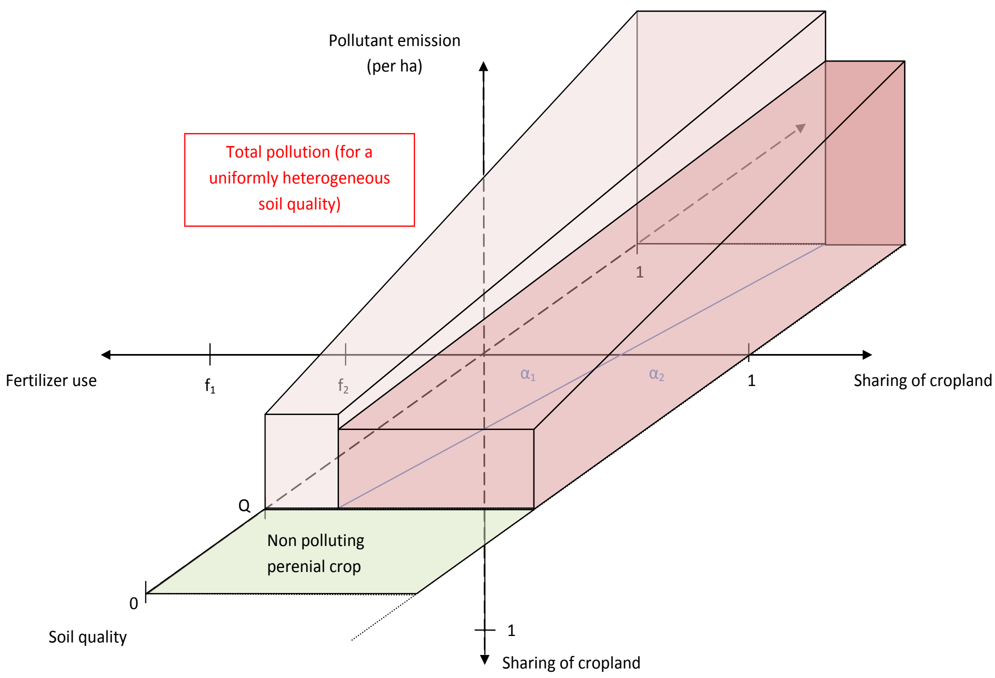


FIGURE 2.3 – Total emission level

the welfare analysis of the various policies. For the time being, we just examine the effect of a tax on pollutant emission on the optimal private decisions of fertilizer use, and land-sharing.

Given the generic emission function (per hectare), $e_i(f_i)$, the first-best policy requires setting an emission tax at some level t. Faced with this tax, the land owners internalize the externality of pollution and modify their emission behavior.

The land owners' optimization program can thus be written again as follows :

$$\begin{aligned}\max_{f,\alpha,Q} \Pi(t) &= \max \int_0^Q \pi_o(q)\phi(q)dq \qquad (2.11)\\ &+ \int_Q^1 \left(\sum_{i=1;2} [p_i y_i(f_i(q), \alpha_i(q); q) - w f_i(q) - t e_i(f_i(q); q)]\alpha_i(q) \right) \phi(q)dq\end{aligned}$$

Assuming interior solutions, we can write the first-order conditions of the optimization (2.11) as :

$$\begin{aligned}[p_1 y_1 + p_1 \frac{\partial y_1}{\partial \alpha_1}\alpha_1 - w - (p_2 y_2 + p_2 \frac{\partial y_2}{\partial \alpha_2}\alpha_2 - w)] &= \tfrac{\partial \pi_1}{\partial \alpha_1} - \tfrac{\partial \pi_2}{\partial \alpha_2} \qquad (2.12)\\ &= t(e_2(f_2) - e_1(f_1)) \ ,\\ p_i \frac{\partial y_i}{\partial f_i} - w = \frac{\partial \pi_i}{\partial f_i} &= t\tfrac{\partial e_i(f_i)}{\partial f_i}, \quad i = 1, 2. \qquad (2.13)\end{aligned}$$

The land-use share between perennial and other crops is determined as the equivalent of condition (2.9) :

$$\pi_o(Q^*) - \left(\sum_{i=1;2} [p_i y_i(f_i(Q^*), \alpha_i(Q^*), Q^*) - w f_i(Q^*) - t e_i(f_i(Q^*))]\alpha_i(Q^*) \right) = 0. \qquad (2.14)$$

Eq. (2.13) is standard. At the optimum, the marginal profit of fertilizers use equals the marginal emission cost (tax times the marginal emissions) for each crop. Eqs. (2.12) and (2.14) are specific to land-use share model. At the optimum, the marginal profit of crop allocation equals the tax times the difference of crop emissions (eq.2.12). Corn and wheat are planted by the farmers once the soil quality is high enough, i.e., higher than the threshold quality Q^* implicitly defined by (eq.2.14).

Figure 2.4 displays the land-use effects, both crop area reallocation and land-use share, induced by the first-best policy.

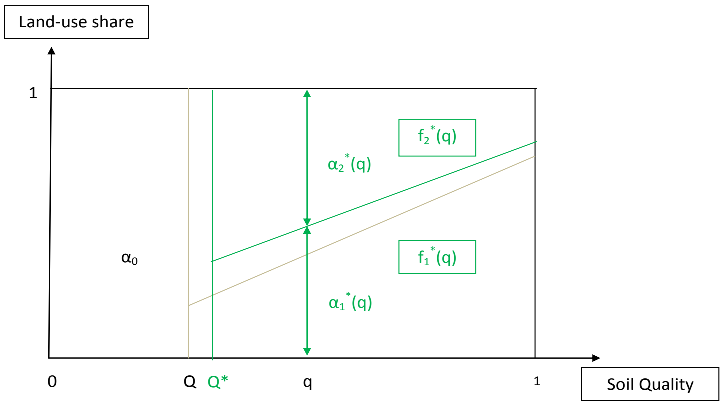


FIGURE 2.4 – Land-use effects induced by the first-best policy

On the impossibility to implement the first best solution for non-point source pollution problems

In a nonpoint source pollution problem, the social planner cannot attribute to each farmer its contribution to the pollution. It is thus not possible to tax the emissions directly. As nitrogen pollution (N-losses) are strongly related to the type of crop and soil, it is theoretically possible to tax each farmer on its crop areas and fertilizer use. It would, however require to control the crop area and the fertilizer use (for each crop) for each farmers (along with climatic conditions) to assess properly the actual emissions of pollutants and reach optimality. Therefore, a second-best solution, based on N-input taxation, is often adopted in the literature.

We shall examine in the next section what the conditions for such a homogeneous taxation to restore the first-best optimum are.

2.3.2 Second best solution : Homogeneous input taxation

In response to this information problem, we consider now a regulation based on an uniform fertilizer tax. Here again, we shall examine the optimal response of private land owners to the policy, without describing the way its (second-best) optimal level is defined. This issue will be addressed in sect 2.6.

Within the framework described in section 2.2, we consider a per unit tax on fertilizer use, at level τ. This tax induces an additional cost τf_i related to agricultural production.

The land owners's optimization problem can be written again as follows :

$$\begin{aligned} \max_{f,\alpha,Q} \Pi(\tau) &= \int_0^Q \pi_o(q)\phi(q)dq && (2.15) \\ &+ \int_Q^1 \left(\sum_{i=1,2} [p_i y_i(f_i(q), \alpha_i(q); q) - (w+\tau) f_i(q)]\alpha_i(q) \right) \phi(q)dq. \end{aligned}$$

For any given q, net return is maximized through decisions on land allocation between crops and input use for each crops. And the following optimization problem is solved as :

$$\max_{f,\alpha} \pi(f,\alpha;q) \quad \text{such that} \quad \sum_{i=c,w} \alpha_i(q) \leq 1 \ \text{ and } \ f_i(q) \geq 0, \quad \alpha_i(q) \geq 0. \quad (2.16)$$

Assuming interior solutions, we can write the first-order conditions of (2.16) as

$$[p_1 y_1 + p_1 \frac{\partial y_1}{\partial \alpha_1}\alpha_1 - w - (p_2 y_2 + p_2 \frac{\partial y_2}{\partial \alpha_2}\alpha_2 - w)] = \quad \tau f_2 - \tau f_1 \ , \tag{2.17}$$

$$p_i \frac{\partial y_i}{\partial f_i} - w = \quad \tau, \qquad i = 1,2. \tag{2.18}$$

The land-use share between perennial and other crops is determined as the equivalent of condition (2.9) :

$$\pi_o(\hat{Q}) - \left(\sum_{i=1;2} [p_i y_i(\hat{f}_i(\hat{Q}), \hat{\alpha}_i(\hat{Q}); \hat{Q}) - (w+\tau)\hat{f}_i(\hat{Q})]\hat{\alpha}_i(\hat{Q}) \right) = 0. \tag{2.19}$$

We can compare these first order condition with that of the first-best problem (2.11).

From a general point of view, homogeneous input taxation will lead to the optimal emissions if the first order conditions of problem (2.11) and problem (2.15) coincide. By comparing eqs. (2.12) and (2.17) on the one hand, and eqs. (2.13) and (2.18) on the other hand, the two optimality conditions coincide if the following system is satisfied :

$$\tau = t \frac{\partial e_i(f_i)}{\partial f_i} \ , \tag{2.20}$$

$$\tau f_2 - \tau f_1 = t(e_2(f_i) - e_1(f_i)). \tag{2.21}$$

The first equation tells us that this is possible only if $\frac{\partial e_i(f_i)}{\partial f_i}$ is constant, i.e., if the emissions of the pollutant are proportional to the quantity of fertilizer applied, with the same emission factor for all crops. This means that the emission functions $e_i(f_i)$ must be of the form $e_i(f_i) = A_i + Bf_i$, where $A_{1,2}$ and B

are constant parameters. Condition (2.21) is then satisfied only if $A_1 = A_2$.

In other words, fertilizer taxation results in the same fertilizer application $f_i^*(q)$ and cropland sharing $\alpha_i^*(q)$ as in the first-best case only if the pollution is linear with respect to the fertilizer application and if the emission function are the same for both crops, i.e., if the crops are "homogeneous" with respect to pollution emission as a function of fertilizer application.

Let us assume from the rest of this section that this condition is satisfied, i.e., that $e_1(f_i) = e_2(f_i) = A + Bf_i$. Under this condition, $\hat{f}_i(q) = f_i^*(q)$ and $\hat{\alpha}_i = \alpha_i^*(q)$.

We now turn to the condition on overall land-use share, i.e., the sharing of land between annual cropland and perennial use. It turns out that condition (2.19) coincides with condition (2.14) if $A = 0$. To prove that point, let us replace the various element in eq. (2.19) by their value under the hypothesis $e_1(f_i) = e_2(f_i) = A + Bf_i$, and set the fertilizer tax at the appropriate level to restore the previously studied optimality conditions, i.e., $\tau = tB$. Condition (2.19) becomes

$$\pi_o(\hat{Q}) - \left(\sum_{i=1;2} [p_i y_i(f_i(\hat{Q}), \alpha_i(\hat{Q}); \hat{Q}) - (w + \tau) f_i(\hat{Q})] \alpha_i(\hat{Q}) \right) = 0$$

$$\Leftrightarrow \quad \pi_o(\hat{Q}) - \left(\sum_{i=1;2} [p_i y_i(f_i^*(\hat{Q}), \alpha_i^*(\hat{Q}); \hat{Q}) - w f_i^*(\hat{Q}) - tBf_i^*(\hat{Q})] \alpha_i^*(\hat{Q}) \right) = 0$$

which is different from condition (2.14) for the linear pollution functions :

$$\pi_o(Q^*) - \left(\sum_{i=1;2} [p_i y_i(f_i(Q^*), \alpha_i(Q^*); Q^*) - w f_i(Q^*) - tA - tBf_i^*(Q^*)] \alpha_i(Q^*) \right) = 0$$

Conditions for optimality

To sum up, from a general point of view, we can say that an input tax is not sufficient to restore optimality, neither in terms of fertilizers use nor in terms of allocation area. The land-use share between perennial and other crop is also non optimal.

Optimality is restored only if the emissions are proportional to the fertilizer use (with the same marginal effect for all crops), with no fix effect. If the crop emissions function are affine and have the same marginal response to fertilizers, i.e : $e_i(f_i) = A_i + Bf_i \ \ \forall i$, an uniform taxation restore the optimal use of fertilizers. The crop allocation is, however, not optimal. If, furthermore, all crops have the same fixed emission, i.e., $e_i(f_i) = A+Bf_i \ \ \forall i$, then the share of crops is optimal. The overall land allocation may, however, still be non-optimal. Full optimality requires the emissions functions to be of the simplest form $e_i(f_i) = Bf_i \ \ \forall i$. We have thus shown that an uniform taxation restore the optimality if and only of the emission functions are homogeneous (i.e., the emission functions do not depend on crops)

In reality, the emission function are heterogeneous and even strongly heterogeneous in case of NO_3 pollutant.[5]

How can we restore the optimum if $e_i(f_i) = A_i + B_i f_i$? As the pollution is heterogeneous by crops, the immediate next step is to focus on input taxation differentiated by crop.

5. An illustration of wide heterogeneity of emission functions differentiated by crops in France is provided in Chapter 5, Appendix 1, Table 1

2.3.3 Second best solution : Heterogeneous input taxation

In this section, we consider that the decision maker is able to tax fertilizer use differently depending on the crop. We denote the fertilizer tax related to crop i by τ_i, i.e., $\boldsymbol{\tau} \equiv (\tau_1, \tau_2)$ is the tax vector.

The land owners' optimization problem becomes :

$$\begin{aligned}\max_{f,\alpha,Q} \Pi(\tau_1,\tau_2) &= \int_0^{Q(\boldsymbol{\tau})} \pi_0(q)\phi(q)dq \qquad (2.22)\\ &+ \int_{Q(\boldsymbol{\tau})}^1 \left(\sum_{i=1,2} [p_i y_i(f_i,\alpha_i;q) - (w+\tau_i) f_i(q)]\alpha_i(q) \right) \phi(q)dq.\end{aligned}$$

Local optimal crop allocation and fertilizer use requires :

$$\max_{f,\alpha,Q} \Pi(\tau_1,\tau_2) \ \text{ such that } \ \sum_{i=c,w} \alpha_i(q) \le 1 \ \text{ and } \ f_i(q) \ge 0, \ \ \alpha_i(q) \ge 0. \quad (2.23)$$

Assuming interior solutions, we can write the first-order conditions of the optimization program (2.23) as :

$$[p_1 y_1 + p_1 \frac{\partial y_1}{\partial \alpha_1}\alpha_1 - w - (p_2 y_2 + p_2 \frac{\partial y_2}{\partial \alpha_2}\alpha_2 - w)] = \tau_2 f_2 - \tau_1 f_1 \qquad (2.24)$$

$$p_i \frac{\partial y_i}{\partial f_i} - w = \tau_i \ , \qquad i = 1,2. \quad (2.25)$$

The land-use share between perennial and other crops is determined as the equivalent of condition (2.9) :

$$\pi_o(\hat{Q}) - \left(\sum_{i=1;2} [p_i y_i(\hat{f}_i(\hat{Q}), \hat{\alpha}_i(\hat{Q}); \hat{Q}) - (w+\tau_i)\hat{f}_i(\hat{Q})]\alpha_i(\hat{Q}) \right) = 0. \qquad (2.26)$$

From a general point of view, heterogeneous input taxation will lead to the optimal emissions if the first order conditions of problem (2.11) and problem (2.23) coincide. By comparing eqs. (2.12) and (2.24) on the one hand, and eqs. (2.13) and (2.25) on the other hand, the two optimality conditions coincide if the following system is satisfied :

$$
\begin{aligned}
\tau_i &= tB_i \ , \qquad i = 1, 2. \\
\tau_2 f_2 - \tau_1 f_1 &= t(A_2 + B_2 f_2 - A_1 - B_1 f_1) \ .
\end{aligned}
$$

The first equation tells us that this is possible only if $\frac{\partial e_i(f_i)}{\partial f_i}$ is constant for both crops, i.e., if the emissions of the pollutant are proportional to the quantity of fertilizer, with an emission factor differentiated by crop. This means that the emission functions $e_i(f_i)$ must be of the form $e_i(f_i) = A_i + B_i f_i$, where $\{A_1, B_1\}$ is different to $\{A_2, B_2\}$. The second condition is then satisfied only if $A_1 = A_2$.

This result allows us to state that a differentiated taxation is required to restore optimal fertilizer use, but that the optimal crop sharing is achieved only if the pollution functions have the same fix effect.

However, as in the previous case of homogeneous input taxation, even if the emission function have the same constant A, it is not sufficient to restore the optimal land-use share allocation defined by eq (2.14) :

Let us replace the various element in eq. (2.26) by their value under the hypothesis $e_i(f_i) = A + B_i f_i$, and set the fertilizer tax at the appropriate level to restore the previously studied optimality conditions, i.e., $\tau_i = tB_i$. Condition

(2.19) becomes

$$\pi_o(\hat{Q}) - \left(\sum_{i=1;2} [p_i y_i(f_i(\hat{Q}), \alpha_i(\hat{Q}); \hat{Q}) - (w + \tau_i) f_i(\hat{Q})] \alpha_i(\hat{Q}) \right) = 0$$

$$\Leftrightarrow \quad \pi_o(\hat{Q}) - \left(\sum_{i=1;2} [p_i y_i(f_i^*(\hat{Q}), \alpha_i^*(\hat{Q}); \hat{Q}) - w f_i^*(\hat{Q}) - t B_i f_i^*(\hat{Q})] \alpha_i^*(\hat{Q}) \right) = 0$$

which is different from condition (2.14) for the linear pollution functions :

$$\neq \quad \pi_o(Q^*) - \left(\sum_{i=1;2} [p_i y_i(f_i(Q^*), \alpha_i(Q^*); Q^*) - w f_i(Q^*) - tA - t B_i f_i^*(Q^*)] \alpha_i(Q^*) \right) = 0$$

Here again, the fix effect of crops on pollution is not supported, and the optimality is achieved only if $A = 0$.

To sum up, two instruments (i.e ; as many instruments as there are crops requiring fertilizer use) are required to restore optimal fertilizer application, but are not sufficient to restore the optimality with respect to both fertilizers use and crop area allocation when the land-use effects are taken into account.

2.4 Research question

2.4.1 Realistic assumptions ?

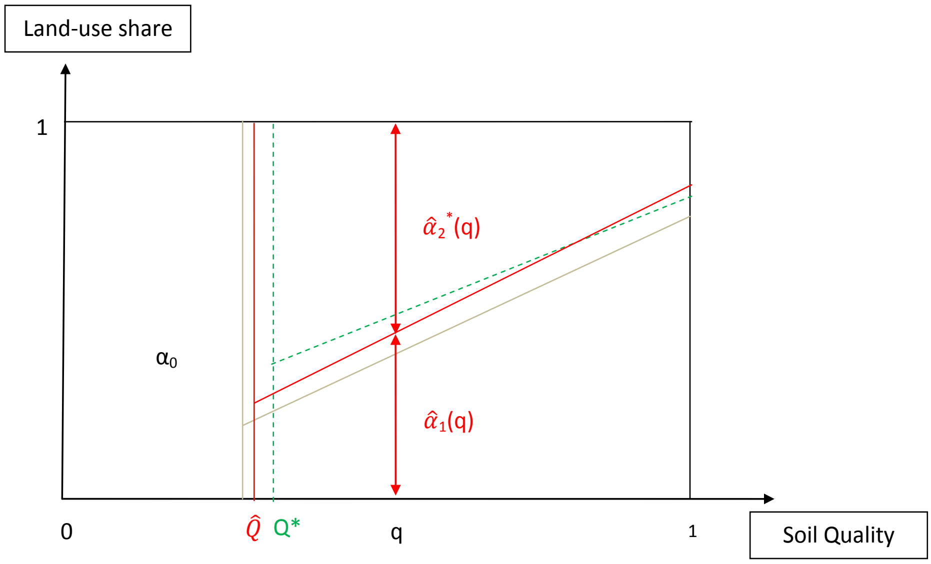
We have shown that, even if a heterogeneous taxation restores the optimal use of fertilizers, such a taxation does not restore the optimal crop allocation area when different crops have different emission rates. Moreover, imposing a fertilizer tax differentiated by crops is an unrealistic policy for two reasons. First, it requires too many instruments, as many instruments as there are crops, to restore the optimal use of fertilizers. Second, it seems unrealistic to fix different prices (or taxation) for the same input. The social planner is thus faced with two problems : decreasing the number of instrument and coming closer to optimal crop allocation.

This second point, optimal land use or crop allocation, is in fact very important, and its effect on pollution has not been studied a lot in the literature. The following section illustrates that, in some cases, input taxation may increase pollution, if there is a land use change favoring the most polluting crops. This is a sort of "rebound" effect.

2.4.2 A paradoxical effect of input taxation

In this section, we focus on the undesirable effects of a policy neglecting land-use effects. For this purpose, we consider the affine emission function described above, i.e., $e_i(f_i) = A_i + B_i f_i$. We consider a heterogeneous taxation on input use, with $\boldsymbol{\tau} \equiv \{\tau_1, \tau_2\}$. We have seen that an input tax, even differentiated by crops, does not lead to the optimal crop allocation area (illustrated in fig. 2.4.2). If we consider one pollution like the NO_3 for which the emissions function are very heterogeneous for different crops, the land use reallocation can result in an increasing pollution (see Fig. 2.4.2, for an illustration of the effect of land-use reallocation for a given soil quality q). The tax is thus non efficient (its optimal value in a second-best setting should be zero). If we consider two pollutants, one of them having emission function slightly heterogeneous over the different crops, such as N_2O pollution, the optimal tax is efficient for this one but can lead to increase the other pollution.

Let $E(\boldsymbol{\tau}) = \int_0^1 (\sum_{1,2} e_i(f_i^*(\boldsymbol{\tau}; q))\alpha_i^*(\boldsymbol{\tau}, q); q)\phi(q)dq$ denote the pollution at the optimum. As we assume zero emissions for the perennial crop, we can express



E as follows :

$$E(\boldsymbol{\tau}) = \int_{Q^*(\boldsymbol{\tau})}^{1} \sum_{i=1,2} e_i(f_i^*(\boldsymbol{\tau};q), \alpha_i^*(\boldsymbol{\tau};q)\phi(q))dq.$$

We denote the total emissions due to the crop i by

$$E_i(\boldsymbol{\tau}) = \int_{Q^*(\boldsymbol{\tau})}^{1} (e_i(f_i^*(\boldsymbol{\tau};q)), \alpha_i^*(\boldsymbol{\tau};q)\phi(q))dq.$$

We decompose the effects of a heterogeneous input tax on pollution in order to emphasize the different drivers. Assume for this purpose that the social planner does not account for the land-use change effect of his policy, and aims at restoring optimal fertilizer use by setting a specific input tax on each crop. We have seen that these taxes one must satisfy $\tau_i = tB_i$, where t is the optimal pollution tax level. In this setting, both taxes τ_1 and τ_2 are proportional. We

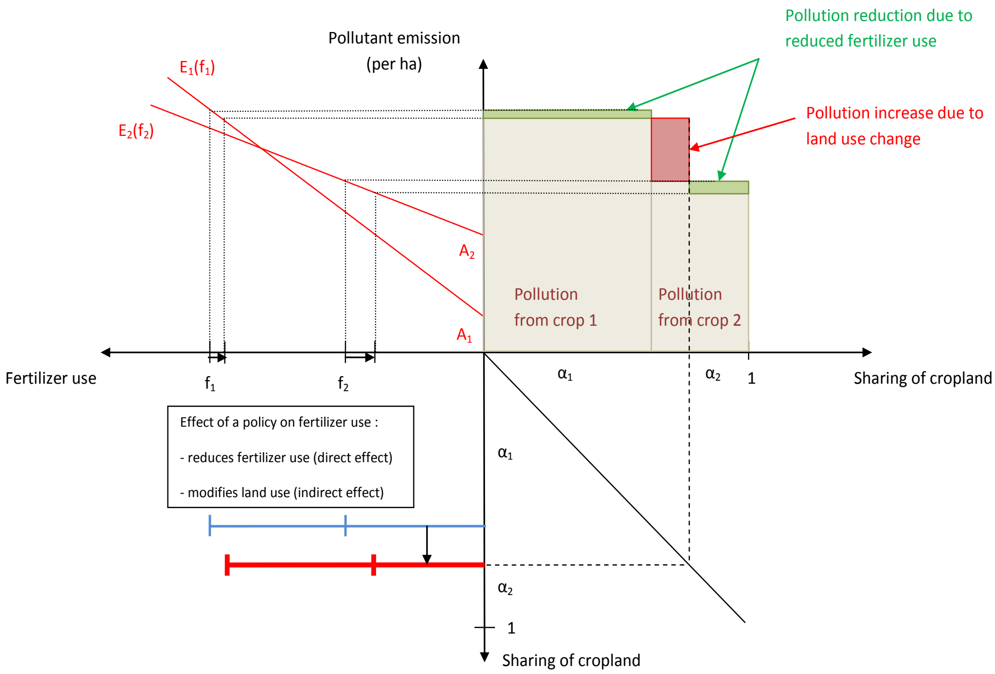


FIGURE 2.5 – emission change level induces by an input taxation for a given q

shall consider a marginal variation of the tax levels, keeping their proportionality. We thus assume that the marginal change of the differentiated taxes (τ_1, τ_2) are related to a marginal change of the pollution tax level t. We consider the change in pollution level when the differentiated input tax on fertilizer increase.

$$
\begin{aligned}
\frac{\partial E}{\partial t} &= \sum_{i=1,2} \frac{\partial E_i}{\partial \tau_i}\frac{\partial \tau_i}{\partial t} = \sum_{i=1,2} B_i \frac{\partial E_i}{\partial \tau_i} \\
&= -\sum_{i=1,2} B_i \Big[e_i(f_i(q; Q(\tau_i))), \alpha_i(q; Q(\boldsymbol{\tau})) * \frac{\partial Q(\boldsymbol{\tau})}{\partial \tau_i} \Big] \\
&\quad + \sum_{i=1,2} B_i \int_{Q(\boldsymbol{\tau})}^{1} \Big[\big(\frac{\partial e_i(f_i(q; Q(\boldsymbol{\tau}))}{\partial \tau_i} \alpha_i(q; Q(\boldsymbol{\tau})) \\
&\quad + \frac{\partial \alpha_i(\boldsymbol{\tau}; q)}{\partial \tau_i} e_i(f_i(q; Q(\boldsymbol{\tau}))) \Big] \phi(q) dq \\
&= \underbrace{-\sum_{i=1,2} B_i \Big[e_i(f_i(q; Q(\boldsymbol{\tau})))\alpha_i(q; Q(\boldsymbol{\tau})) * \frac{\partial Q(\tau_1, \tau_2)}{\partial \tau_i} \Big]}_{\text{Total land effect}} \\
&\quad \underbrace{+ \int_{Q(\boldsymbol{\tau})}^{1} \Big(\sum_{i=1,2} B_i \frac{\partial e_i(f_i(q; Q(\boldsymbol{\tau}))}{\partial \tau_i} \alpha_i(\tau_i; q) \Big) \phi(q) dq}_{\text{Input price effect}} \\
&\quad \underbrace{+ \int_{Q(\boldsymbol{\tau})}^{1} \Big(\sum_{i=1,2} B_i \frac{\partial \alpha_i(\boldsymbol{\tau}; q)}{\partial \tau_i} e_i(f_i(q; Q(\boldsymbol{\tau})\Big)) \phi(q) dq}_{\text{Land-use reallocation effect}}
\end{aligned}
$$

The change in the level of emissions with a heterogeneous taxation is decomposed in three terms. The first term, called total land effect, concerns the reduction of pollution due to the decrease of cropland area. Indeed, the land with the lowest marginal profit becomes unprofitable due to an increase of tax levels, $\frac{\partial Q(\boldsymbol{\tau})}{\partial \tau_i} > 0$.

The second term, called input prices effect, concerns the decrease of pollution due to a reduction of fertilizers use, $\frac{\partial e_i(f_i(q;Q(\boldsymbol{\tau}))}{\partial \tau_i}\alpha_i(\tau_i; q^*)) < 0$.

These two first terms are always negative and contribute to a decrease of emission level. We can also note that they are the only two terms which drives the emission level if we neglect the land-use consideration.

The third term, called land-use reallocation effect, concerns the change in emission levels due to crop reallocation area. As there is no reason that this reallocation is in favor of the least polluting crop, this term is undetermined. If it is positive, the total effect of an input taxation on emission level is ambiguous and can lead to an increase of pollution.

To illustrate this, we rewrite the system of previous equation with the help of the following land constraint :

$$\int_{Q(\boldsymbol{\tau})}^{1} \sum_{i=1,2} \frac{\partial \alpha_i(\tau;q)}{\partial \tau} dq = 0 \Rightarrow \int_{Q(\boldsymbol{\tau})}^{1} \frac{\partial \alpha_2(\tau;q)}{\partial \tau} dq = -\int_{Q(\boldsymbol{\tau})}^{1} \frac{\partial \alpha_1(\tau;q)}{\partial \tau} dq$$

$$\begin{aligned}\frac{\partial E}{\partial t} = \sum_{i=1,2} B_i \frac{\partial E_i}{\partial \tau_i} &= -\sum_{i=1,2} B_i [e_i(f_i(q;Q(\tau_i)))\alpha_i(q;Q(\tau_i)) * \underbrace{\frac{\partial Q(\boldsymbol{\tau})}{\partial \tau_i}}_{\geq 0}] \\ &+ \sum_{i=1,2} B_i \underbrace{\int_{Q^*(\boldsymbol{\tau})}^{1} \Big(\frac{\partial e_i(f_i(q;Q(\tau_i))}{\partial \tau_i} \alpha_i(\tau_i;q)\Big)\phi(q)dq}_{\leq 0} \\ &+ \underbrace{\int_{Q(\boldsymbol{\tau})}^{1} \Big(\frac{\partial \alpha_c(\tau_i;q)}{\partial \tau_i})(B_1 e_1(f_1(\tau_1;q)) - B_2 e_2(f_2(\tau_2;q^*)\Big)\phi(q)dq}_{X}\end{aligned}$$

$$\begin{aligned} &if \quad \tfrac{\pi_{\tau_1}(f(\tau_1),\alpha_1,Q)}{\pi_{\tau_2}(f(\tau_2),\alpha_2;Q)} > \tfrac{\pi(f,\alpha_1,Q)}{\pi(f,\alpha_2,Q)} \Rightarrow \tfrac{\partial \alpha_1}{\partial \tau} > 0 \ \Rightarrow \\ &\begin{cases} \text{if } B_1 e_1(f_1(\tau_1;Q)) > B_2 e_2(f_2(\tau_2;Q)) \Rightarrow X < 0 \\ \text{if } B_1 e_1(f_1(\tau_1;Q)) < B_2 e_2(f_2(\tau_2;Q*)) \Rightarrow X > 0 \end{cases} \end{aligned} \tag{2.27}$$

There are two possible cases :

If $\frac{\partial \alpha_o}{\partial \tau} \geq 0$ and $\frac{\partial \alpha_1}{\partial \tau_1}, \frac{\partial \alpha_2}{\partial \tau_2} \leq 0$, pollution decreases without ambiguity when the taxes levels increase, i.e., $\frac{\partial E}{\partial \tau} \leq 0$. If $\frac{\partial \alpha_o}{\partial \tau}, \frac{\partial \alpha_1}{\partial \tau_1} \geq 0$ and $\frac{\partial \alpha_2}{\partial \tau_2} \leq 0$, the effect of the tax in undetermined.

Therefore there is a paradoxical effect. The pollution may increases whereas the input use decreases if the marginal pollution due to the crop area reallocation is higher than the marginal emission due to both the decrease of cropland area and the decrease of input.

In reason of heterogeneous emission functions, differentiated input taxes cannot restore the optimal crop area allocation.

2.4.3 Objectives

How can we restore the social optimum when emissions depend on crop ? We have shown that a heterogeneous input tax restores the optimum use of fertilizers. The non optimality of crop allocation area is due to the presence of a constant terms, A_i, differentiated by crop in emission functions. One possibility is to subsidy (or tax) the crop area in addition to an input tax. Indeed, a subsidy will not change the first-order condition driving the optimal fertilizer use for each crop on a land of a given quality. Subsidies s_i differentiated by crops could, however, make it possible to equalize $t(A_j - A_i)$ to s_i.

2.5 Controlling non-point source pollutions : A policy mix

2.5.1 First best solution

Additionally to a heterogeneous taxation of input, τ_i, let us introduce a heterogeneous subsidy (or per ha taxation if negative), s_i, differentiated by crops. We denote $\boldsymbol{\tau} \equiv (\tau_1, \tau_2)$ and $\mathbf{s} \equiv (s_1, s_2, s_o)$.

The modified land-owners' optimization program can be written as follows :

$$\begin{aligned}\max_{f,\alpha,Q} \Pi(\boldsymbol{\tau}, \mathbf{s}) &= \int_0^Q [\pi_o(q) + s_o]\phi(q)dq \qquad (2.28)\\ &+ \int_Q^1 \left(\sum_{i=1,2} [p_i y_i(f_i(q), \alpha_i(q); q) - (w + \tau_i) f_i(q) + s_i]\alpha_i(q) \right) \phi(q)dq \ ,\\ &\text{subject to} \quad \sum_{i=c,w} \alpha_i(q) \leq 1 \ \text{ and } \ f_i(q) \geq 0, \ \ \alpha_i(q) \geq 0.\end{aligned}$$

Assuming interior solutions, we can write the first-order conditions of (2.28) as :

$$[p_1 y_1 + p_1 \frac{\partial y_1}{\partial \alpha_1}\alpha_1 - w - (p_2 y_2 + p_2 \frac{\partial y_2}{\partial \alpha_2}\alpha_2 - w)] = -(s_2 - s_1) + \tau_2 f_2 - \tau_1 f_1 \qquad (2.29)$$

$$p_i \frac{\partial y_i}{\partial f_i} - w = \frac{\partial \pi_i}{\partial f_i} = \tau_i \qquad i = 1, 2 \qquad (2.30)$$

From a general point of view, homogeneous input taxation will lead to the optimal emissions if the first order conditions of problem (2.11) and problem (2.28) are equivalent. By comparing eqs. (2.12) and (2.29) on the one hand, and eqs. (2.13) and (2.30) on the other hand, the two optimality conditions coincide if the following system is satisfied :

$$\tau_i = t B_i \ , \qquad i = 1, 2. \qquad (2.31)$$

$$-(s_2 - s_1) + \tau_2 f_2 - \tau_1 f_1 = t(A_2 + B_2 f_2 - A_1 - B_1 f_1) \ . \qquad (2.32)$$

As in the case of a heterogeneous taxation of inputs differentiated by crops, the optimal fertilizer use can be restored by setting $\tau_i = tB_i$. A differentiated subsidy can restore optimal crop allocation by setting $-(s_2 - s_1) = t(A_2 - A_1)$. From a theoretical point of view, a single instrument is sufficient to satisfy this condition. What matters is the difference between subsidies / taxes levels, that must equal the difference between the two social costs of fix pollution emission of the two crops (i.e., the fix emission times the optimal emission tax rate, which is equal to the social marginal cost of pollution). By setting $s_2 = 0$, the condition reduces to $s_1 = t(A_2 - A_1)$. If the natural losses of crop 1 is lower than that of crop 2, s_1 is a subsidy. In the other case, s_1 takes the form of taxation.

The land-use share between perennial and other crops is determined as the equivalent of condition (2.9) :

$$\pi_o(Q^*) + s_o - \left(\sum_{i=1;2} [p_i y_i(f_i(Q^*), \alpha_i(Q^*); Q^*) - (w + \tau_i) f_i(Q^*) + s_i] \alpha_i(Q^*) \right) = 0. \tag{2.33}$$

One can easily verify that, replacing τ_i and s_1 by tBi and $s_1 = t(A_2 - A_1)$, this program is equivalent to :

$$\begin{aligned} \pi_o(Q^*) + s_o \quad &- \quad \alpha_1(Q^*) \left([p_1 y_1(f_1(Q^*), \alpha_1(Q^*); Q^*) - (w + tB_1) f_1(Q^*)] + t(A_2 - A_1) \right) \\ &- \quad \alpha_2(Q^*) \left([p_2 y_2(f_2(Q^*), \alpha_2(Q^*); Q^*) - (w + tB_2) f_2(Q^*)] \right) = 0. \end{aligned}$$

This means that, setting $s_o = tA_2$, one obtains the optimality condition (2.14), and gets the optimal land use, achieving the first best solution.

With this condition, we restore both the optimal crop allocation and the land-use share. However, in the case of n crops, this first-best optimum necessitate the adding $(n-1)$ instruments (in order to restore the optimal land-use allocation)to the $(n-1)$ instruments (in order to restore the optimal fertilizer use). Without consideration of question of feasibility and cost control, it is unrealistic to consider so many instruments.

2.5.2 A second best approach based on two simple instruments

In this subsection, we consider a homogeneous input tax, τ, and a subsidy for the perennial crop, s. Obviously, a subsidy for perennial crops cannot lead to an optimal crop area allocation for all crops. However, a subsidy decreases the overall cropland area, which leads to two effects. Reducing the crop area, the subsidy decreases the emissions due to the crops that which could compensate the increase of emission due to the crop reallocation.

$$\begin{aligned}
\max_{f,\alpha,Q} \Pi(\tau, s) &= \int_0^Q [\pi_o(q) + s]\phi(q)dq \qquad (2.34)\\
&+ \int_Q^1 \sum_{i=1,2} \left([p_i y_i(f_i(q), \alpha_i(q); q) - (w+\tau) f_i(q)]\alpha_i(q)\right)\phi(q)dq \ , \\
s.t. & \sum_{i=1,2} \alpha_i(q) = 1 \ \text{ and } \ f_i(q) \geq 0, \quad \alpha_i(q) \geq 0.
\end{aligned}$$

Assuming interior solutions, we can write the first-order conditions of (2.34) as

$$\begin{aligned}
[p_1 y_1 + p_1 \frac{\partial y_1}{\partial \alpha_1}\alpha_1 - w - (p_2 y_2 + p_2 \frac{\partial y_2}{\partial \alpha_2}\alpha_2 - w)] &= \tau(f_2 - f_1) \ , \qquad (2.35)\\
p_i \frac{\partial y_i}{\partial f_i} - w = \frac{\partial \pi_i}{\partial f_i} &= \tau \qquad i = 1, 2. \qquad (2.36)
\end{aligned}$$

The land-use share between perennial and other crops, $\hat{Q}$ is determined as the equivalent of condition (2.9) :

$$\pi_o(\hat{Q}) + s_o - \left(\sum_{i=1;2} [p_i y_i(f_i(\hat{Q}), \alpha_i(\hat{Q}); \hat{Q}) - (w+\tau) f_i(\hat{Q})] \alpha_i(\hat{Q}) \right) = 0. \quad (2.37)$$

Equation (2.9) and (2.37) are equivalent if and only if :

$$\begin{aligned} & s_o - \left(\sum_{i=1;2} [p_i y_i(f_i(\hat{Q}), \alpha_i(\hat{Q}); \hat{Q}) - (w+\tau) f_i(\hat{Q})] \alpha_i(\hat{Q}) \right) \quad (2.38) \\ = \quad & - \left(\sum_{i=1;2} [p_i y_i(f_i(Q^*), \alpha_i(Q^*); Q^*) - w f_i(Q^*) - tA - tB f_i^*(Q^*)] \alpha_i(Q^*) \right) \end{aligned}$$

On the optimal land use share between perennial and annual crops

From the previous equations, it appears that a subsidy for perennial crop may restore the optimal sharing of land between perennial and annual crops (for some very specific value $\tilde{s}_o = \alpha_1(Q^*) \left[f_1^*(Q^*) \left(\hat{\tau} - tB_1 \right) - tA_1 \right] + \alpha_1(Q^*) \left[f_2^*(Q^*) \left(\hat{\tau} - tB_2 \right) - tA_2$

However, as the homogeneous taxation does not lead to both the fertilizer use and the crop allocation at their optimal value, we can improve the regulation efficiency if $Q(\tau, s_o)$ differs from Q^*.

Indeed, a homogeneous taxation induces a land-use reallocation in favor of one of two crops. If this reallocation is in favor of the most polluting (resp the least), it could be interesting that $Q(\tau, s_o) \geq Q^*$ (resp $Q(\tau, s_o) \leq Q^*$).

This is what we will now determine, by examining the optimal public policies, and the associated welfare analysis, for both first-best and second-best policies.

2.6 Social planner problem and welfare analysis

In this section, we examine the social planner problem of regulating pollution. We assume that the social planner defines the optimal level for policy instruments in a given regulation context, i.e., either in the first-best case of pollution taxation, or in the second-bes cases of indirect regulations (e.g., polluting input taxation). In each case, we assume that the decision-maker takes the best response of private land-owners as given, and defines the level of the instruments that maximize the social welfare.

The total level of emissions is defined as follows : $E = E_1 + E_2 = \int_Q^1 e_1(f_1; q)\alpha_1(q)\phi(q)dq + \int_Q^1 e_2(f_2; q)\alpha_2(q)\phi(q)dq$.
We denote by D the damage function due to total level of emissions E. Finally, we assume the damage is a linear function of emissions : $D(E) = \gamma E$ where γ is the monetary damage parameter. We can thus express the damage as $D(E) = \gamma \sum_i \int_Q^1 e_i(f_1; q)\alpha_i(q)\phi(q)dq$. Then, we can express the welfare as a function of the total private profit generated by the agricultural sector minus the damage associated to agricultural pollution. By assuming that subsidies and tax are lump-sum monetary transfers between the farmers and the rest of the society, we can suppress them from the welfare computation. Note, however, that their level influences optimal private decisions, and that they are parameters of the welfare. The modification of fertilizer use, crop area allocation and land-use share induced by policy instruments are taken into account in the welfare function. We assume a partial equilibrium approach with no price feedbacks from the rest of the economy. Mathematically, we write the

welfare, for given optimal private decisions $(f(q), \alpha(q), Q)$ as follows :

$$W = \int_0^Q \pi_o(q)\phi(q)dq + \int_Q^1 [\sum_{i=c,w}(p_i y_i(f_i(q), \alpha_i(q); q) - wf_i(q) - \gamma e_i(f_i(q); q))]\alpha_i(q)\phi(q)dq \tag{2.39}$$

The objective of this section is to not to compare the welfare under first-best and second-best regulation, as it is obvious that the former is higher than the latter. Our focus is more to examine in which way the second-best optimum deviates from the first-best optimum. In particular, we aim at showing that the introduction of a subsidy for perennial crops leads to a change in level of input taxation, and that input should be less taxed than in the first best case when the regulator is able to influence the cropland area. For this purpose, we compare the first-order condition of social planner objective function with respect to the input taxation when a subsidy for perennial crops is implemented.

2.6.1 Welfare with respect to the tax

We consider the welfare when a homogeneous taxation on fertilizer use, τ and a subsidy for perennial crop, s is implemented

The social planner maximizes the following program :

$$\max_{\tau,s} W(\tau, s) = \int_0^{Q(\tau,s)} \pi_o(q)\phi(q)dq + \int_{Q(\tau,s)}^1 \sum_{i=c,w} \alpha_i(\tau; q)\left[p_i y_i(f_i(\tau; q), \alpha_i(\tau; q); q) - wf_i(\tau; q) - \gamma e_i(f_i(\tau; q); q))\right]\phi(q)dq$$

(2.40)

(2.41)

The first-order conditions with respect to τ and s are :

$$\begin{aligned}\frac{\partial W(\tau,s)}{\partial \tau} &= \pi_o(Q(\tau,s))\frac{\partial Q(\tau,s)}{\partial \tau} \\ &- \sum_{i=c,w}[p_i y_i(f_i(\tau,Q),\alpha_i(\tau;Q);Q(\tau,s)) - wf_i(\tau;Q) - \gamma e_i(f_i(\tau;Q);Q(\tau,s)))]\frac{\partial Q(\tau,s)}{\partial \tau} \\ &+ \int_{Q(\tau,s)}^{1}\sum_i[\frac{\partial\big(p_i y_i(f_i(\tau;q),\alpha_i(\tau;q);q) - wf_i(\tau;q)\big)}{\partial \tau}\alpha_i(\tau;q) - \\ &\frac{\partial \gamma e_i(f_i(\tau;q);q)}{\partial \tau}\alpha_i(\tau;q)]\phi(q)dq \\ &+ \int_{Q(\tau,s)}^{1}\sum_i[\frac{\partial \alpha_i(\tau;q)}{\partial \tau}(p_i y_i(f_i(\tau;q),\alpha_i(\tau;q);q) - wf_i(\tau;q) \\ &-\gamma e_i(f_i(\tau;q);q)]\phi(q)dq = 0\end{aligned} \tag{2.42}$$

$$\begin{aligned}\frac{\partial W(\tau,s)}{\partial s} &= (\pi_o(Q(\tau,s)))\frac{\partial Q(\tau,s)}{\partial s} \\ &- [\sum_{i=c,w}(p_i y_i(f_i(\tau;Q),\alpha_i(\tau;Q);Q(\tau,s)) - \gamma e_i(f_i(\tau;Q);Q(\tau,s)))]\alpha_i(\tau;Q(\tau,s)]\frac{\partial Q(\tau,s)}{\partial s}\end{aligned} \tag{2.43}$$

The two first terms of eq. (2.42) represent the variation of welfare due to a change in the land-use share between perennial and annual crops. The two middle terms are standard conditions and represent the variation of welfare due to a decrease in amount of fertilizer use (variation of welfare due to the intensive margin effect). The last terms of these equations represent the land-use area reallocation due to the input taxation which plays a role when the crop area is a decision variable (variation of welfare due to the extensive margin effect). We see that the first-order condition (2.42) are affected by the subsidy for perennial crops s through the land-use share $(Q(\tau,s) \neq Q(\tau,0))$. This implies that τ is modified by the introduction of subsidy for perennial crops.

2.6.2 Efficiency of second best policies

To sum up, three effects play a role in the determination of the effect of introducing a subsidy to non-polluting perennial crop on the optimal taxation level of the polluting input (i.e., if the tax is increased or decreased with respect to its optimal level without the subsidy).

The potential interest in decreasing the tax is (i) lower when the pollution is highly correlated to input quantity (ii) higher when the pollution emission function has a larger fix factor. Indeed, the damage avoided when Q moves to the right is higher when the natural losses, A_i, are greater. The crop reallocation induced by the tax has an ambiguous effect on the welfare cross-derivative. Indeed, if this reallocation is in favor of the crops having the greatest natural losses, and is lowly correlated to the input, then, it is welfare augmenting to decrease the tax while introducing a subsidy to non-polluting crops. This reduces the negative effect on damage due to the land-use reallocation. Increasing the subsidy compensates the increase of pollution per hectare due to the increase in fertilizer uses. This is due to the fact that the lower B_i, the cheaper it is to compensate an increase of pollution due to fertilizer by a decrease in cropland land area driven by subsidies to perennial crops.

To conclude, depending the type of pollutant considered,[6] a trade-off appears between being close to optimal fertilizer use or being close to the optimal land-use allocation (of both land-use share and crop allocation area).

6. For example, gas pollutant (N_2O) are highly correlated to the quantity of input, and water pollutants ($N0-3$) are mainly related to the fix natural losses

2.7 Conclusion

With the help of a land-use model, we have studied agricultural non point source pollution management. Comparing the first-order conditions of different second-best policies in order to compare their efficiency to the first-best policy, we have shown that an input taxation, differentiated or not by crop, leads to feedback effects on crop area allocation. An input taxation differentiated by crop, as advocated by Helfand et al. (2003), appears to lead to optimal fertilizer use, and restores optimality when there is no land-use feedback effect. However, such land-use changes can lead to a paradoxical effect on pollution and thus, it appears necessary to implement a policy on crop allocation choice. If this policy takes the form of subsidy differentiated by crops, associated to a heterogeneous input taxation differentiated by crops, the first-best optimum can be reached. However, such a policy mix is not realistic, requiring too many instruments and to high control costs. An acceptable second-best solution could be to couple a homogeneous taxation with a subsidy for a perennial crop which is assumed nonpolluting. Such policy could overcome the inefficiency of input taxation highlighted by (Lacroix et al., 2005; Aftab et al., 2010). The introduction of a subsidy modifies the level of optimal input taxation. This deviation depends on types of pollution considered, and there is a trade-off between being close to the optimal land-use or being close to the optimal fertilizer use. To sum up, neglecting the land-use dimension in non-point source pollution management can lead to wrong recommendations for the social planner.

For further research, we can extend this analysis by considering multiple pollutant. Following the kind of pollution, we can anticipate a trade-off between come closer to optimal fertilizer use or optimal crop allocation.

Chapitre 3

Revisited water-oriented relationships between a set of farmers and an aquifer : accounting for lag effect

Ce chapitre est issu d'un working paper en collobaration avec Pierre-Alain Jayet et présenté à Ascona au congrés 2010 sur Sustainable Resource Use and Economic Dynamics (SURED) et à Angers aux 27eme journées de Micro-Appliquée (JMA,2010).

3.1 Introduction

Aquifers constitute about 89 % of the freshwater on our planet, providing most of the drinking water in the world, and they are vulnerable to surface pollution especially from agricultural nitrates (Koundouri, 2004). When nitrates are ingested in too large quantities, they have a toxic effect on human health such as baby-blue syndrome and stomach cancer (Addiscott, 1996). More generally, the nitrates "has the potential to become one of the costliest and the most challenging environmental problem" the environmental agencies face (Stoner, 2011). The U.S. EPA established a maximum contaminant level (MCL) of $10.mg.l^{-1}$ for nitrate in drinking water (U.S. EPA, 1995). Moreover, nitrates also contribute to soil eutrophication. Therefore, to preserve the water quality is an important issue. The Water Framework Directive (2000/60/EC) adopted by the European Commissions requires all ground water bodies to achieve a good status by 2015. This goal includes nitrates limit of $50.mg.l^{-1}$ set by the Nitrate Directive (91/676/EEC). However, this threshold is already exceeded in many groundwater bodies in Europe (Rivett et al., 2008).

Water pollution problem by nitrates from agriculture is a typical case of non-point source (NPS) pollution because individual emissions cannot be measured precisely by the social planner. NPS pollution problems have received considerable attention in the economic literature, mainly to identify the appropriate regulation instruments. These include various mechanism, essentially based on ambient concentration of pollutants (e.g.,Segerson (1988), Xepapadeas (1991)) and on emission proxies like the inputs (e.g., Griffin and Bromley (1982), Shortle and Dunn (1986), Shortle and Abler (1994)). Moreover, managing such pollution is made more difficult by the fact that the social planner is faced with a situation of moral hazard and adverse selection. Firstly, in the case of ambient-pollutant based instrument, it could be prohibitively costly to measure with sufficient precision the farmers' actual efforts in pollution abatement. Indeed, the social planner can only measure ambient pollutant concentration at prespecified 'receptor points'. To eliminate this moral hazard problem, Xepapadeas (1991) proposes a system of subsidies and random penalties in case of nonrespect of desired ambient levels, and Bystrom and Bromley (1998) suggest the use of non-individual contracts and collective penalties. Secondly, the social planner is also faced an adverse selection problem (both in the case of ambient and input based instrument) which may be related to soil spatial heterogeneity. This means that the same management for the same crop in different fields will not necessarily lead to similar nitrate losses (**?**Cabe and Herriges, 1992).

Aquifer polluted by nitrates is similar to a stock pollutant problem which requires to take into account the dynamics of pollutant accumulation in order to regulate efficiently the pollution. However, few studies has considered the dynamic characteristics of this pollutant as noted by Shortle and Horan (2001). Xepapadeas (1992), shows that applying static ambient-incentive schemes in dynamic situations leads to suboptimalities (pollutant overaccumulation), particularly when polluters follow feedback strategies. Then, he proposes schemes which take the form of charges per unit deviation between desired and observed pollutant accumulation paths. However, due to the slow transfer of nitrates through the unsaturated zone of aquifers, this effects are visible 10-60 years after their use (the nitrate transfer velocity varies between 0.60 and $2.50 m.year^{-1}$ (Legout et al., 2007; Gutierrez and Baran, 2009)). Existing lag in nitrate pollution invalidate the incentive schemes proposed by Xepapadeas

(1992). Indeed, the social planner cannot impose penalties today when the pollution is due to fertilizers used many decades ago. In this context, optimal regulation of aquifer polluted by nitrates requires to take into account the lag effect in a dynamic framework.

In this paper, we examine an optimal management problem of a NPS pollution to a pollutant stock which accumulates with a lag. In the economic literature, the delay between agent' action or decision and his consequences has been firstly introduced into the accumulation of capital (Rustichini (1989), Asea and Zak (1999)), and more recently into the pollutant accumulation problem (Brandt-Pollmann et al., 2008; Winkler, 2010). They analyze a *point source* problem in a generic optimal control model devoted to stock accumulation with a fixed delay. We aim to extend this literature to the case of NPS pollution with heterogeneous agents and adverse selection problem. To our knowledge, literature does not deliver how accounting for lag effect in pollution stock can modify the policy of the social planner. *A fortiori* the problem strengthens in case of asymmetric information. To address this issue, we develop an optimal control problem with heterogeneous agents (farmers) and which emissions accumulate with a time lag. The pollution is caused by a continuous set of producers characterized by their individual performance index, the soil quality, and by their individual marginal contribution to the pollution. Following Winkler (2010) who shows that an unique saddle point stationary state exists and that the system dynamics crucially depend on the functional form of the objective function, we assume a separable objective function. This allows us to solve the lag problem in the case of perfect and asymmetric information. We show that the lag acts as increasing the stock and its shadow price at the steady state. It implies that the social planner have to impose more efforts to the farmers while the environmental result is weaker. This effect is augmented when asymmetric information occurs between the farmers and the social planner.

The paper is organized as follows. Section 2 is devoted to the presentation of the basic model. In section 3, we set out the generic control problem with time lagged stock accumulation when the social planner is completely informed about individual farm characteristics. In section 4, we develop the analysis of the optimal control problem when asymmetric information drives the mechanism design which should be implemented by the regulator. Finally in section 5, we illustrate the differences when different time lags are considered. We also compare results in case of perfect information to results in case of asymmetric information.

3.2 Basic elements of the model

Let us consider a set of farmers contributing to the nitrate pollution of an aquifer. Farming activity is represented by the demand of a representative farmer for nitrogen fertilizers which is denoted by x. Activity depends on efficiency index summarized by the one-dimensional θ parameter. The individual farm profit is represented by the function $\pi(x, \theta)$, in which the farm efficiency index θ is spread over the interval $\Theta = [\underline{\theta}, \bar{\theta}]$.The probability distribution function is denoted by $\gamma(\theta)$ and assumed to be strictly positive at any point within the interval :

$$\gamma(\theta) > 0 \quad \forall\theta \tag{H1}$$

The related cumulative function is denoted by $\Gamma(\theta)$. In agriculture, nitrate losses depend on N-fertilizer demand and use at the soil-root zone. The social planner is assumed to know the aquifer characteristics and the transfer process from arable soils to aquifers at the regional scale by the mean of hydrogeological modelling. Asymmetric information comes with wide heterogeneity of soil. The efficiency index, θ, represent the heterogeneity in the soil quality In other words, when asymmetric information on farm characteristics comes in our analysis, we face an adverse selection problem.

The π function is assumed to be twice continuously differentiable. The usual assumption of decreasing returns to scale and the assumed positive marginal profit when x is close to 0 hold here :

$$\pi_{xx} < 0 \tag{H2}$$

$$\pi_x(0, \theta) > 0 \quad \forall \theta \tag{H3}$$

We assume that the marginal profit variation regarding the θ characteristics keeps the same sign. We choose the positive sign, so the marginal profit increases when the θ characteristics increases :

$$\pi_{x\theta} > 0 \tag{H4}$$

Regarding the marginal farm profit and further formal analysis coming in the paper, let us consider the x-variable equation $\pi_x(x, \theta) = c$. Hypotheses (H2) and (H4) lead to characterize the solution $x = \phi(\theta, c)$ as a decreasing function of the unit cost of nitrogen fertilizer, c [1] and an increasing function of the performance index θ :

1. The real important point relies on the constant sign of $\Pi_{x,\theta}$. It has to be noticed that the reverse sign, i.e. ;$\Pi_{x,\theta}$<0, does not change our qualitative results, and only (R4) should be concerned (see section 4)

$$\pi_x(\theta, \phi(\theta, c)) = c \quad \Rightarrow \begin{cases} \qquad H2 & \Rightarrow \phi_c < 0 \\ H2 \ \text{ and } \ H4 & \Rightarrow \phi_\theta > 0 \end{cases} \tag{R1}$$

The demand of nitrogen fertilizer increases when the soil quality increases, and decreases when the unit cost of nitrogen fertilizer increases.

The farming activity is assumed to occur over time. At any time t, the θ-farm use of x leads to an increase in the mean gross profits by $\pi(x(\theta, t), \theta)$ (normalized by prices). Accordingly, the global profit, at time t, is expressed by $\int_\Theta \pi(x(\theta, t), \theta)\gamma(\theta)d\theta$.

Regarding the environmental impact and related damage, we can start by applying a standard framework. The state of our aquifer system is characterized by the nitrate stock per volume unit and denoted by z. The dynamic evolution over time is the result of a double-side effect. On one hand, the clearing effect takes the form of an usual exponential decline characterized by the decline rate τ. On the other hand, the amount of N-fertilizer consumed by the θ farm additively contributes to increase pollution. The marginal contribution related to x depends on θ and the contribution of the θ farm at, the time $t + \beta$, is $a(\theta)\ x(\theta, t)$. β, the lag parameter, represent the delay between the apply of fertilizer and its environmental impact in aquifer. However, a slight difficulty arises when we introduce the lag effect of nitrogen fertilizer use on the nitrate concentration in the aquifer. The time evolution of the environmental system is described by the equation :

$$\dot{z}(t) = -\tau z(t) + \int_\Theta a(\theta)x(\theta, t - \beta)\gamma(\theta)d\theta \tag{3.1}$$

We can note that the parameter $a(\theta)$ relates to the subsoil and to the aquifer more than to the soil per se, and the theta-dependency should be seen as a extension of the model. Expecting that the regulatory body will be asked to design the optimal individual farm demand for the input $x(\theta, t)$ at time 0 for any further time t, we assume that the social planner or environmental regulator integrates knowledge related to the initial state of the aquifer and to the short past farming activity. In addition, we recall that the input has to be non negative. This is expressed by the following assumption [2] :

$$z(0) = z_0 \ ; \ x(\theta, t) = \epsilon(\theta, t) \ \ \forall \theta \in \Theta \ \ \forall t \in [-\beta, 0[\ ; \ x(\theta, t) \geq 0 \ \forall \theta \in \Theta, \forall t \geq 0 \tag{H5}$$

The damage function is expressed by the twice differentiable function depending on z and denoted by $D(z)$. The assumptions related to the damage function are :

$$D_z(0) = 0 \text{ and } D_{zz} > 0 \tag{H6}$$

Let us notice that (H6) leads to $D_z > 0 \ \forall z > 0$.

Finally, the discount rate is denoted by δ, and the marginal cost of public funds is denoted by ρ. This last parameter enters the analysis when contractual incentives are taken into consideration.

The economic analysis that follows is based on a partial equilibrium approach with no price feedbacks from the rest of the economy.

2. It is to be noticed that the relation $x(\theta, t) = \epsilon(\theta, t)$ does not imply that the social planner knows every θ *individually* before time 0. The understanding of the point is especially of interest in the case of asymmetric information between the social planner and the agents. In the rest of the paper, the analytical resolution of the dynamic problems is based on the knowledge of the integral $\int_\Theta a(\theta)x(\theta, t-\beta)\gamma(\theta)d\theta$ for any time t in $[-\beta, 0[$ making possible to weaken H5 by substituting the individual $x()$ for the sum of weighted $x()$.

3.3 Long run optimal trade-off between production and pollution in the case of complete information

When information upon farmers is complete, the social planner's objective is :

$$W = \int_0^{\infty} [\int_{\Theta} \pi(x(\theta,t),\theta)\gamma(\theta)d\theta - D(z(t))]e^{-\delta t}dt \tag{3.2}$$

Accordingly, this programme is expressed below :

$$\max_{x(\theta,t)} W \quad \text{subject to (6.1)}, \ (H5) \tag{3.3}$$

Differently from the usual optimal control programme, the lag term appearing in the state dynamics (6.1) does not allow us to directly apply the Pontryagin theorem. The solution arises when we consider the transformation of the command variable $y(\theta,t) = x(\theta,t-\beta)$. The objective function and the state evolution equation are transformed as follows :

$$W = -\int_{-\beta}^{0}\int_{\Theta} \pi(x(\theta,t),\theta)\gamma(\theta)d\theta e^{-\delta t}dt + \int_0^{\infty} [e^{\delta\beta}\int_{\Theta} \pi(y(\theta,t),\theta)\gamma(\theta)d\theta - D(z(t))]e^{-\delta t}dt \tag{3.4}$$

$$\dot{z}(t) = -\tau z(t) + \int_{\Theta} a(\theta)y(\theta,t)\gamma(\theta)d\theta \tag{3.5}$$

Thanks to the (H5) assumption, the first integral component of this last W expression can be taken out of the programme. Aiming at the use of the maximum principle, we define the current-value Hamiltonian in which the shadow price of the pollution stock is denoted $\lambda(t)$ and is designed to take a positive value :

$$H^c = e^{\delta\beta} \int_\Theta \pi(y(\theta,t),\theta)\gamma(\theta)d\theta - D(z(t)) - \lambda(t)[\int_\Theta a(\theta)y(\theta,t)\gamma(\theta)d\theta - \tau z(t)] \tag{3.6}$$

According to our technical assumptions, the Pontryagin theorem delivers the conditions holding the optimal solution : $\{y^*(\theta,t), z^*(t), \lambda^*(t)\}$:

$$y^*(\theta,t) \text{ maximizes } H^c(y, z^*, \lambda^*) \tag{3.7}$$

$$\dot{\lambda}^*(t) - \tau\lambda^*(t) = H^c_z(y^*, z^*, \lambda^*) \tag{3.8}$$

Our "convex" problem leads to the following equations : [3]

$$\pi_x(y^*(\theta,t),\theta) = a(\theta)\lambda^*(t)e^{-\delta\beta} \tag{3.9}$$

$$\dot{\lambda}^*(t) - (\tau+\delta)\lambda^*(t) = D_z(z^*(t)) \tag{3.10}$$

The transversality condition satisfied :

$$\lim_{t\to\infty} \lambda(t)e^{-\delta t}z(t) = 0 \tag{3.11}$$

Condition (3.9) expresses that the θ farmer's profit provided by one additional unit of polluting input equals the discounted cost of the related marginal pollution evaluated at time $t+\beta$ and weighted by the individual polluting contribution $a(\theta)$. The solution in y to this equation is obtained from the expression (R1) $y^*(\theta,t) = \phi(\theta, a(\theta)\lambda(t)e^{-\delta\beta})$. The complete solution of the Regu-

3. Equation (9) relies on the Euler-Lagrange relation(see Seierstad and Sydsaeter, 1987, pp 16-19)

lator's programme is provided by the implicit relation between the command x and the shadow price λ, and by the two-dimension differential system, as summarized by the equation set (R2) :

$$\begin{aligned}
&\forall\theta,\ \forall t>0\ :\ x^*(\theta,t)=\phi(\theta,a(\theta)\lambda(t+\beta)e^{-\delta\beta})\\
&\dot{z}^*(t)=-\tau z^*(t)+\int_\Theta a(\theta)x^*(\theta,t-\beta)\gamma(\theta)d\theta\\
&\dot{\lambda}^*(t)-(\tau+\delta)\lambda^*(t)=-D_z(z^*(t))\\
&(H5)\quad ;\text{ the tranversality condition satisfied}
\end{aligned}\tag{R2}$$

There is only one steady state related to this system (proof in appendix 3.7.1). The technical assumptions described above lead to also deliver a graphics describing the paths related to this differential system.

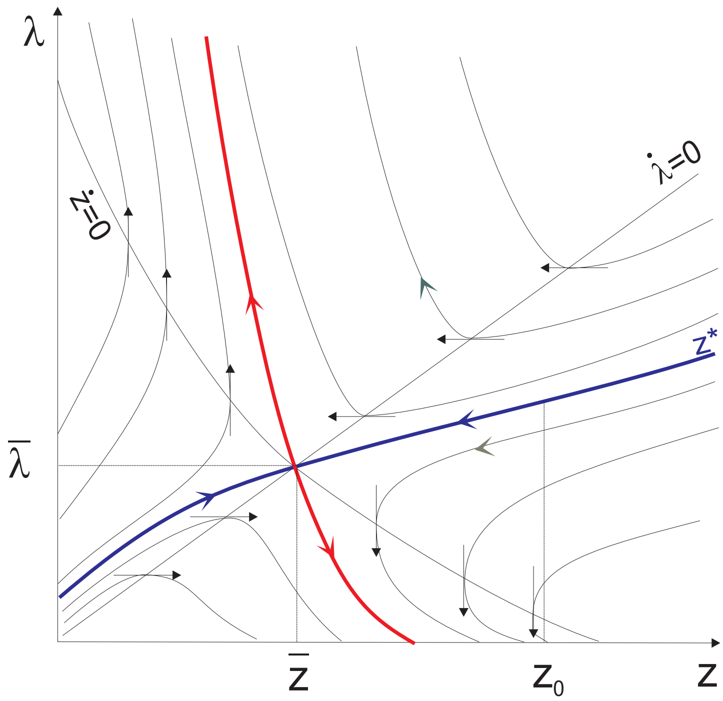


FIGURE 3.1 – Phase diagram describing the paths linking the pollution state z and its shadow price λ.

Let us focus on the steady state $(\bar{z},\bar{\lambda})$ defined by $\{\dot{y}=0\,;\ \dot{z}=0\}$. We are interested by the impacts of the parameters β, δ, τ on the steady state, leading us to summarize the results in propositions 3.3.1, 3.3.2 and 3.3.3.

Proposition 3.3.1 *At the steady-state, the higher the delay between nitrogen-fertilizer application and its environmental impact, the greater the pollution level and the shadow price.*

The two next propositions are expected and refer to more usual approaches. It is of interest to keep in mind the main parameter sensitiveness.

Proposition 3.3.2 *When the discount rate increases, the steady state pollution level and the steady state shadow price both increase.*

Proposition 3.3.3 *When the decline rate increases, i.e more nitrates are absorbed by the aquifer, the steady state pollution level and the steady state shadow price both increase.*

Proofs are delivered in appendix 3.7.2.

Having in mind the contractual approach which supports the analysis of the asymmetric information problem (see the section 3.4), we introduce the Regulator's choice in supplying contracts to any θ farm. A contract is characterized by a two dimensions function $(q(\theta, t), s(\theta, t))$ in which q refers to the upper limit of x-use of polluting input and s refers to the lump sum transfer as the counterpart of profit decrease. Contracts are designed to be freely accepted by the farms, consequently the Regulator has to prevent farmers from refusing the contracts when their participation is expected as socially beneficial.

The transfers call for costly public funds (i.e. one budget unit costs $1+\rho$) and the social objective is now expressed like :

$$W = \int_0^{\infty} \{ \int_{\Theta} [\pi(q(\theta,t),\theta) - \rho s(\theta,t)]\gamma(\theta) d\theta - D(z(t)) \} e^{-\delta t} dt \qquad (3.12)$$

In the complete information case, there is no place for informational rent. The reservation utility of the θ farm is the unconstrained profit characterized by the q-consumption equal to $\phi(\theta,0)$ (constant along time). When public funds are costly the individual discounted transfer is equal to the individual profit variation :

$$\int_0^{\infty} s(\theta,t) e^{-\delta t} dt = \int_0^{\infty} [\pi(\phi(\theta,0),\theta) - \pi(q(\theta,t),\theta)] e^{-\delta t} dt \qquad (3.13)$$

The public objective can be rewritten by substitution of the transfer expressed above, so that the Regulator's programme is now :

$$\max_{q(.,.)} W = \int_0^{\infty} \{ \int_{\Theta} [(1+\rho)\pi(q(\theta,t),\theta) - \rho \pi(\phi(\theta,0),\theta)]\gamma(\theta) d\theta - D(z(t)) \} e^{-\delta t} dt \qquad (3.14)$$

arising with the unchanged dynamics of the state variable (still delivered by equation 3.5). The implicit solution of this programme is still provided through the change in the control variable with respect to the time lag parameter β. The contract (for any θ at any time for the quota q, and under an integral equation for any θ-transfer s) and the (z,λ) path are completely characterized by the system (R3) :

$$\begin{aligned}
&\forall\theta,\ \forall t>0\ :\ q^*(\theta,t)=\phi(\theta,\tfrac{a(\theta)\lambda^*(t+\beta)e^{-\delta\beta}}{1+\rho})\\
&\forall\theta\ :\ \int_0^\infty s^*(\theta,t)e^{-\delta t}dt=\int_0^\infty[\pi(\phi(\theta,0),\theta)-\pi(q^*(\theta,t),\theta)]e^{-\delta t}dt\\
&\dot{z}^*(t)=-\tau z^*(t)+\int_\Theta a(\theta)q^*(\theta,t-\beta)\gamma(\theta)d\theta\\
&\dot{\lambda}^*(t)-(\tau+\delta)\lambda^*(t)=-D_z(z^*(t))\\
&(H5)\quad \text{; the tranversality condition satisfied}
\end{aligned} \tag{R3}$$

When the parameter related to the shadow cost of public funds tends toward 0 (i.e. $\rho \to 0$), the system (R3) tends toward the system (R2). The non-costly transfers do not affect the solution (q, z, λ).
The parameters θ, λ and δ have similar effects on the steady state as mentionned in the R2-analysis. Proposition 3.3.4 delivers the qualitative impact of the cost of public funds on the steady state (proof in appendix 3.7.2).

Proposition 3.3.4 *When the marginal cost of public funds increases, the pollution level and the shadow price in the steady state increase.*

3.4 The dynamic problem in the case of asymmetric information

In the case of asymmetric information. the social planner has no individual information on any θ farm, but he knows the statistical distribution of θ. In others words, the social planner is unable to assess the nitrate losses related to each farm. But he is assumed to know the distribution of the information characteristics. We place our adverse selection problem in the framework of the incentive theory developed by Laffont and Tirole (1993) among others.

We consider that the social planner offers a menu of contracts to any farm, and either the farmer θ selects one of the contracts or he refuses all of them. The problem of the social planner is to design the optimal menu regarding the social objective including the farm profits, the environmental damage, and the regulation costs.

The menu of contracts is a two dimension function $(q(\theta,t), s(\theta,t))$. Like in the previous complete information context, q denotes the "quota" and s denotes the "subsidy". Formally the regulator acts as asking any farmer at time 0 for contracting or not, and for the characteristics of his θ farm in the case of acceptance. The participating farmer selects a contract through the report $\tilde{\theta}$. The acceptance by the farmer implies that he complies at time 0 with the upper bound $q(\tilde{\theta},t)$ holding the q-input at any time t. He will receive the transfer $s(\tilde{\theta},t)$.

The θ farmer's programme is to declare his optimal report. Based on the revelation principle, the menu proposed by the social planner is a mechanism designed in such a way that the θ farmer's dominant strategy is to report his true characteristics θ. Theoretically the social planner keeps the possibility to design the menu in such a way that the optimal set of participating farmers is a subset of Θ. This opportunity is explored in some papers devoted to applications of the incentive theory (see Bourgeon et al. (1995)). For simplicity, we do not keep here this opportunity, even if the menu is possibly suboptimal.

Formally we consider that the functions q and s have the requested mathematical properties allowing us to use derivatives as long as necessary. The first step of the analysis leads to characterize the incentive-compatibility constraints and the participation constraint (the so-called rationality constraint). The starting point is the following θ farmer's programme which defines the farmer's optimal report :

$$\max_{\tilde{\theta}} \int_0^{\infty} [\pi(q(\tilde{\theta},t),\theta) + s(\tilde{\theta},t)]e^{-\delta t}dt \tag{3.15}$$

Let us notice that the private discount rate is supposed to be equal to the public one δ. Solving this programme using first and second order conditions, and with the help of the revelation principle, we derive incentives constraints summarized by the relations IC1 and IC2 [4]

$$\textstyle\int_0^{\infty}[\pi_x(q,\theta)\frac{\partial q}{\partial \theta} + \frac{\partial s}{\partial \theta}]e^{-\delta t}dt = 0 \tag{IC1}$$

$$\textstyle\int_0^{\infty} \pi_{x\theta}(q,\theta)\frac{\partial q}{\partial \theta}e^{-\delta t}dt > 0 \tag{IC2}$$

The contract is supposed to be freely accepted by the θ farmer. When the regulator aims at leading the farmer to accept the contract, he has to ensure that the farmer does not lose with the contract acceptance. The “reservation profit” of the θ farm is expressed by $\pi(\phi(\theta,0),\theta)$ (which has a constant current value along time). That leads to define the information rent $R(\theta)$ which has to be non negative as following :

$$R(\theta) = \int_0^{\infty} [\pi(q(\theta,t),\theta) + s(\theta,t) - \pi(\phi(\theta,0),\theta)]e^{-\delta t}dt \geq 0$$

Assumption H4 leads to demonstrate that the rent decreases when the θ type increases. Considering that a contract has to be accepted by any θ in Θ, we can write the rationality constraint under the form (IR1) in which only the

4. Usually in the field of mechanism design, the IC2 condition is firstly supposed to hold, and has to be checked regarding the resulting contract (i.e. ; R4). Third-derivatives condition should be added to ensure that the contract R4 complies with IC2.

upper type $\bar{\theta}$ matters :

$$R(\bar{\theta}) = \int_0^{\infty} [\pi(q(\bar{\theta},t),\bar{\theta}) + s(\bar{\theta},t) - \pi(\phi(\bar{\theta},0),\bar{\theta})]e^{-\delta t}dt \geq 0 \qquad \text{(IR1)}$$

Let us consider the social welfare function W which is now :

$$W = \int_0^{\infty} \left\{ \int_{\underline{\theta}}^{\bar{\theta}} [\pi(q(\theta,t),\theta) - \rho s(\theta,t),\theta)]\, \gamma(\theta)d\theta - D(z(t)) \right\} e^{-\delta t}dt \geq 0$$

The subsidy term is easily replaced with the help of the first order incentive condition (IC1) and integration by parts :

$$\int_0^{\infty} \int_{\underline{\theta}}^{\bar{\theta}} s(\theta,t)\gamma(\theta)d\theta e^{-\delta t}dt = \int_0^{\infty} s(\bar{\theta},t)e^{-\delta t}dt + \int_0^{\infty} \int_{\underline{\theta}}^{\bar{\theta}} \pi_x(q,t)\frac{\partial q}{\partial \theta}\Gamma(\theta)d\theta e^{-\delta t}dt$$

Like in the previous sections of the paper, regarding the state dynamic equation (6.1) which calls for the time lag command variable, we choose to replace the command $q(\theta,t)$ by the variable $r(\theta,t) = q(\theta,t-\beta)$ in the function W :

$$\begin{aligned} W \;=\; & \int_0^{\infty} \left\{ e^{\delta\beta} \int_{\underline{\theta}}^{\bar{\theta}} \left[\pi(r(\theta,t),\theta)\gamma(\theta) - \rho\pi_x(r(\theta,t),\theta)\frac{\partial r}{\partial \theta}(r(\theta,t),\theta)\Gamma(\theta) \right] d\theta - D(z(t)) \right\} e^{-\delta t}dt \\ & -\rho \int_0^{\infty} s(\bar{\theta},t)e^{-\delta t}dt - \int_0^{\beta} \int_{\underline{\theta}}^{\bar{\theta}} \left[\pi(r(\theta,t),\theta)\gamma(\theta) - \rho\pi_x(r(\theta,t),\theta)\frac{\partial r}{\partial \theta}(r(\theta,t),\theta)\Gamma(\theta) \right] d\theta \end{aligned}$$

In the second line of this expression, the first negative term related to the $\bar{\theta}$ subsidy weighted by ρ should be as small as possible. The rationality constraint (IR1) leads to design the $\bar{\theta}$ contract in such a way that the rent $R(\bar{\theta})$ is equal to zero. The second term of the second line is explicitly computed thanks to the (H5) hypothesis ($r(\theta,t) = q(\theta,t-\beta) = \epsilon(\theta,t-\beta)$, $\forall\ t \in [0,\beta]$).

The optimal control problem of the regulator can be limited to the first line of the expression above, so that the current hamiltonian function related to the problem is :

$$H^c = e^{\delta\beta}\int_{\underline{\theta}}^{\bar{\theta}} \left[\pi(r,\theta)\gamma - \rho\pi_x(r,\theta)r_\theta\Gamma\right]d\theta - D(z) - \lambda\left[-\tau z + \int_{\underline{\theta}}^{\bar{\theta}} ar\gamma d\theta\right]$$

The Pontryagin theorem leads to maximize the H^c function according to the command r. Regarding the integral form of the hamiltonian as a function of r and r_θ, the problem can be solved by the Euler relation ($\frac{\partial H^c}{\partial r} = \frac{d}{d\theta}\frac{\partial H^c}{\partial r_\theta}$). Finally, the characterization of the full menu of contracts, the dynamic equations describing the evolution of the state z and the shadow price λ is summarized by the system R4 [5].

$$\begin{array}{l}
\forall\theta,\ \forall t>0\ :\ q^*(\theta,t) = \phi(\theta, \frac{a(\theta)\lambda^*(t+\beta)e^{-\delta\beta}}{1+\rho} - \frac{\rho}{1+\rho}\pi_{x\theta}(q^*,\theta)\frac{\Gamma(\theta)}{\gamma(\theta)}) \\
S(\bar{\theta}) = \int_0^\infty s(\bar{\theta},t)e^{-\delta t}dt = \int_0^\infty [\pi(\phi(\bar{\theta},0),\bar{\theta}) - \pi(q(\bar{\theta},t),\bar{\theta})]e^{-\delta t}dt \\
\forall\theta\ :\ \int_0^\infty s(\theta,t)e^{-\delta t}dt = S(\bar{\theta}) + \int_0^\infty \int_\theta^{\bar{\theta}} \pi_x(q(u,t),u)\frac{\partial q}{\partial\theta}(u,t)du e^{-\delta t}dt \\
\dot{z}^*(t) = -\tau z^*(t) + \int_\Theta a(\theta)q^*(\theta,t-\beta)\gamma(\theta)d\theta \\
\dot{\lambda}^*(t) - (\tau+\delta)\lambda^*(t) = -D_z(z^*(t)) \\
(H5)\quad ;\ \text{the tranversality condition satisfied}
\end{array} \tag{R4}$$

We assume that added technical conditions referring to the (IC2) conditions hold and allow us to consider that the necessary conditions delivered by the system (R4) describe the optimal solution. The optimal menu of contracts leads the regulator to design the quota q for any θ at any time t. The subsidy appears through an integral condition.

5. Regarding the hypotheses (H4), it has to be noted that the change of sign $\Pi_{x,\theta}$ leads to change $\Gamma(\theta)$ in $-(1-\Gamma(\theta))$ .In addition, $\bar{\theta}$ is change with $\underline{\theta}$ when the sign of $\pi_{x,\theta}$ is negative (information rent increasing). We demonstrate that qualitative results hold whatever sign of the sign of $\Pi_{x,\theta}$

Compared to the system R3, the steady state related to the system R4 lets an additional negative term appearing in the expression of the optimal quota ($q^* = \phi(\theta, \frac{a\lambda^* e^{-\delta\beta}}{1+\rho} - \frac{\rho}{1+\rho}\pi_{x\theta}(q^*, \theta)\frac{\Gamma}{\gamma})$. This additional term does not allow us to deliver a general result in term of lag effect. The sign of third derivatives enters the conditions which lead to the proposition 3.4.2. Moreover, this sign plays a crucial role in the comparison between system R3 and system R4 (proposition 3.4.1).

Proposition 3.4.1 *In case of asymmetric information (R4), the level of pollution stock, the shadow price and the total amount of instantaneous polluting input at the steady state are higher than in case of perfect information (R3) when the third-derivative,* $\Pi_{xx\theta}$*, is negative. Otherwise, the effects are ambiguous.*

Proof is delivered in appendix 3.7.3.

Proposition 3.4.2 *When the delay between the spreading of $N-$fertilizer on the farm and the impact of it is increased, i.e., the higher the lag, the greater the increase in the pollution level and the higher the shadow price in the steady state, if the third-derivative* $\Pi_{xx\theta}$ *is negative. Otherwise, the effects are ambiguous.*

In other words, when the complementary between soil quality and fertilizers decreases with respect to fertilizers, $\Pi_{xx\theta}<0$, we find the same results that in the case of perfect information. Otherwise, when the complementary between production factors increases ($\Pi_{xx\theta}>0$) with respect to the fertilizers, the effects on shadow price and pollution level are ambiguous.
Proof is delivered in appendix 3.7.4.

These results imply that the lag effect would differ in the case of asymmetric information compared to the complete information when the derivative, $\Pi_{xx\theta}$, is non negative for all θ

3.5 Discussion and perspective

To open the discussion, in this section we present a numerical application of our analytical approach. Numerical simulations are based on the following added elements, i.e. the specification of the damage function, the specification of the profit function, the specification of the density function, and a set of values for parameters. The damage takes an usual quadratic form :

$$D(z) = \frac{k}{2} z^2, k > 0 \tag{3.16}$$

The profit function is normalized by prices and takes a form in accordance with usual *Nitrogen*-yield functions suitable for numerous crops :

$$\Pi(x, \theta) = 1 - e^{-\theta x} - x \text{ with } \theta \in |1, e] \tag{3.17}$$

The function $1 - e^{-\theta x}$ refers to a yield function based on agronomic observations. We note that the third-derivative $\Pi_{xx\theta}$ is negative. Regarding the input and the output in our analysis, we consider the less performing farm such that $\bar{\theta} = 1$. The best performing θ consistent with the hypothesis H4 is e. The contribution of farmers to a stock of pollution is considered here not $\theta-$dependent ($a(\theta) = a$ for any θ). We assume that the density function follows a uniform distribution. As for the exogenous parameters, the selected values of a, k and ρ aim at clearly illustrating the different effects (noting that ρ is in line with previous analytical analyses, for example the value proposed by Laffont and Tirole (1993)). The value of the discount rate, δ, suits the one recommended by regulatory bodies (Lebègue et al. (2005)). According to hydro-geologists, a minimum of 10 to 60 years, depending on the aquifer, is necessary for N-fertilizer to leach into the groundwater (**?**Gutierrez and Baran, 2009). We set an intermediate value, $\beta = 30$ years, by default. It is to be noted that the US Ogallala aquifer (covering 8 States of the U.S. and providing 80% of the drinking water of people living within the aquifer boundary) may enter this category of groundwater, regarding the usual values of transfer velocity and depth of water. Finally, aquifers need up to several decades to eliminate traces of N-fertilizers. We thus deduce the decline rate, $\tau = 0.02$. Table 3.5 resumes the values of parameters.

After optimization and solving in perfect information, we obtain the phase

parameters	a	k	$1+\rho$	δ	β	τ
values	10^{-3}	10^{-5}	1.3	0.04	30	0.02
units	dam^{-3}	$eurokgN^{-2}.dam^{6}.ha^{-1}.year^{-1}$		$year^{-1}$	an	$year^{-1}$

diagram illustrated by figure 3.2 in line with figure 1, and providing figures in addition. [6]

Figure 3.3 illustrates the fact that the lag effect, when we focus on the steady state and on the optimal path for three values of β, including the case $\beta = 0$ (i.e. no lag) and the two others lags, respectively $\beta = 15$ and $\beta = 30$ (years). In our example the introduction of a time lag of 15 years increases the pollution stock by fifty percent in the steady state. A lag time of 30 years would double the pollution stock. Meanwhile the higher is the lag time, significantly

6. All computations and related graphs are obtained by the use of Mathematica-7.

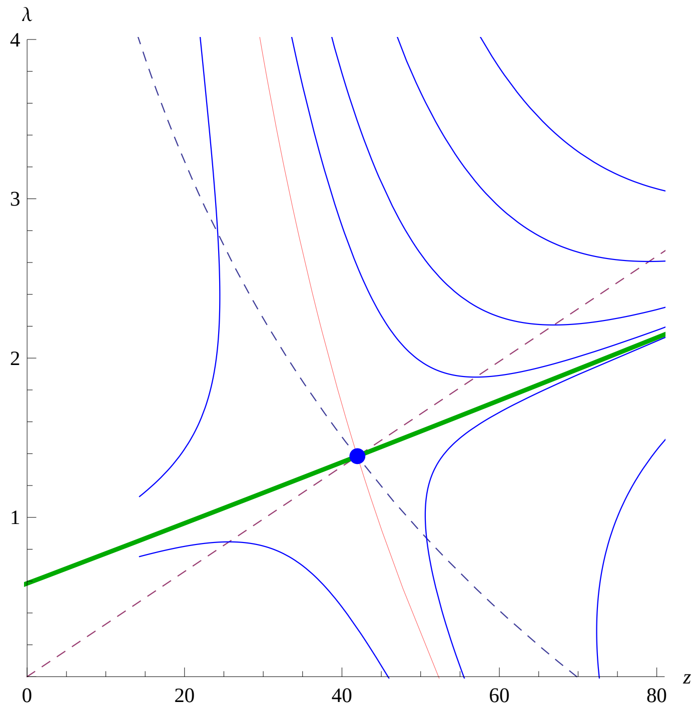


FIGURE 3.2 – Numerical phase diagram in the case of perfect information describing the paths linking the pollution state z and its shadow price λ.The dashed curves represent the set of points for which time-derivatives (respectively z and λ) are equal to 0. The two other curves (green and red) passing through the steady state describe respectively the convergent (green) and divergent (red) paths. The optimal path is the green one.

the higher is the shadow price of the pollution.

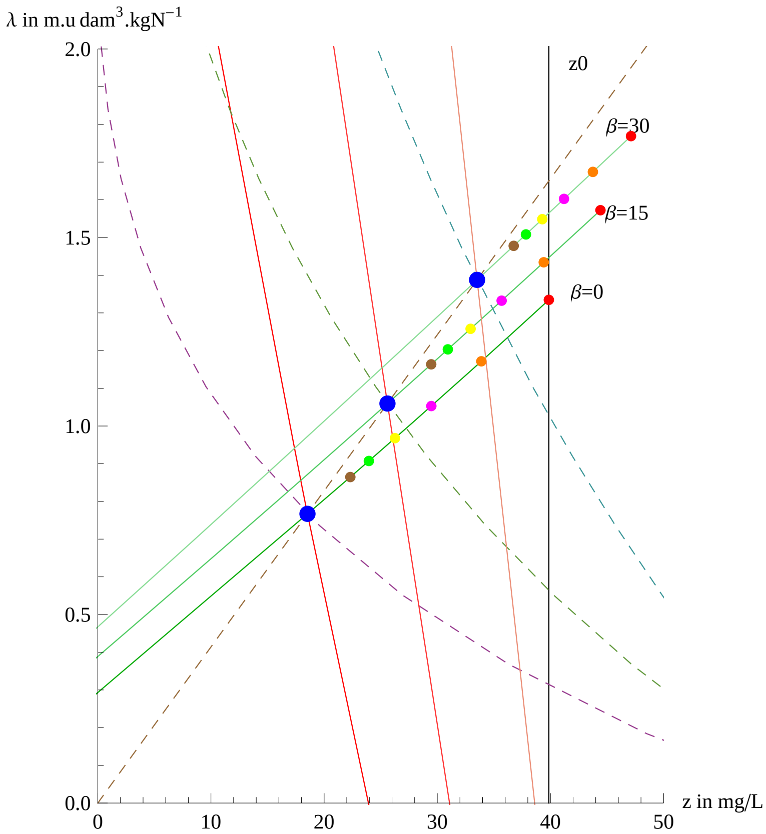


FIGURE 3.3 – Comparison of both steady state and optimal dynamics regarding the pollution state z and its shadow price λ in case of perfect information, for different values of the time lag β (respectively 0, 15, 30 years). Green paths starting from the initial state $z0 + \beta$ (i.e. the red point on the right) match colored points related to ten-year steps and tend to the steady state at time ∞ (the big blue circle on the figure)..

An illustration of the lag influence on the dynamics regarding the pollution stock z is displayed on figure 3.4. The lag obviously does not only impact the steady state. The pollution stock goes on increasing during the time interval $]0, \beta]$. In other words the time lag modifies all the dynamics. 3.5 illustrates the fact that an optimal management of pollution which takes into account the lag implies more efforts (λ higher) for the farmers while the environmental results will be weaker.

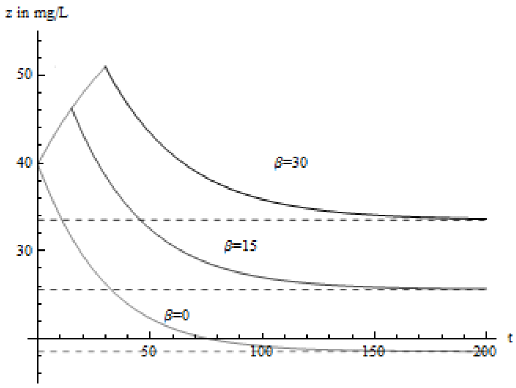


FIGURE 3.4 – The dynamics of the pollution stock when $\beta = 0$, $\beta = 15$ and when $\beta = 30$.

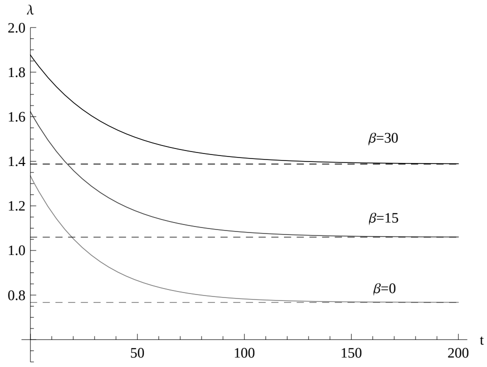


FIGURE 3.5 – The dynamics of the shadow price when $\beta = 0$, $\beta = 15$ and when $\beta = 30$

.

Asymmetric information implies a cost on the regulatory body side through the informational rent paid to the farmers. The production allowed to each farmer is higher than in the case of perfect information. However, some farmers can do no better than not to produce and therefore they receive a subsidy as a compensation for the income loss. The global effect on pollution stock is ambiguous (see proposition 3.4.2). Regarding our profit function and its negative third-derivative $\Pi_{xx\theta}$, the level of the pollution stock increases when we move from perfect information toward asymmetric information. The time lag effect is amplified in case of asymmetric information. Figure 3.6 illustrates both the steady state in perfect and asymmetric information for different values of the time lag and for different values of the opportunity cost of public funds. Even when asymmetric information leads to increase the stock pollution and the shadow price in the steady state, its impact appears as less important than the time lag impact.

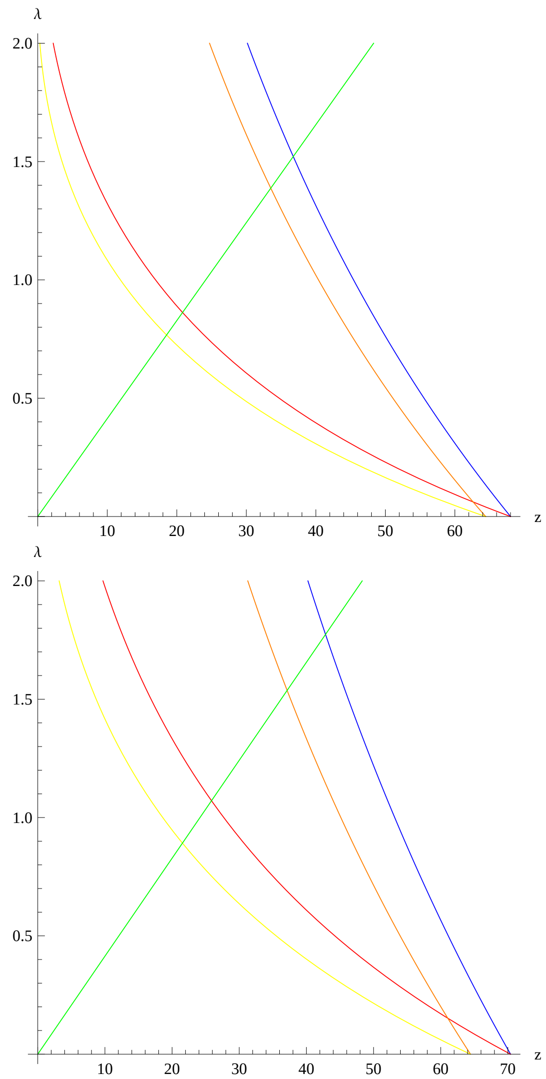


FIGURE 3.6 – Impacts of the time lag β and of the opportunity cost of public funds ρ on the steady state, given perfect and asymmetric information : the steady state results of matching the green curve (i.e. the optimal path) and the yellow, purple, orange and blue curves which respectively relate to $\beta = 0$ and in the case of perfect and asymetric information, and $\beta = 30$ in the case of perfect and asymmetric information, when $\rho = 0.3$ on the left and $\rho = 0.7$ on the right.

3.6 Conclusion

We have developed a dynamic economic framework to assess the impacts of lag time on the NPS optimal management . We analyzed this impact with soil heterogeneity and adverse selection problem. The solution takes the form on individual contracts between the social planner and the farmers.

We have shown that that the shadow price and the stock of pollutant at the steady state increase with the lag. This result is important for the design of optimal policy by the social planner. Indeed, an optimal management of pollution which takes into account the lag implies more efforts for the farmers while the environmental results will be weaker. In the case of asymmetric information, only a profit function with a third derivative positive could lead to an unambiguous results. However, for the standard function (e.g ; quadratic, Mitscherlich) used to represent the agricultural activities, the asymmetric information strengthens those obtained in perfect information.

The simulated results give an idea to the amplitude of lag effect. For example, the introduction of a time lag of 15 years increases the pollution stock by fifty percent in the steady state. And a lag time of 30 years would double the pollution stock. These results show that the lag time leads the social planner to shift from the optimal policy more than he would do when neglecting information asymmetry. The regulation policy, viewed through a menu of contracts in our analysis, is appropriate to deal with the heterogeneity of agents polluting an aquifer and with the diversity of aquifers regarding the lag time. Even if the problem of ex post control makes difficult any incentives mechanism per se, implementation is promoted by existing specifications related to public policies. In the European Union, farmers have to meet specified information when they benefit by CAP support. Thanks to the wide range of real lag time, the policy has nevertheless to be adapted to each aquifer. Even in the case of homogenous marginal contribution of pollution (i.e. the a parameter) and in the case of weak oppotunity cost of public funds (equivalent to the case of complete information when $\rho = 0$), the lag time is one of the major physical drivers of the environmental policy, leading the water body to adapt to the aquifer and not only to the river basin.

For further research, added simulations based on more realistic parameters regarding the crop system and the hydrological system should be carried out at the appropriate scale, given the USA and European Union policy contexts and directives focusing on nitrate pollution and water quality.

APPENDIX

3.7 Annexe

3.7.1 Uniqueness of the steady state (R2)

Let us consider the set of points $\{(z(t), \lambda(t));\ \dot{z} = 0\}$. The relation $\dot{z} = 0$ is equivalently verified when $\tau z = \int_\Theta a(\theta)\phi(\theta, a(\theta)\lambda^\gamma(\theta)d\theta)$. This relation and the relation R1 imply that z decreases when λ increases. The considered set of points is a continuous curve of positive value points meeting the z-axis and the λ-axis.

Let us consider now the set of points $\{(z(t), \lambda(t));\ \dot{\lambda} = 0\}$, i.e. $(\delta + \tau)\lambda = D_z e^{-\delta\beta}$. The assumption (H6) immediately leads to an increasing (in z and λ) curve starting from 0 on the $\{z, \lambda\}$ plane.

There is only one crossing point belonging to the two previous sets.

For further demonstration, any x variable at the steady state is denoted by $\bar{x}$.

3.7.2 Propositions 3.3.1, 3.3.2, 3.3.3, 3.3.4

For propositions 3.3.1, 3.3.2 and 3.3.3 we start from the R2-system at the steady state :

$$\begin{cases} \bar{x_2}(\theta) = \phi(\theta, a(\theta)\bar{\lambda_2}e^{-\delta\beta}) \\ \tau\bar{z_2} = \int_\theta a(\theta)\bar{x_2}\gamma(\theta)d\theta \\ (\tau+\delta)\bar{\lambda_2} = D_z(\bar{z_2}) \end{cases}$$

Influence of β

We differentiate the previous system with respect to β.

$$\begin{cases} \frac{\partial\bar{x_2}}{\partial\beta} = \phi_c(-\delta e^{-\delta\beta}a(\theta)\bar{\lambda_2} + a(\theta)\frac{\partial\bar{\lambda_2}}{\partial\beta}e^{-\delta\beta}) & \text{(a1)} \\ \tau\frac{\partial\bar{z_2}}{\partial\beta} = \int_\theta a(\theta)\gamma(\theta)\frac{\partial\bar{x_2}}{\partial\beta}d\theta & \text{(a2)} \\ (\tau+\delta)\frac{\partial\bar{\lambda_2}}{\partial\beta} = D_{zz}(z_2)\frac{\partial\bar{z_2}}{\partial\beta} & \text{(a3)} \end{cases}$$

Solving (a3) for $\frac{\partial\bar{\lambda_2}}{\partial\beta}$ and substituting into (a1), we get :

$$\frac{\partial\bar{x_2}}{\partial\beta} = \phi_c e^{-\delta\beta}(-\delta a(\theta)\bar{\lambda_2} + a(\theta)\frac{D_{zz}(\bar{z_2})}{\tau+\delta}\frac{\partial\bar{z_2}}{\partial\beta}) \qquad \text{(a4)}$$

Combining (a2) and (a4), we eliminate the term $\frac{\partial\bar{x_2}}{\partial\beta}$ and thus the system can be written as a single expression which depends only on $\frac{\partial\bar{z_2}}{\partial\beta}$

$$\tau\frac{\partial\bar{z_2}}{\partial\beta} = \phi_c e^{-\delta\beta}\int_\theta a(\theta)^2(-\delta\bar{\lambda_2} + \frac{D_{zz}(\bar{z_2})}{\tau+\delta}\frac{\partial\bar{z_2}}{\partial\beta})\gamma(\theta)d\theta \qquad \text{(a5)}$$

By rearranging terms (a5) can be written as :

$$[\tau - e^{-\delta\beta}\frac{D_{zz}(\bar{z_2})}{\tau+\delta}\int_\theta a(\theta)^2\gamma(\theta)\phi_c d\theta]\frac{\partial\bar{z_2}}{\beta} = -\delta\bar{\lambda_2}e^{-\delta\beta}\int_\theta a^2\gamma(\theta)\phi_c d\theta]$$

Since $\phi_c(\theta, c) < 0$ (thanks to R1), $\frac{\partial z_2}{\partial \beta}$ is positive. Consequently, we deduce that :

$$\frac{\partial \bar{z_2}}{\partial \beta} > 0, \text{ therefore } \frac{\partial \bar{\lambda_2}}{\partial \beta} > 0 \text{ thakns to a3 and therefore } \int_\theta a(\theta)\gamma(\theta)\frac{\partial \bar{x_2}}{\partial \beta}d\theta > 0$$

Influence of δ

The proof is the same as above but the R2-system at the steady state is differentiated with respect to δ :

$$\begin{cases} \frac{\partial \bar{x_2}}{\partial \delta} = \phi_c a(\theta)e^{-\delta\beta}(-\beta\bar{\lambda_2} + \frac{\partial \bar{\lambda_2}}{\partial \delta}) \\ \tau\frac{\partial \bar{z_2}}{\partial \delta} = \int_\theta a(\theta)\gamma(\theta)\frac{\partial \bar{x_2}}{\partial \delta}d\theta \\ (\tau+\delta)\frac{\delta\bar{\lambda_2}}{\partial \delta} + \bar{\lambda_2} = D_{zz}(z_2)\frac{\partial \bar{z_2}}{\partial \delta} \end{cases}$$

Influence of τ

The proof is the same as above but the R2-system at the steady state is differentiated with respect to τ :

$$\begin{cases} \frac{\partial \bar{x_2}}{\partial \tau} = a(\theta)e^{-\delta\beta}\phi_c\frac{\partial \bar{\lambda_2}}{\partial \tau} \\ \bar{z_2} + \bar{\tau}\frac{\partial \bar{z_2}}{\partial \tau} = \int_\theta a(\theta)\gamma(\theta)\frac{\partial \bar{x_2}}{\partial \tau}d\theta \\ \bar{\lambda} + (\tau+\delta)\frac{\partial \bar{\lambda_2}}{\partial \tau} = D_{zz}\frac{\partial \bar{z_2}}{\partial \tau} \end{cases}$$

Influence of ρ

We start from the R3-system at the steady state :

$$\begin{cases} \bar{q_3} = \phi(\theta, \dfrac{a(\theta)\bar{\lambda_3}e^{-\delta\beta}}{1+\rho}) \\ \bar{z_3} = \dfrac{1}{\tau}\displaystyle\int_\theta a(\theta)q_3\bar{(}\theta)\gamma(\theta)d\theta \\ \bar{\lambda_3} = \dfrac{D_z\bar{z_3}}{\tau+\delta} \end{cases}$$

Then, we follow the procedure used above after differentiating of the system with respect to ρ :

$$\begin{cases} \dfrac{\partial\bar{q_3}}{\partial\rho} = \phi_c\dfrac{(a(\theta)e^{-\delta\beta}\frac{\partial\bar{\lambda_3}}{\partial\rho}(1+\rho)+a(\theta)\bar{\lambda_3}e^{-\delta\beta})}{(1+\rho)^2} \\ \bar{z_3}+\tau\dfrac{\partial\bar{z_3}}{\partial\rho} = \displaystyle\int_\theta a(\theta)\gamma(\theta)\dfrac{\partial\bar{q_3}}{\partial\rho}d\theta \\ (\tau+\delta)\dfrac{\partial\lambda_3}{\partial\rho} = D_{zz}(\bar{z_3})\dfrac{\partial\bar{z_3}}{\partial\rho} \end{cases}$$

And finally, we show that : $\frac{\partial\bar{z}}{\partial\rho} > 0\,;\ \frac{\partial\bar{\lambda}}{\partial\rho} > 0\,;\ \frac{\partial\bar{x}}{\partial\rho} < 0\,;$

3.7.3 Proposition 3.4.1

We apply the Taylor's theorem at the first order to the $R4$-system regarding to the steady state :

$$\begin{cases} \bar{q_4}-\bar{q_3} = \dfrac{a(\theta)(\bar{\lambda_4}-\bar{\lambda_3})-\rho\frac{\Gamma(\theta)}{\gamma(\theta)}(\bar{q_4}-\bar{q_3})\Pi_{xx\theta}(\bar{q_3},\theta)}{1+\rho}\phi_c(\theta,\dfrac{a(\theta)\bar{\lambda_3}}{1+\rho}) & \text{(a6)} \\ \tau(\bar{z_4}-\bar{z_3}) = \displaystyle\int_\theta a(\theta)(\bar{q_4}-\bar{q_3})\gamma d\theta & \text{(a7)} \\ \bar{\lambda_4}-\bar{\lambda_3} = \dfrac{1}{\tau+\delta}(\bar{z_4}-\bar{z_3})D_{zz}(\bar{z_3}) & \text{(a8)} \end{cases}$$

We rewrite (a6) :

$$(\bar{q_4}-\bar{q_3}) = \frac{a(\theta)(\bar{\lambda_4}-\bar{\lambda_3})}{1+\rho+\rho\frac{\Gamma(\theta)}{\gamma(\theta)}\Pi_{xx\theta}(\bar{q_3})\phi_c(\theta,\frac{a(\theta)\bar{\lambda_3}}{1+\rho})}$$

Since $\phi_c(\theta,c) <= 0$ (R1) and $\Pi_{x\theta}(q,\theta) >= 0$ (H3), if $\Pi_{xx\theta} < 0$, then $\bar{z_4}-\bar{z_3} > 0$ (immediate with a7). As a result, $\bar{z_4}-\bar{z_4} > 0 \Rightarrow \bar{\lambda_4}-\bar{\lambda_3} > 0$ and the global instantaneous pollution is such that $\int_\theta a(\theta)(\bar{q_4}-\bar{q_3})\gamma(\theta)d\theta > 0$.

3.7.4 Proposition 3.4.2

We differentiate the $R4$-system from the steady state with respect to β :

$$\begin{cases} \frac{\partial\bar{q_4}}{\partial\beta} = \frac{\phi_c}{1+\rho}[a(\theta)e^{-\delta\beta}(\frac{\partial\bar{\lambda_4}}{\partial\beta}-\delta\bar{\lambda_4})-\rho\Pi_{xx\theta}\frac{\partial\bar{q_4}}{\partial\beta}\frac{\Gamma(\theta)}{\gamma(\theta)}] & \text{(a10)} \\ \frac{\partial\bar{z_4}}{\partial\beta} = \frac{1}{\tau}\int_\theta a(\theta)\frac{\partial\bar{q_4}}{\partial\beta}\gamma(\theta)d\theta & \text{(a11)} \\ (\tau+\delta)\frac{\partial\bar{\lambda_4}}{\partial\beta} = D_{zz}(\bar{z_4})\frac{\partial\bar{z_4}}{\partial\beta} & \text{(a12)} \end{cases}$$

which after rearranging (a10) and combining (a11) and (a12), we get :

$$\begin{cases} \frac{\partial\bar{q_4}}{\partial\beta} = \frac{\phi_c\frac{a(\theta)}{1+\rho}e^{-\delta\beta}(\frac{\partial\bar{\lambda_4}}{\partial\beta}-\delta\bar{\lambda_4})}{1+\frac{\rho}{1+\rho}\Pi_{xx\theta}\phi_c\frac{\Gamma(\theta)}{\gamma(\theta)}} & \text{(a13)} \\ (\tau+\delta)\frac{\bar{\lambda_4}}{\partial\beta} = \frac{D_{zz}(\bar{z_4})}{\tau}\int_\theta a(\theta)\frac{\partial\bar{q_4}}{\partial\beta}\gamma(\theta)d\theta & \text{(a14)} \end{cases}$$

Combining (a13) and (a14), we deduce :

$$(\tau+\delta)\frac{\partial\bar{\lambda_4}}{\partial\beta} = \frac{D_{zz}(\bar{z_4})e^{-\delta\beta}}{\tau(1+\rho)}(\frac{\partial\bar{\lambda_4}}{\partial\beta}-\delta\bar{\lambda_4})\int_\theta\frac{a^2(\theta)\phi_c\gamma(\theta)}{1+\frac{\rho}{1+\rho}\Pi_{xx\theta}\phi_c\frac{\gamma}{\Gamma}}d\theta$$

$$\underbrace{(\tau+\delta)-\frac{D_{zz}(\bar{z_4})}{\tau}\frac{e^{-\delta\beta}}{1+\rho}\int_\theta\frac{a^2(\theta)\phi_c\gamma(\theta)}{1+\frac{\rho}{1+\rho}\Pi_{xx\theta}\phi_c\frac{\gamma(\theta)}{\Gamma(\theta)}}d\theta}_{>0\ if\ \Pi_{xx\theta}<0}\frac{\partial\bar{\lambda_4}}{\partial\beta} = \underbrace{-\delta\bar{\lambda_4}\frac{D_{zz}(\bar{z_4})}{\tau}\frac{e^{-\delta\beta}}{1+\rho}\int_\theta\frac{a^2(\theta)\phi_c\gamma(\theta)}{1+\frac{\rho}{1+\rho}\Pi_{xx\theta}\phi_c\frac{\gamma(\theta)}{\Gamma(\theta)}}d\theta}_{>0\ if\ \Pi xx\theta<0}$$

If $\pi_{xx\theta} < 0$, then we immediately get that $\frac{\partial \bar{\lambda_4}}{\partial \beta} > 0$, $\frac{\partial \bar{z_4}}{\partial \beta} > 0$. The derivative of total amount of instantaneous polluting input at the steady state is such as $\frac{\partial \bar{z_4}}{\partial \beta} = \int_\theta a(\theta) \frac{\partial \bar{q_4}}{\partial \beta} \gamma(\theta) d\theta > 0$.

Deuxième partie

Evaluation des politiques de régulation des pollutions azotées

Chapitre 4

Description des modèles

Les principaux intérêts de la modélisation économique sont de pouvoir évaluer ex-ante les impacts environnementaux, économiques, voire géographiques, des décisions publiques et d'étudier différentes politiques visant à satisfaire un objectif précis afin de fournir des recommandations aux différentes agences de régulation.

4.1 Les Modèles agricoles bio-économiques

Les modèles agricoles bio-économiques (MABE) sont particulièrement adaptés pour répondre à ces objectifs. Ils permettent d'établir des liens entre le comportement des agriculteurs, les possibilités des productions actuelles et alternatives, et les résultats observés aux niveaux des productions effectives, des usages des sols et des externalités associées (Janssen and Van Ittersum, 2007). Par rapport à d'autres approches appliquées comme les modèles multi-agents, les indicateurs agri-environnementaux ou l'évaluation des impacts environnementaux, les MABEs : (i) sont fondés sur une procédure d'optimisation sous contrainte ce qui permet de mieux prendre en compte la réalité agricole (Anderson et al., 1985) ; (ii) comportent de nombreuses activités, de restrictions et de nouvelles techniques de production avec leurs cahiers des charges respectifs peuvent être considérées en même temps (Wossink et al., 2001), y compris les liens entre les cultures et l'élevage (Antle et al., 2001) ; (iii) et peuvent être utilisés à la fois pour des scénarios de court et long terme.

Dans la littérature, plusieurs termes sont utilisés pour distinguer ces modèles, par exemple, "bio-économique", "ecological-economic" et "combining the environmental and economic". Suivant la typologie utilisée par Janssen and Van Ittersum (2007), nous distinguons les MABEs selon qu'ils adoptent une approche empirique ou mécanistes. Les premiers sont le plus souvent basés sur des séries chronologiques et les comportements passés. A partir de base de données, ils visent à trouver une relation entre ces données et ce qui n'est pas connu *ex-ante* (Austin et al., 1998). Ces modèles peuvent ainsi difficilement prendre en compte de nouvelles contraintes, des productions alternatives ou de nouvelles politiques économiques (Falconer and Hodge, 2000). A l'inverse, le terme mécaniste fait référence à la capacité des modèles à simuler les procédés physiques à travers un ensemble de variables et de contraintes. Ils visent à pouvoir simuler des comportements cohérents entre les données et les connaissances scientifiques (Antle et al., 2001). Le modèle économique d'offre agricole utilisé dans la suite de la thèse fait partie de cette dernière catégorie.

Ces modèles utilisent généralement des modèles mathématiques d'optimisation, souvent basés sur la programmation linéaire (LP). Ils représentent la ferme comme une combinaison linéaire d'activités pré-définies qui représente un ensemble cohérent d'opérations avec les entrées et les sorties correspondantes. Une activité est caractérisée par un ensemble de coefficients qui exprime la contribution de l'activité à la réalisation d'objectifs définis. Les contraintes représentent le montant minimum ou maximum d'une certaine entrée ou ressources qui peuvent être utilisées. Ce système d'activités et de contraintes est alors optimisé pour une fonction-objectif, reflétant un objectif spécifié par l'utilisateur, par exemple la marge brute agricole.

Les contraintes biophysiques sont, en général, assurées par des modèles de cultures qui permettent de simuler leur croissance et les rendements obtenus en fonction, par exemple, de l'apport azoté et en prenant en compte les conditions climatiques et les diverses qualités des sols. Ils permettent également de déduire les pertes azotés par différence entre l'azote consommé par les cultures au sein de leur processus de croissance et celui qui leur est apporté. Cependant ces modèles ne peuvent pas être extrapolés à d'autres zones géographiques que celles qui ont servi à l'élaboration du modèle.

On peut distinguer deux façons de coupler les modèles de cultures et les modèles économiques (Petsakos, 2008). La première approche utilise les résultats des simulations de modèles de cultures pour déterminer des coefficients entre les apports azotés d'une part, et les rendements et les pertes azotées d'autres part. Ils sont ensuite utilisés comme intrants dans le modèle économique. Cette approche est utilisée, par exemple, par Johnson et al. (1991) qui cherchent à déterminer les impacts de diverses méthodes pour réduire la pollution aux nitrates dans le bassin de Columbia (Oregon, USA). Plus récemment, Semaan et al. (2007) analysent les effets des différentes options politiques sur le revenu des agriculteurs et le lessivage des nitrates dans une zone de 100 hectares agricoles dans le sud de l'Italie. Ce type d'approche nous paraît clairement incomplète pour étudier l'impact de tels scénarios car non coût-efficace. En effet, l'utilisation de coefficients techniques empêche de modéliser l'impact de scénarios incitatifs comme une taxe sur les engrais. Ainsi, seuls des scénarios de réduction uniforme d'engrais sur chaque culture peuvent être modélisés. Or une politique coût-efficace chercherait à ce que, pour un même objectif de réduction, l'agriculteur puisse choisir son niveau de réduction d'intrants sur chaque culture en fonction des rapports des prix.

La deuxième approche améliore le lien entre modèles de cultures et modèles économiques en substituant les coefficients techniques de rendements par des fonctions de réponses à l'azote. Les fonctions de réponse obtenues peuvent alors être directement incorporées au modèle économique. On dispose alors d'une représentation plus réaliste et détaillée de la relation entre les apports d'azote, les rendements et les émissions. Ce couplage permet aussi d'améliorer l'analyse de réduction uniforme qui ne porte plus directement sur les cultures mais sur l'ensemble des exploitations et permet d'introduire des politiques incitatives comme la taxation des engrais azotés. Cette approche étant utilisée dans les chapitres suivants, une description plus détaillée de la construction des fonctions de réponse est exposée dans les sections suivantes. Ces approches ont été utilisées par Helfand and House (1995) pour mesurer la pollution par les nitrates due à la production de laitue dans la vallée de Salinas en Californie. Ils utilisent le modèle de culture EPIC et font varier deux types de sol pour créer 750 combinaisons différentes en fonction de l'irrigation et des niveaux d'intrants, afin d'estimer la production agricole et la pollution par les nitrates. Des fonctions de réponse pour estimer à la fois les rendements et la pollution sont également construites de façon similaire par Martínez and Albiac (2006) pour examiner l'efficacité des politiques de contrôle des pertes azotées dues à la production de maïs en Espagne.

4.2 Un modèle d'offre Agricole : AROPAj

Présentation générale AROPAj est un modèle d'offre agricole européen mono-périodique, basé sur de la programmation mathématique mixte, linéaire et en nombres entiers, et développé par l'INRA-Grignon. Il est alimenté par le RICA qui est une base de données européenne à composante micro-économique conçue à partir d'enquêtes nationales, réalisées chaque année auprès des agriculteurs. Ainsi AROPAj peut couvrir tous les pays présents dans la base de données RICA, i.e, les états membres de l'Union Européenne.
AROPAj maximise une fonction-objectif pour chaque "groupe-type d'exploitation agricole" ayant leur propre ensemble de contraintes. AROPAj consiste donc en un ensemble de modèles indépendants qui décrit le comportement économique du groupe-type correspondant. Il permet de représenter la diversité des pratiques agricoles observées dans l'Union Européenne, et la plupart des cultures annuelles, des prairies et la majeure partie des productions animales. Au sein de chaque "région RICA" (i.e. les régions administratives, en France), la typologie des groupe-types est basée sur trois caractéristiques : (i) l'activité agricole (14 types d'activités agricoles sont différenciées dans le RICA), (ii) l'altitude (<300m, 300-600m, >600m) et (iii) la taille économique. En outre, chaque groupe-type doit être au minimum associé à 15 exploitations agricoles de l'échantillon RICA pour des raisons de protection des données individuelles. Ainsi, un groupe-type représente l'exploitation agricole moyenne d'un ensemble d'exploitations supposées proches en termes de pratiques agricoles, d'altitude et de taille économique.

Optimisation et contraintes Le problème d'optimisation de chaque groupe-type est celui de la maximisation de sa marge brute sous un ensemble de contraintes que l'on peut exprimer mathématiquement de la façon suivante :

$$\max_{\mathbf{x}_k} \quad \pi_k = \mathbf{g}_k.\mathbf{x}_k \tag{4.1}$$

$$s.c \quad \mathbf{A}_k.\mathbf{x}_k \leq \mathbf{z}_k \tag{4.2}$$

$$\mathbf{x}_k \geq 0 \tag{4.3}$$

Les groupes-types sont indicés par k, π_k représente la marge brute totale du groupe-type k. $\mathbf{x}_k$ et $\mathbf{b}_k$ sont respectivement les vecteurs n x 1 du niveau, inconnu, de production et du niveau, connu, des prix de vente moins les coûts associés à la production (appelés également opérateur de marge brute). L'ensemble de contraintes est représenté par la matrice A, n x m, des coefficients techniques associés aux contraintes de technologie de production et par le vecteur $\mathbf{z}_k$, m x 1, de ressources disponibles et de paramètres formant le coté droit de l'ensemble de contraintes.

Le vecteur $\mathbf{x}_k$ concerne la production et la surface associée à chaque culture, le stock animal, la production animale (lait et viande) et la quantité nécessaire pour la consommation animale. Au total, 32 activités concernant les cultures sont modélisées, dont le blé, dur et tendre, le maïs, le colza, le riz, la betterave, le tournesol, le soja, la pomme de terre, la betterave sucrière, le pois ainsi que le fourrage, la prairie et la friche. 31 activités concernent l'activité animale, dont 27 concernent les bovins (différenciés suivant l'âge et l'usage). Les 4 restantes concernent les moutons, les chèvres, les porcs et les volailles. Le vecteur $\mathbf{g}_k$ est constant pour chaque activité. Il définit la différence entre le revenu net unitaire et le coût variable correspondant pour chaque activité. Pour les cultures destinées à la consommation animale, ce prix représente leur coût variable de production (De Cara et al., 2005). C'est de ce vecteur que viennent les hypothèses les plus fortes, car il implique que les variations de production induites, par exemple, par une politique de taxation des engrais, n'ont pas d'influence sur les prix (agents "price takers") et que les coûts sont linéaires.

L'ensemble de contraintes inclut (i) les rotations de cultures et les contraintes agronomiques, (ii) les restrictions concernant la démographie et l'alimentation animale, (iii) les restrictions concernant les facteurs de productions quasi fixes (Surface agricole et cheptel animal) et (iv) les restrictions induites par la PAC (De Cara and Jayet, 2000).

La base de données RICA ne fournit pas d'information sur l'utilisation d'input par activité mais uniquement des informations agrégées (récolte, fertilisation totale) en valeurs monétaires et non en quantité.

Calibration et validation Afin d'être utilisé pour évaluer des politiques publiques ou pour estimer l'impact de différents scénarios, AROPAj doit être soumis à un processus de validation. Parmi les différentes méthodes de validation proposées par Hazell and Norton (1986), la plus standard est celle consistant à comparer les résultats du modèle avec les valeurs observées des variables. En effet, un modèle incapable de reproduire les observations de l'année de référence ne peut pas être jugé pertinent et admissible pour l'analyse de politiques publiques. La procédure de calibration est basée sur la ré-estimation d'une partie des paramètres du modèle par la combinaison de la méthode des gradients et de Monte Carlo afin de minimiser la différence entre les observations et les résultats du modèle pour chaque groupe-type (De Cara and Jayet, 2000).

4.3 Couplage d'AROPAj avec STICS

STICS (Brisson et al., 2003) est un modèle de culture à pas de temps journalier développé par l'INRA depuis 1996, basé sur les bilans physiques d'azote et d'eau, et prenant comme inputs les conditions climatiques, les types de sol et les pratiques agricoles. Il est constitué de plusieurs modules, chacun traitant avec un ensemble de fonctions biophysiques (rendements, formation de la biomasse, bilan azoté et bilan d'eau par exemple). Un dernier module est consacré à la simulation de pratiques agricoles comme l'irrigation et l'usage de fertilisants.

L'élément essentiel pour le couplage concerne le module de bilan azoté. Il permet d'estimer l'azote consommé par les plantes et les pertes (fraction d'azote non-consommé). Le couplage avec STICS consiste à remplacer la moyenne des rendements des cultures de chaque "groupe-type" par des fonctions de réponse à l'azote. Cela revient à la transformation d'AROPAj, modèle linéaire, en un modèle non-linéaire par rapport à l'azote.

Les fonctions de réponses sont définies comme un cas particulier des fonctions Misterliech :

$$Y_{j,k}(N) = Y_{j,k}^{max} - (Y_{j,k}^{max} - Y_{j,k}^{min})e^{-C_{j,k}*N} \tag{4.4}$$

Y représente le rendement estimé de la culture j sur le groupe-type k, $Y_{j,k}^{max}$ est le rendement maximal obtenu pour une culture sur un type de sol, d'altitude et de climat donné si elle n'est soumise à aucun stress du à un manque d'azote. Y_{min} est le rendement minimal, $C_{j,k}$ représente la courbure de la fonction de réponse à la quantité d'azote, N, appliquée. Ce type de fonction correspond aux standard économique en étant croissante et concave croissante et concave et a, en outre, l'avantage de rendre son interprétation agronomique aisée. L'élaboration de la fonction de réponse se fait suivant l'approche décrite par Godard

(2005) et Godard et al. (2008) :

- Choix des années climatiques n et $n+1$, n étant l'année RICA.

- Pour chaque couple {groupe-type, culture} préexistant dans le modèle AROPAj, un fichier STICS est constitué regroupant les 5 types de sols les plus représentatifs, 3 variétés ou dates de semis, 2 précédents possibles soit un total de 30 options. Deux options possibles d'approvisionnement en eau sont également possibles selon que les cultures sont considérées comme fortement irriguées ou non.
- Pour chacune de ces options, 31 simulations STICS sont effectuées pour un apport d'azote allant de 0 à 600 kg d'engrais de référence par hectare, par pas de 20 unités.
- Pour chacune de ces séries de points, on ajuste les paramètres de la fonction $Y(N)$.
- Parmi les fonctions ayant un Y_{max} supérieur aux rendements estimés par les donnés RICA, on sélectionne celle qui minimise l'écart entre la dérivée $Y'(N)$ au point d'intersection avec le rendement RICA de référence et le rapport du prix de l'azote sur le prix du produit agricole.

Cette fonction est enfin implémentée dans AROPAj en remplacement des données RICA et la dose d'azote devient ainsi une variable de commande. Cette dernière étape suppose que les agriculteurs sont rationnels, i.e maximisent leur profits, et risque-neutre. En outre, l'intérêt de ce couplage est de pouvoir étudier l'impact de politiques incitatives comme une taxe (sur les engrais, sur les pertes) plutôt que des scénarios de réduction arbitraires d'azote.

Estimation des pertes Une fois les fonctions de rendement sélectionnées, le module bilan azoté de STICS peut nous permettre d'estimer les pertes azotées comme étant la différence entre l'azote appliqué et l'azote consommé par les cultures. Les modules azotés permettent également d'établir une différence entre les pertes d'azote sous forme de nitrates (NO_3), d'oxyde d'azote (N_20) et d'ammoniac (NH_3). Les émissions sont obtenues en interpolant les pertes azotés données par STICS et les doses d'engrais apportées de la façon suivante :

$$e_{i,j,k} = A_{i,j,k}.N + B_{i,j,k} \tag{4.5}$$

i étant le polluant (NO_3, N_2O, NH_3) de la culture j présent sur le groupe-type k. Pour avoir des paramètres les plus réalistes, l'interpolation n'est pas réalisée entre 0 et 600 kg d'engrais mais entre 80 et 420, qui sont les valeurs minimales et maximales d'engrais appliquées en réalité. Le paramètre $A_{i,j,k}$ s'interprète comme étant la contribution marginale à la pollution$i$ et$B_{i,j,k}$ exprime les pertes minimales d'azote dès lors qu'on décide de fertiliser. Ces pertes proviennent des processus physiques naturels et du climat considéré à travers les précipitations pour les nitrates par exemple. On peut rappeler que l'IPCC utilise pour comptabiliser les émissions de N_2O un paramètre A uniquement d'une valeur de 1.25%. Durandeau et al. (2010) comparent alors les émissions de N_2O pour la France du Nord, soit en utilisant les coefficients IPCC, soit en utilisant des fonctions de réponses. Ils montrent alors qu'utiliser les coefficients IPCC surestime les flux de N_2O, ceux obtenus avec les fonctions de réponse étant en moyenne 20% inférieurs.

4.4 Spatialisation

Connaissant à travers le RICA les principales orientations technico-économiques des exploitations et leur regroupement en groupe-type agro-économiques, l'objectif est de pouvoir localiser géographiquement les groupe-types sur la base de données biophysiques et/ou statistiques disponibles en Europe. Cet objectif est atteint à l'aide d'un modèle spatial implémenté dans un Système d'Information Géographique qui permet la réalisation d'une cartographie de la localisation des groupes-types au sein des régions RICA. La spatialisation permet donc de s'affranchir de l'unité régionale sur la base de laquelle sont définis les groupes-types statistiquement représentatifs de l'activité agricole régionale. La méthode de spatialisation, basée sur le travail de Chakir (2009), est présentée par Cantelaube et al. (2012) suit deux étapes : (i) Spatialisation de l'utilisation du sol à une échelle fine (maille ou cellule élémentaire) par le calcul de la probabilité de rencontrer sur chaque maille, tel type d'activité agricole et (ii) Spatialisation des groupes-types AROPAj par le calcul des contributions de chacun des groupes-types à l'activité agricole sur chacune des mailles élémentaires. La première étape consiste à associer chaque cellule de 100mx100m issue de la base de données géo-référencée CORINE Land Cover (CLC) avec les données de l'enquête statistique LUCAS. L'association est nécessaire car la base de données CLC contient des informations sur la nature de la cellule (urbain, agricole, ...) mais ne permet pas de distinguer les activités agricoles. LUCAS, à l'inverse, comporte de nombreuses classes relatives aux pratiques agricoles. Il fournit ainsi un échantillonnage du territoire grâce à un ensemble de points d'observation géo-référencés, sans pour autant constituer une carte. L'association entre l'usage des terres observé dans la base de données LUCAS et les cellules de la carte CLC se fait à l'aide d'un modèle logit multinomial qui permet l'estimation de la probabilité p_{ij} qu'une catégorie d'activité agricole j observée dans LUCAS existe dans la cellule CLC i. D'autres variables sont ajoutées pour améliorer l'estimation : les caractéristiques des sols, le climat et l'altitude. La première étape est un processus en deux parties :

La première partie concerne l'estimation des p_{ij} pour chaque région RICA inclue dans AROPAj et le modèle mathématique correspondant peut-être écrit de la façon suivante :

$$p_{ij} = \frac{e^{x_{ij}}\beta_j}{\sum_{i=1}^{I} e^{x_{ij}}\beta_j} \tag{4.6}$$

$$avec \qquad \sum_{i=1}^{I} p_{ij} = 1, \qquad p_{ij} > 0 \tag{4.7}$$

x_{ij} sont les variables explicatives associées à l'activité agricole j et à la cellule i. β_j est le vecteur de paramètres devant être estimé pour l'activité agricole j.

La deuxième partie est une méthode d'optimisation de type "Generalized Cross Entropy" qui recalcule les probabilités de sorte que les surfaces régionales occupées par les différents types de cultures soient égales aux surface estimées à l'aide des données RICA.

$$\min_{\omega_{ij}} \qquad \sum_{i=1}^{I}\sum_{j=1}^{J} \omega ij ln(\frac{w_{ij}}{p_{ij}}) \tag{4.8}$$

$$s.c \qquad \sum_{i=1}^{I} \omega iju_i = R_j \tag{4.9}$$

$$\sum_{i=1}^{I} \omega ij = 1 \tag{4.10}$$

La contrainte 4.9 impose que la taille de l'activité j sur chaque pixel (en hec-

tare) définie comme la somme des probabilités ω_{ij} de trouver l'activité j sur le pixel i multipliée par la surface totale, u_i du pixel i soit égale à la surface R_j allouée à la même activité à l'échelle de la région issue de la base de données RICA. La contrainte 4.10 permet de s'assurer que pour chaque pixel, i, la surface allouée à l'ensemble des activités soit égale à la surface du pixel.
Enfin, la deuxième étape concerne la distribution des groupes-types AROPAj par estimation de la probabilité π_{ik} de localisation du groupe-type k sur le pixel i grâce à la probabilité ω_{ij} estimée précédemment. Parmi les trois variables utilisées dans la constitution des groupes-types (taille économique, altitude, pratiques agricoles), seule l'altitude a une correspondance avec les variables utilisées pour la procédure de désagrégation. On estime alors π_{ik} comme la probabilité conditionnelle de l'altitude, a, de la façon suivante :

$$\pi_{ik} = \sum_{j=1}^{J} \omega_{ij} \frac{S_{jk}|a}{\sum_{k=1}^{K} S_{jk}|a} \tag{4.11}$$

$$avec \quad \sum_{i=1}^{I} \pi_{ik} = 1 \tag{4.12}$$

S_{jk} désigne la surface allouée à la culture j par le groupe-type k, selon les estimations préliminaires basées sur le RICA. π_{ik} s'interprète alors comme la contribution de l'activité agricole j du groupe-type k sur le pixel i, qui à son tour exprime la probabilité que l'activité j soit présente sur le pixel i.

4.5 Applications

AROPAj-STICS va nous permettre d'implémenter différents scénarios basés sur les chapitres 2 et 3 et d'en quantifier les différents impacts.

Le chapitre 2 a mis en évidence le fait que coupler une subvention aux cultures pérennes à une taxe sur les engrais pourrait être coût-efficace comparé à un seul instrument-prix sur les inputs. L'objectif du chapitre suivant, chap. 5, est ainsi d'estimer les différents impacts d'une telle régulation au niveau de la France. Nous mettons alors en évidence un arbitrage entre la régulation des pollutions gazeuses, où ces instruments n'apparaissent pas coût-efficaces, et la pollution de l'eau où le gain apparait substantiel. En effet dans ce dernier cas et selon l'objectif de réduction considéré, les coûts d'abattement peuvent être divisés par 2 à 3. AROPAj-STICS permet également de mettre en évidence la forte hétérogénéité des émissions non seulement entre les cultures mais aussi d'une région à l'autre. Cette hétérogénéité rejaillit, en effet, fortement sur les gains attendus d'une telle régulation suivant la région considérée.

En couplant AROPAj et STICS à un modèle hydrologique , MODCOU le chapitre 6 permet de mettre en évidence l'impact du délai entre l'application d'engrais et la pollution des aquifères. Ces modèles, intégrés à l'analyse théorique effectuée dans le chapitre 3, permettent également d'estimer la valeur que le régulateur accorde aux dommages liés à la pollution des aquifères par les nitrates. En effet si l'on considère que le seuil de concentration fixé par le régulateur est à son niveau optimal, cela revient à inverser dans la modélisation le rôle du paramètre de dommage avec celui du seuil de concentration. Obtenir cette valeur permet également de définir le sentier optimal de réduction des nitrates au cours du temps et ainsi d'évaluer le coût social associé à l'adoption de sentiers non-optimaux.

Chapitre 5

How cost-effective is a mixed policy targeting the management of three pollutants from N-fertilizers ?

Ce chapitre est issu d'un working paper en collaboration avec Pierre-Alain Jayet, Nosra Ben Fradj et Melissa Clodic et présenté à Rome au 18eme congrès de l' Annual Conference of the European Association of Environmental and Resource Economists (EAERE,2011) et à Zurich au 13eme congrés de l' European Association of Agricultural Economists (EAAE,2011).

5.1 Introduction

The use of nitrogen fertilizers is a major source of three environmental pollutants : nitrates (NO_3), nitrous oxyde (N_20), and amnonium (NH_3). NO_3 pollutes water, causing soil eutrophication and human health problems,such as the baby-blue syndrome and stomach cancer(Addiscott, 1996). N_20 contribute to global warming [1]. NH_3 plays a role in acid rain. Policies have been addressing these issues on a pollutant-by-pollutant basis. For example, the Water Framework Directive (2000/60/EC), adopted by the European Commission, deals with water quality conservation by preventing nitrogen pollution arising from the agricultural sector (mineral fertilizer and livestock manure) through a set of good agricultural practices (especially in terms of N-input consumption). More recently, the European commission decided to reduce total EU greenhouse gas emissions, from the sectors currently not covered by the ETS [2], by approximately 10% in 2020, relative to 2005 levels (European Union, 2009).

In the field of economics, nitrogen pollution is characterized by the fact it is a nonpoint source pollution (NPS). First-best emission instruments are therefore inappropriate in attempts to regulate it (Helfand and House, 1995). Second best policies include various mechanisms, mainly land-use policy, the regulation of ambient concentration and input tax, as reviewed in Shortle and Horan (2001). Most studies have focused on these mechanism one at a time. For example,in the case of land-use, it has been shown that perennial crops dedicated to bioenergy can have positive impacts on the environment (Lankoski and Ollikainen, 2008, 2011). These authors studied the environmental impacts

1. According to the latest greenhouse gas (GHG) inventories by the European Environment Agency (2010), agricultural emissions represent about 10% of total EU emissions.
2. Emission Trading Scheme

of biofuel policy in Finland and showed that contrary to rape seed and cereals, when they are dedicated to the production of biodiesel or ethanol, only reed canary grass is unambiguously desirable, because it requires low doses of fertilizer. Consequently, they recommended a subsidy for this crop in order to restore the social optimum.

Ambient concentration mechanisms and input tax are policy instruments whose limits have been shown. Studies focusing on the application of an ambient concentration mechanism found it difficult because of problems in terms of measuring discharges from individual farms and/or implementation as well as in terms of legally attributing any random or collective penalties. Studies focusing on input tax found that it equals the marginal profit of fertilizer use across heterogeneous land but not the marginal pollution control cost and the land-use reallocation does not necessarily favored the environment (extensive margin effects). Moreover, a single-tax policy is not sufficient regarding the Water Framework Directives and, for example, an increase in grassland or an introduction of catch crops can appear more cost effective (Lacroix et al., 2005).

Given this context, cost-effective management of pollution arising from the use of fertilizers requires accounting for both intensive (input regulation) and extensive margin (land-use regulation) effects (Shortle et al., 1998; Goetz et al., 2006). However, few studies have taken into consideration both these aspects at the same time. Aftab et al. (2010) did combine managerial measures (farm land retirement and a reduction in livestock stocking density reduction) with an input tax. Goetz et al. (2006) combined a tax/subsidy on each crop with an input tax. These studies, focusing on one pollutant ($N0_3$), reported the superiority of a mixed approach in terms of cost effectiveness. However, the introduction of multi-crop instruments or managerial measures in order to regulate the extensive margin would likely lead to a rise in enforcement costs and it would be difficult to implement. Moreover, the relatively small areas [3] and the limited number of crops taken into account [4] can fail to capture the inherent heterogeneity among real-world farms and thus, can bias the estimation (Balana et al., 2011). There is clearly a need for a better comprehension of gains, in terms of cost effectiveness, if a well-specified mixed policy would to be adapted. In addition, there are numerous land-use policies. Which one would be well adapted, to add to an N-tax in a mixed policy approach, is still being debated.

In this paper, we raise the following question : How cost effective a mixed policy targeting the management of three pollutants arising from N-fertilizers ? We combine a common N-input tax and a subsidy for a perennial bioenergy crop. Miscanthus is chosen thanks to high nitrogen, energy and land-use efficiencies at all N fertilizer and energy input levels compared to other perennial crops, especially reed canary grass (Lewandowski and Schmidt, 2005). To tackle this question, we provide an integrated economic and agronomic approach. A supply agricultural model is combined with a crop model in order to capture the soil and crop heterogeneities in fertilizer response (in terms of yields and emissions) as well as compute nitrogen pollutions.

We show that a mixed policy increases the pollutant-abatements compared to the N-tax. Moreover, a mixed policy is cost-effective in the case of the NO_3 while the single tax is cost-effective for gas emissions. Another interesting advantage of a mixed policy is the possibility of choosing the couple of instruments which grants the largest abatement for a given agricultural in-

3. Respectively, eastern Scotland and the aquifer in the watershed of Lake Baldegg, Switzerland.

4. Respectively 4 and 5.

come loss and maximizing the abatement for a given revenue. Particularly, for example, for a given revenue between 0.6 and 0.8 M€ for the social planner, the abatement range is between 7 and 35 % in the case of NO_3 and between 16 and 50% in the case of N_2O. Moreover, the soils, crops and emissions being very heterogeneous according to the considered region, we provide an analysis by river basin. First of all, we show that these heterogeneities are reflected in the abatement cost curves. Then,we show that the mixed policy, and especially the land-use policy, should be adapted per basin in order to improve the cost-effectiveness of regulation.

The paper is organized as follows. The model is explained in section 2. Results for France and per river basin are given in section 3. In section 4, results are discusses in terms of land-use, crop area allocation and emission functions.

5.2 The Model

5.2.1 The agro-economic model

The investigation undertaken here relies on an updated version of the economic model, AROPAj, initially presented by De Cara et al. (2005) and now described by Galko and Jayet (2011). In short, the overall model consists in a set of 1,074 independent [5], mixed integer linear-programming models (MILP). Each model describes the annual supply choices of a representative farmer (or a 'farm group') who optimizes crop and livestock production with respect to a set of technical and economic constraints. AROPAj covers the main annual crops and animal categories relevant to European agriculture. Farmers are grouped into farm types according to : the farm techno-economic orientations within the region, the farm economic size, and the farm altitude class. These farms are statistically representative of the different production systems at the regional level thanks to the Farm Accountancy Data Network (FADN). Each farm group is assumed to maximize its total gross margin. The set of constraints includes ; (i) crop rotation and agronomic constraints ; (ii) CAP-related constraints ; (iii) restrictions concerning animal demography and nutritional requirements, and (iv) restrictions concerning quasi-fixed production factors (land and livestock). [6] FADN data overcome the problems of data confidentiality and improve the representation of farm heterogeneity in comparison to many bioeconomic models (e.g., Aftab et al. (2010), Aftab et al. (2007).) For a more detailed and technical description of model structure, the

5. In the case of the V2 version based on the EU-15 which, for this study, was adapted to France.

6. Following De Cara et al. (2005), we assume in our central set of simulations that livestock numbers are allowed to vary within +/- 15 % of the values reported in the FADN database.

set of constraints and the main features, see De Cara et al. (2005). For this investigation, we have introduced the changes in agricultural policy that were put into action in 2005, more specifically the decoupling schemes triggered by the implementation of the 2003 CAP reform (Galko and Jayet, 2011).

Balana et al. (2011) note that a large number of agri-environmental studies, which are based on a 'stylized' farm, may fail to capture the inherent heterogeneity of real-world farms, thereby sending wrong signals to the decision-making process.They suggest the use of actual farm data instead of 'stylized' farms. Our methodology retains the advantages of these two approaches. Indeed, on the one hand, our farm groups are representative at the regional scale thanks to the use of FADN data, and, on the other hand, the functional form of a farm group allows us to take into account changes in environmental and economic constraints.

Coupling the economic model AROPAj with the agronomic crop model STICS (Brisson et al., 1998, 2003) captures the heterogeneity more precisely by taking into account the wide-range of soil characteristics, the impact of soil nitrogen supply on nitrogen plant uptake, farm practices, and the climate. This also allows us to deal with most of the crops relevant to EU agriculture.[7] This coupling involves the replacement of average ' point N-Yields' with a Mitscherlich-response function of nitrogen dependent on physical (climatic and soil) and economic (price and economic policy) factors for crops in each farm group. This improves the representations because the farmers can adapt their nitrogen fertilizer practices to physical constraints and the economic context (For a more detailed technical description, see Godard et al. (2008)). This leads to the transformation of AROPAj from an LP to a NLP model, with respect

7. Accounting for 14 crops, including soft and hard wheat, maize and colza

to nitrogen. Moreover, the nitrogen-yield function response improves the impact assessment regarding price scenarios like a nitrogen taxation, better than relying on arbitrary uniform changes in practices (e.g. ; 20 % less fertilizers, Lacroix et al. (2005)). Finally, after coupling, the AROPAj model determines the most profitable land and nitrogen allocation to each farm group as well as animal numbers, animal feeds consumption, output, and gross margin. We can note that livestock waste is accounted for as a source of fertilizer (and nitrate) and a substitute for mineral fertilizers.

In addition to the amount of fertilizers, the STICS crops model provides us with the leaching associated with each type of soil and crops as well as climatic conditions and application date. Similar to Aftab et al. (2010), the leaching functions are derived by regressing STICS outputs within a reasonable range of nitrogen applications.

5.2.2 Introduction of the perennial bioenergy crop into the AROPAj model

Miscanthus x Giganteus[8] is a rhizomatous grass which originated in the tropics and subtropics, but is found under different species throughout a wide range of climate in eastern Asia (Greef and Deuter, 1993). The remarkable adaptability of miscanthus to different environments (Numata, 1974) makes it suitable for growth under European and North American climatic conditions (Lewandowski et al., 2000). Physiologically, miscanthus, like maize, is a C_4 species, fixing carbon by multiple metabolic pathways with a high water use efficiency (Koshi et al., 1982; Moss et al., 1969). Miscanthus roots can penetrate to a

8. Miscanthus x Giganteus is a steril hybrid between *M. Sinensis* and *M. Sacchariflorus.*

depth of around 2 meters, which can provide a good protection against soil erosion. Field trials have shown the high biomass yield potential, 15 to 20 tons dry matter per hectare, in comparison to other herbeceous crops (Clifton-Brown and Jones, 1996; Jorgensen, 1997; Lewandowski et al., 2000). Despite its high biomass yield potential, this crop requires small amounts of input and can therefore decrease the risk of ground water pollution by pesticides and nitrates.

To introduce miscanthus into the AROPAj model, two main elements must first be calculated for each farm-group : the Average Net Present Value (NPV^*) and the average yield (Y), both corresponding to the optimal rotation (T^*). The determination of NPV^* is based on a Faustmann's dynamic optimization over time, used in the case of a perennial crop harvested annually, and a function in terms of growth is used to compute Y. The growth function used at this level is calibrated on some data available from the work of Miguez et al. (2008), Clifton-Brown et al. (2007), and Christian et al. (2008), and adjusted to the average yield of a common annual crop. Indeed, miscanthus yields are not available in the FADN database, and knowing that miscanthus have been recently introduced in France, yield information for the full rotation period (15-20 years) is not provided as well. So, we suppose that miscanthus yield increases with the quality of the land, as does the wheat yield [9]. Wheat is considered to be a common crop, present in four-fifths of the French farm-group in AROPAj.

9. Wheat yield data are provided by the FADN database.

5.3 Results of the mixed policy abatement

We aim to assess the impacts of a mixed policy in terms of cost-effectiveness given three main N-pollutants : NO_3, N_2O and NH_3. The abatement cost curves in function of abatement for the three pollutants are evaluated for a range of subsidies varying from 0 to 250€/ha by increments of 25€/ha and for a range of tax values from 0% to 100% of the N-price by increments of 10%. The tax receipt and the subsidy payment being monetary transfers between the farmers and the social planner, this value is subtracted from the abatement costs in order to estimate net social welfare (omitting the other indirect economics effects). Thereby, the abatement costs represent the loss (in value) of production for the farmers due to both the lower yields and the area reallocation between crops induced by both the tax and the subsidy. The following sections present the results in the case for France.

5.3.1 French impact assessment

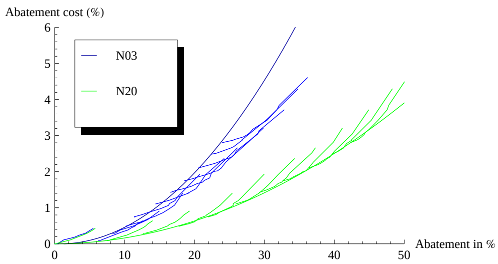


FIGURE 5.1 – These curves illustrate abatement costs when (1) only N-tax is implemented (the main curves), (2) the mixed policy is implemented (branching off from the main curves)

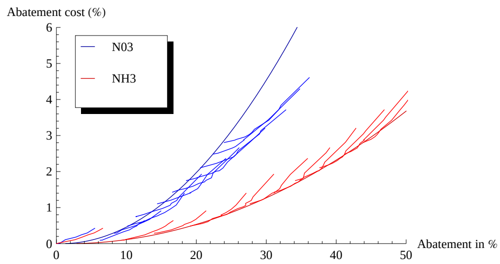


FIGURE 5.2 – These curves illustrate abatement costs when (1) only N-tax is implemented (the main curves), (2) the mixed policy is implemented (branching off from the main curves)

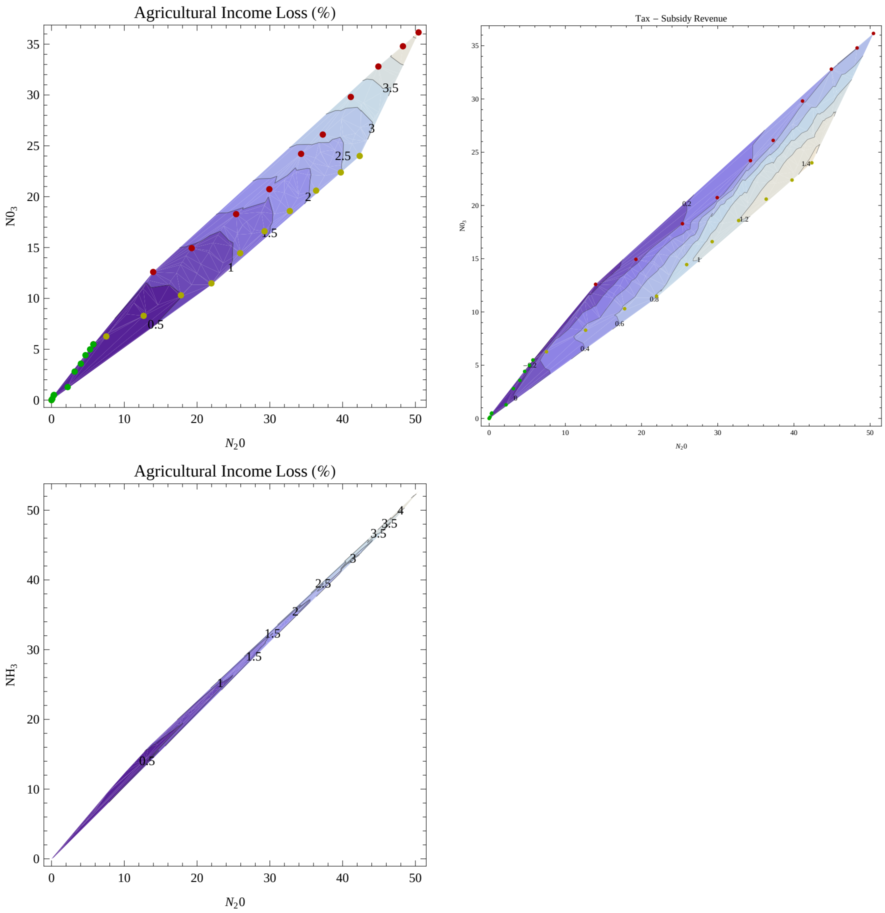
Figures 5.1 and 5.2 describe the abatement cost curves (in % of gross margin) in function of pollutant abatement. The bold curves illustrate the results obtained when a single N-input tax is applied. The other curves, branching off from the single tax curves, correspond to the additional abatement when subsidies are added to the tax.

Regardless the tax and subsidy combinations, the abatement costs concerning the gas pollutants are lower than cost for the $N0_3$ pollutant. For a given abatement cost, and in the case where only a single N-tax is implemented, the abatement obtained for the gas pollutants is twice as important as for the $N0_3$ pollutants. This result is in line with the numerous studies (ref) referring to difficulties in water pollution management in the case of N-tax implementation. When a land-use policy is added to a single N-tax, these effects are lessened. Indeed, when the maximum tax and subsidy levels are reached, the abatement gain of the mixed policy is lower than 10 points for the gas emissions whereas the abatement increases to half (from 24% to 36 %) for NO_3. Even when the land-use policy increase abatement of the three pollutants is added, the mixed policy is only cost-effective in the case of NO_3. As shown in figures 5.1 and 5.2, adding a miscanthus subsidy sharply decreases the abatement cost of NO_3 (in comparison with the tax alone) and slightly increases the abatement costs of gas pollutants (N_2O and NH_3).

Indeed, for a given abatement target, a mixed policy is less costly in the case of NO_3 whereas the single tax is less costly in the case of the gas pollutants. For example, a target of a 20% abatement leads to a decrease in cost of 0.8 points (from 2.8% to 2% of gross margin) for t $N0_3$. In other words, for a given $N0_3$ abatement target, we can find a couple (tax , subsidy) less costly than a single tax. This is noteworthy because $N0_3$ is the pollutant for which the abatement costs are the highest and the abatement range (with a tax alone) is restricted.

5.3.2 Impact on social planner's income

We have seen that a mixed policy can improve the cost-effectiveness of regulation. Another interesting advantage of a mixed policy is the possibility of choosing the couple which grants the best abatement without exceeding a given agricultural income loss. Indeed, the social planner has to face social reality (and farmers' lobby) and meet environmental objectives.This can lead the social planner to look for the best abatement for a given agricultural income loss. The following graphs draw a parallel between these two aspects. For example, in the first graph, if the social planner self-imposes not to exceed a loss of 1.5% t of agricultural income, we see that the possible $N0_3$ abatement is between 16 and 20%, and the possible N_20 abatement is between 22 and 30%. In the similar way, we can set a target for one of the two pollutants and look for the tax/subsidy combination which maximizes the second pollutant abatement. The second graph displays the income received by the social planner associated with each tax/subsidy combination. For a given revenue (between 0.6 and 0.8 M€), a wide-range of possible abatements exists. The third figure shows the symmetric characteristics when the gas pollutants are under consideration. Indeed, if we focus only on the gas pollutants, we see that the mixed policy does not lead to a flexibility on the abatement levels.



5.3.3 Impact assessment at the river-basin scale

We have shown that the cost-effectiveness of a mixed policy depends on both the given kind of pollutants and the possibility of replacing some existing crops with miscanthus. However, the nature of these two aspects is very heterogeneous, depending on the region under consideration. On the one hand, the crops and various constraints (soils, animal localization) are strongly heterogeneous given the river basin. On the other hand, the potential miscanthus area is not equally spatially distributed no matter the basin. Moreover, the emission function can vary in intensity according to the river basin. A cost-effective policy must therefore be adapted to the given river-basin characteristics.

In France, the basin agencies are in charge of water pollution management (six for metropolitan France), ensuring consistency between the scale of pollution and the area falling under the control of the agency.

We assess below the cost-effectiveness of a mixed policy according to the various river basin.

The graphs for each basin reflect heterogeneity in terms of abatement costs. For two basins (Seine and Artois basin), the gains associated with a mixed policy in terms of both abatement and abatement costs are very important. In the Seine basin, for an NO_3 abatement target of 20 %, the abatement costs are divided by 2.5 (2.5% to 1% of gross margin) and the range of possible abatement increases from 25% to 45% (for the given values of tax and subsidy). For the Artois basin, we see that an abatement increase based on only a tax is very expensive (compared to other basins) and limited (about 15% for a N-tax of 100%). However, the mixed policy increases the possible NO_3 abatement from 15 to 35 % and divides the cost by 3 to reach an abatement of 15%. Regarding N_2O, the additional abatement related to a mixed policy decreases with the level of N-tax.

For the Rhin and Rhone basin, we find the same results as for the above two basins : the mixed policy is only cost-effective in the case og the NO_3. However, there is less to gain when a mixed policy is adapted. For an NO_3 abatement target of 20 %, the abatement costs are decrease of 0.5 and 0.6 points (respectively 1.3% to 0.8% and 1.8% to 1.2 % to of gross margin for the Rhin and Rhone basin)

This can be explained by the fact that the abatement costs in the case of a N-tax policy are lower : respectively 1.2 % and 1.8% of gross margin to reach a NO_3 abatement of 20 % in comparison to 2.5% and > 3 % for the Seine and Artois basins. It is 1.6% to reach 40% of N_2O abatement for the Rhone and Rhin basins whereas abatement costs are 2.6% and 3.2% for the above basins and the same target.

Graph 5.7 shows that even if the mixed policy always increases the abatement range it cannot be cost-effective in the case the NO_3 in the Adour-Garonne basin. For the Loire-Bretagne basin, the benefits of a mixed policy in the case $N0_3$ is negative, except for the highest level of N-tax.

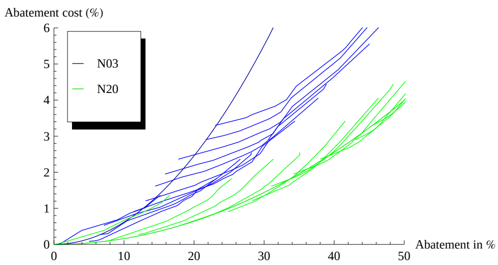


FIGURE 5.3 – Abatement costs (% of revenue) in function of NO_3 and N_2O abatements in mixed policy in the Seine basin

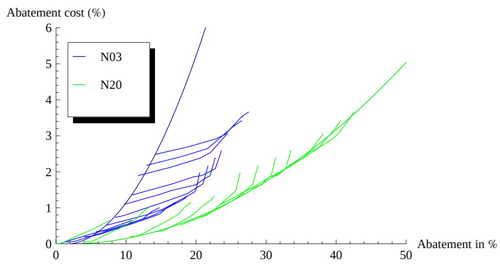


FIGURE 5.4 – Abatement costs (% of revenue) in function of NO_3 and N_2O abatements in mixed policy in the Artois basin

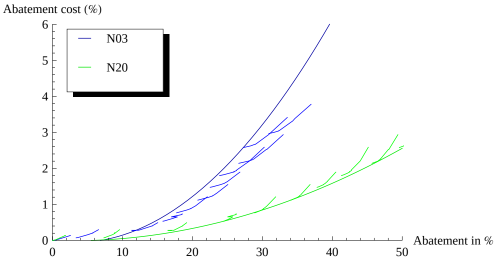


FIGURE 5.5 – Abatement costs (% of revenue) in function of NO_3 and N_2O abatements in mixed policy in the Rhin-Meuse basin

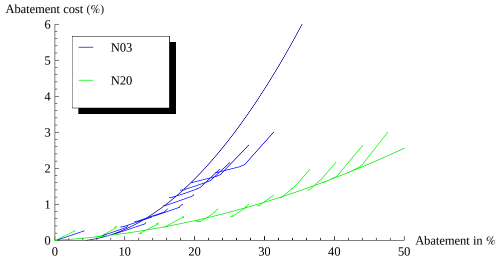


FIGURE 5.6 – Abatement costs (% of revenue) in function of NO_3 and N_2O abatements in mixed policy in the Rhone-Méditerannée basin

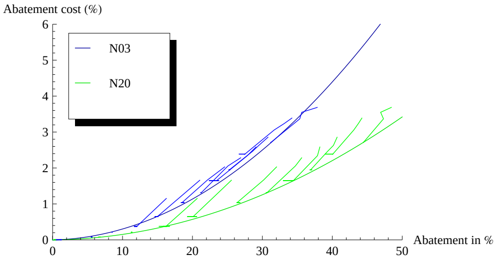


FIGURE 5.7 – Abatement costs (% of revenue) in function of NO_3 and N_2O abatements in mixed policy in the Adour-Garonne basin

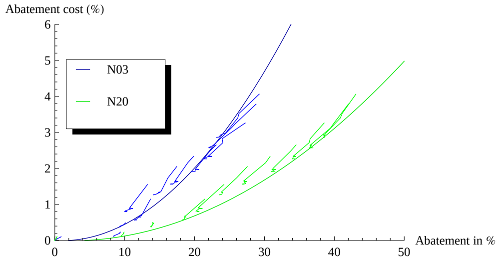


FIGURE 5.8 – Abatement costs (% of revenue) in function of NO_3 and N_2O abatements in mixed policy in the Loire-Bretagne basin

The abatement costs with a single N-tax being relatively high for the Loire-Bretagne basin, (like those for the Seine and Artois basins), one would have expected stronger results in the case of a mixed policy. To explain the extent of this difference, let us now discuss the impact of a mixed policy on land use.

5.4 Discussion

5.4.1 Result Interpretation

Natural N-losses, when there is no N-input, are greater in the case of NO_3 than in the case of N_2O and NH_3 (see Table1). This fact plays an important role in the cost-effectiveness of a mixed policy. Indeed, when the emission functions are not linear [10], an input tax is not sufficient to restore the optimal crop area allocation. [11] Moreover, this optimal allocation depends on the given pollution. The constant (β) for the gas pollutant is low (cf **??**), so an N-input tax policy leads to a crop area allocation close to the optimality for these pollutants.

The cost effectiveness of a mixed policy for the NO_3 pollutant is mainly due to the extensive margin effects : This is to say that the land is reallocated in favor of miscanthus (for which we assumed $\beta = 0$) due to the N-input tax and subsidies. In addition, since the natural losses (β) for the other pollutants are low for commonly found crops (especially soft wheat), the land reallocation in favor of miscanthus leads to fewer additional abatement in comparison to the N-fertilizer tax alone.

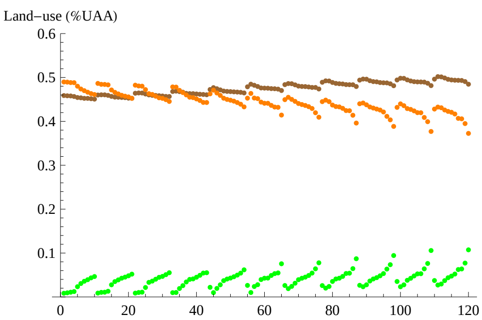


FIGURE 5.9 – Land use repartition : Miscanthus (Green), Grassland and Fallow land (Brown), Other crops (Orange)

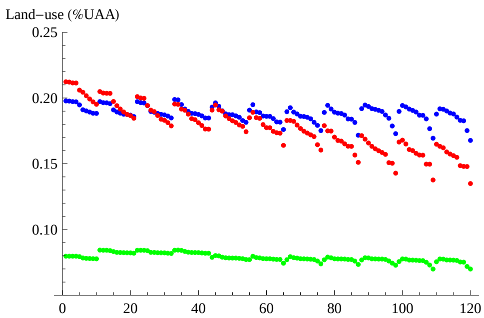


FIGURE 5.10 – Crop repartition in function of NO_3 constant emission

Indeed, as shown in (figure 5.9), the implementation of the tax alone leads to a moderate decrease in crop areas (about 6% of Utilizer Agricultural Area (UAA)) in favor mainly of grassland. Miscanthus, which is close to zero if the tax is null (lower profitability) can cover one third of these abandoned lands at the maximum of tax (1.9 % of UAA). At the maximum subsidy level (and

10. We call linear, the functional form : emission = $\alpha*$ fertilizers. In this paper, the emissions are interpolated as follows : emission = $\beta + \alpha * N$.

11. This is the subject of an article in progress. The proof can be provided on request

without N-tax), miscanthus can cover up to 4.6 % of UAA. At the maximum level of two instruments, miscanthus can cover up to 11% of UAA. This concave characteristic [12] makes a mixed policy more interesting in the cases of pollutants whose emission function depends on soil and crop, like NO_3. In other words, the higher the level of tax, the higher the impact of a subsidy on land use. Moreover, the increase in area of miscanthus also decreases the increae in the area of grassland due to the N-tax.

In order to focus on the effects of various policies on the crops, we group the crops into three categories, according to their level of NO_3 natural losses : low for natural losses below 20 k.N/ha, medium for natural losses between 20 and 40 k.N/ha, and high for natural losses greater than 40 k.N/ha. We see that the medium polluting crops are unaffected by the tax alone while the most polluting crops (for example, soft wheat) are strongly impacted. The most polluting crop areas for NO_3 are reduced. Taxes on input use intensity have a double effect on agricultural production. First, taxes reduce cropland yields (because less inputs are used : intensive margin effect) and decrease the area devoted to some crops (extensive margin effect). The subsidizing of miscanthus only reduces the proportion of area devoted to other cropland production, without reducing the yields of those croplands. However, except for the most polluting crop, the reallocation induced by a N-tax (alone) and a subsidy (alone) is relatively low (about 2 or3 %).

On the contrary, the mixed policy strongly modifies the allocation of crops, except for the least polluting crop. Especially, adding a subsidy decreases the medium and the most polluting crop areas. With a tax alone, the lands with lo-

12. Let $f(x, y)$ denote the miscanthus area due to tax (x) and subsidy (y). $f(x, y) \geq f(x, 0) + f(0, y)$

wer marginal yields for soft wheat become unprofitable and are then converted into fallow or grassland. However, the higher the tax, the higher the marginal profits of plots the last abandoned. Therefore, miscanthus can show up on these last abandoned plots. For other crops, plots with lower marginal profits remain higher than plots dedicated to miscanthus. When adding the subsidy, miscanthus profitability increases, and it can become more profitable than both soft wheat and medium polluting crops. This is a conversion of crops with high and medium natural losses of NO_3, explaining the great abatement gain in the case of this pollutant. Obviously, both N_2O and NH_3 abatement also increases because some crops for which fertilizers are applied are converted into an environmentally-friendly crop. However, the mixed policy is only cost-effective for NO_3. The cost-effectiveness of a mixed policy arises from the trade-off between the gain in agricultural income due to a lower tax and the loss of agricultural income due to the conversion of crops to a less profitable crop (miscanthus) induced by the subsidy income. As the natural losses (β) are important for NO_3, and a subsidy decreases the medium polluting crop areas (this is not case with a tax alone), we can decrease the N-tax to have a positive trade-off while keeping the same abatement target. In the case of N_2O and NH_3, as natural losses (β) and emission factor (α) are very low for the medium polluting crops, the abatement gain due to the conversion of these crop to miscanthus is low and a higher reallocation in favor of miscanthus is needed to compensate the additional emissions due to a lower tax.

The emission structure and the reallocation area induced by the tax and subsidy and the extensive margin effect explain why the mixed policy is cost-effective in the case the NO_3 and not for the gases. Therefore, the regulatory body is faced with a trade off between $N0_3$ and gas pollution control.

5.4.2 Basin heterogeneities

Abatement cost heterogeneity reflects the various crop production allocation between the basins as well as the differences in emission functions (cf **??**) between the basins due to the diversity of soils and climatic conditions. First, we can see on figures 5.11 and 5.13 that the area devoted to the crops are mainly in the regions allowing a greater possibility of miscanthus reallocation. Indeed, miscanthus can reach respectively 28 and 16 % of the UAA. Second, when the cropland covers the greater part of UAA, the gain in production induced by a decrease in a tax is more important than for a basin where the UAA is slightly covered by croplands. For a given NO_3 abatement target, rather than implement a tax alone, it is less costly, and thus more cost-effective, to convert the crop with the lowest marginal profit to miscanthus and decrease the tax leading to higher yields for other crops. This is the case in the Seine basin. However, we can note some differences for the Artois basin. Indeed, if the mixed policy is cost-effective, the possible NO_3 abatement is lower. Two reasons can be advanced. Firstly, the tax does not decrease the area devoted to crops. The most polluting crops are substituted by the medium polluting crop and not by grassland, fallow and miscanthus which reduces the expected abatement. Secondly, the low polluting crops cover a great proportion of cropland, reducing the possible abatement.

The Rhin river basin is similar to the Artois and Seine basins. It is a region with a majority of cropland leading to great gains (in production and profit) if the tax is reduced. However, in this region, the grassland area (and consequently livestock) increases instead of miscanthus, except when levels of the subsidy are high. The reallocation advantage is thus less advantageous than in the Seine and Artois basin. The situation in the Rhone basin is different. Less land is dedicated to crops (in percent of UAA) and the gain linked to a decrease in tax is less important. Moreover, the tax induces a slight crop reallocation in favor of the most polluting crop and only a high level of subisdy combined with a high level of tax significantly decreases the crop area and leads to a significant emergence of miscanthus. For these two basins, the mixed policy is cost-effective for NO_3. However, the substitution between livestock and cropland in the Rhin basin and the slight coverage of cropland (in % of UAA) limit the effect (about 0.5% of gain with a mixed policy).

In the Adour-Garonne river basin, contrary to other basins, the crops having natural losses higher than 40 k.N/ha represent less than 4 % of UAA. Moreover, these crop areas increase with a tax. Adding a subsidy does not bring the added extent of this area back under the initial size of it. In addition, we can note that cereals crops are less cultivated in this region, limiting the land reallocation effect. We find the same results for the Loire-Bretagne basin in the case of low tax levels.

5.5 Conclusion

We have assessed the improvement in nitrogen pollutant regulation under the combination of an N-input tax and a subsidy for a perennial crop.

We show that the policy mix is cost-effective in terms of water pollution (NO_3) and can lead to some levels of abatement which are difficult to reach (in terms of costs and "possibility")if the policy is only based on an N-tax .

A specific policy by basin is needed because of heterogeneity of crop emission and the diversity of crop allocation between the basins. Moreover, water pollution being local, the abatement targets are not necessarily equal between basins. This strengthens the need for specific land-use policy (for example, a subsidy for perennial crops) by basin.

5.6 Annexe

5.6.1 France Emission functions

Crops	a_{N_20}	b_{N_20}	a_{NH_3}	b_{NH_3}	a_{N0_3}	b_{N0_3}
bd	0.0232	0.7106	0.0537	1.6084	0.1081	36.3959
bl	0.0207	0.7923	0.0477	1.8510	0.0366	46.3375
bt	0.0216	0.0526	0.1109	0.3511	0.1412	18.3436
cz	0.0183	0.6038	0.0691	2.1157	0.1854	41.5223
ma	0.0214	0.0779	0.0810	0.2499	0.1998	28.3517
oh	0.0075	0.2624	0.0163	0.5724	0.0550	26.4918
pt	0.0202	0.0268	0.0469	0.0321	0.2552	2.6380
tr	0.0209	0.0105	0.0440	0.0560	0.1331	16.2587

TABLE 5.1 – Interpolated as linear function : $E_i = b + a * Nitrogen$; b,nitrogen in kg/ha/year

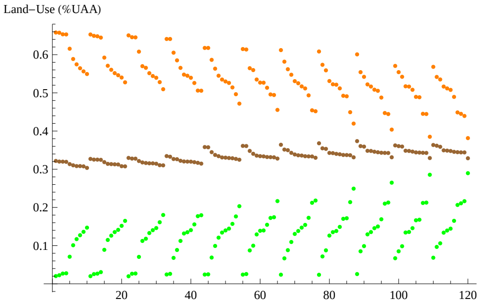


FIGURE 5.11 – Seine basin land-use share Miscanthus Grassland Total crops

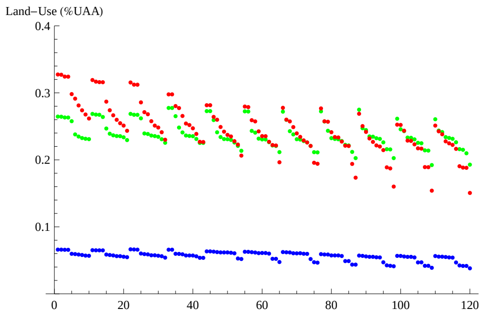


FIGURE 5.12 – Seine basin crop share low polluting crops medium high

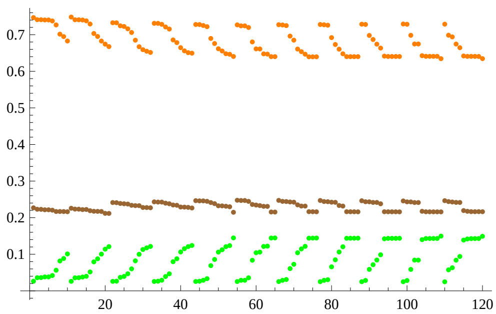


FIGURE 5.13 – Artois basin land-use share

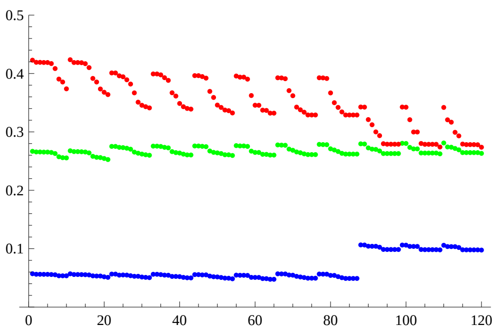


FIGURE 5.14 – Artois basin crop share

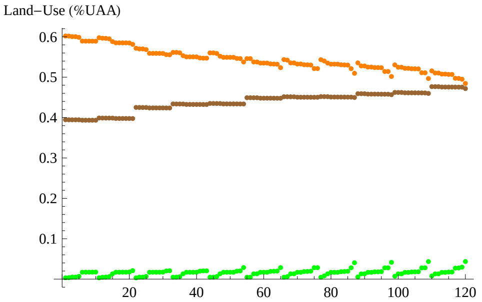


FIGURE 5.15 – Rhin-Meuse basin land-use share

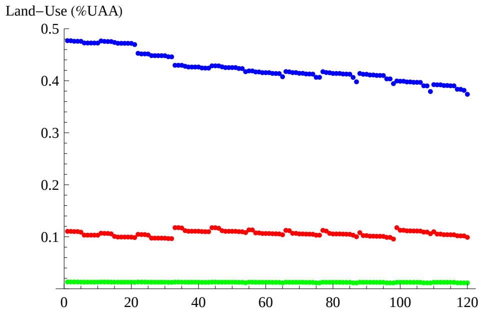


FIGURE 5.16 – Rhin-Meuse basin crop share

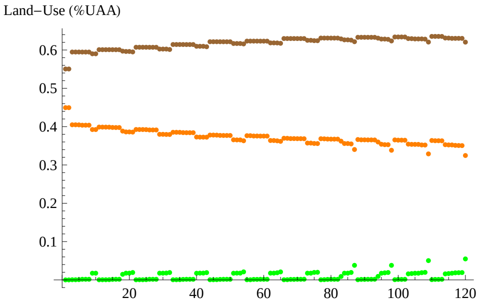


FIGURE 5.17 – Rhone-Méditerranée basin land-use share

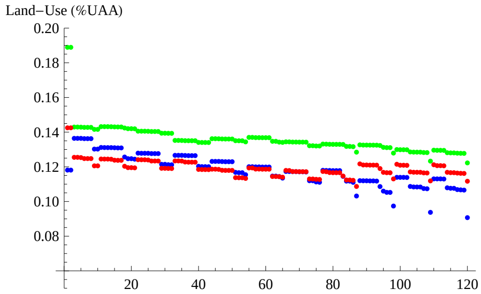


FIGURE 5.18 – Rhone-Méditerranée basin crop share

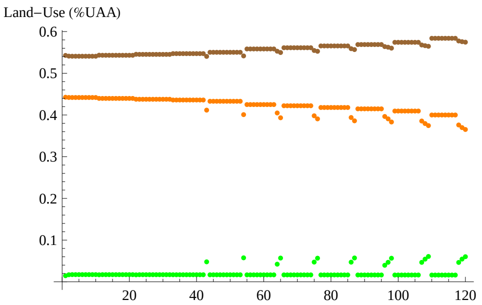


FIGURE 5.19 – Adour-Garonne basin land-use share

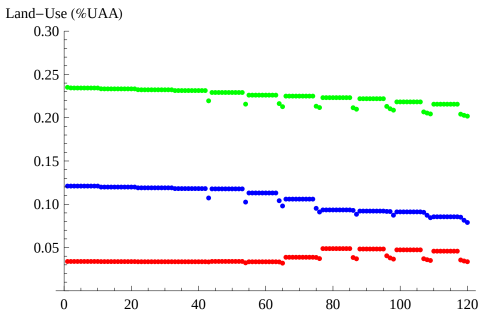


FIGURE 5.20 – Adour-Garonne basin crop share

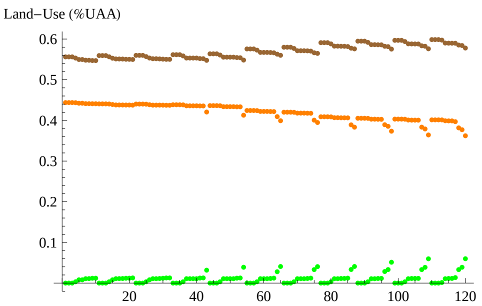


FIGURE 5.21 – Loire-Bretagne basin land-use share

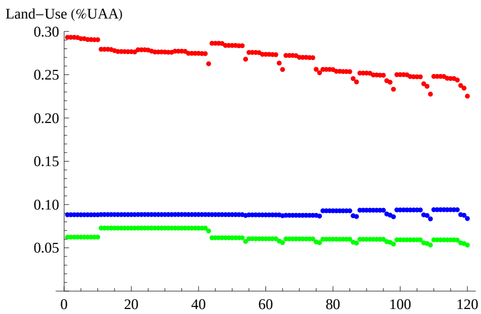


FIGURE 5.22 – Loire-Bretagne basin crop share

Crops	a_{N_20}						b_{N_20}					
	Artois	Seine	Rhin	Rhone	Adour	Loire	Artois	Seine	Rhin	Rhone	Adour	Loire
bd	0.0212	0.0216	0.0000	0.0230	0.0246	0.0220	0.8751	0.8365	0.0000	0.7480	0.6662	0.6966
bl	0.0198	0.0215	0.0197	0.0226	0.0205	0.0201	0.9170	0.7152	0.8317	0.6838	0.8586	0.8107
bt	0.0218	0.0213	0.0220	0.0226	0.0232	0.0231	0.0584	0.0424	0.0680	0.0917	0.0823	0.1045
cz	0.0178	0.0175	0.0199	0.0180	0.0216	0.0182	0.4979	0.6753	0.6177	0.6878	0.4636	0.5274
ma	0.0261	0.0265	0.0219	0.0064	0.0194	0.0252	0.0912	0.0856	0.0667	0.0289	0.0914	0.0774
oh	0.0000	0.0033	0.0163	0.0144	0.0191	0.0071	0.0000	0.1390	0.6295	0.4744	0.5745	0.2366
pt	0.0227	0.0215	0.0037	0.0102	0.0110	0.0210	0.0028	0.0303	0.0161	0.0099	0.0198	0.0604
tr	0.0192	0.0217	0.0198	0.0120	0.0205	0.0232	0.0764	0.0098	0.0086	0.0096	0.0084	0.0126
crop	a_{N_H3}						b_{N_H3}					
bd	0.0537	0.0483	0.0000	0.0566	0.0626	0.0390	2.0286	1.8718	0.0000	1.6497	1.6628	1.4092
bl	0.0376	0.0593	0.0641	0.0568	0.0703	0.0279	1.9503	1.9096	3.0990	1.8497	2.8686	1.1755
bt	0.0838	0.1155	0.1570	0.1280	0.2173	0.1412	0.1626	0.3940	0.6201	0.4821	0.8463	0.4860
cz	0.0950	0.0651	0.0499	0.0495	0.0683	0.0839	2.3147	2.1995	1.5542	1.8869	1.6576	2.3558
ma	0.1478	0.1221	0.1713	0.0275	0.0530	0.0797	0.5076	0.3221	0.3764	0.0821	0.2407	0.2249
oh	0.0000	0.0088	0.0514	0.0310	0.0426	0.0104	0.0000	0.3223	1.9020	1.0373	1.3983	0.3644
pt	0.0409	0.0682	0.0216	0.0291	0.0349	0.0376	0.0005	0.0515	0.0397	0.0057	0.0687	0.0528
tr	0.0320	0.0310	0.0390	0.0240	0.0642	0.0345	0.1558	0.1447	0.1712	0.0435	0.0264	0.0630
crop	a_{NO_3}						b_{N0_3}					
bd	0.0271	0.0385	0.0000	0.1327	0.0974	0.1143	13.5053	20.7712	0.0000	41.3432	16.5158	63.4847
bl	0.0010	0.0136	0.0378	0.0357	0.0659	0.0626	39.2732	44.3916	27.6987	22.8874	26.0977	66.1056
bt	0.1928	0.1228	0.0990	0.1038	0.1683	0.1478	18.6311	17.5251	21.1387	12.2787	8.4692	25.0989
cz	0.1819	0.1347	0.1461	0.1425	0.1866	0.2647	48.5449	36.6884	33.0033	28.3168	25.3778	53.9368
ma	0.3457	0.2862	0.1226	0.0589	0.1114	0.2966	30.7785	20.4566	31.5656	9.5284	14.6718	49.4585
oh	0.0000	0.0130	0.0747	0.1116	0.1714	0.0666	0.0000	12.0412	51.5851	39.8196	73.2135	27.8953
pt	0.3182	0.3823	0.0681	0.0748	0.2007	0.0988	0.1445	1.8484	0.0000	2.3200	5.3359	7.1029
tr	0.0202	0.1244	0.4090	0.1344	0.1591	0.1094	11.4561	17.8355	25.3069	8.8684	16.5017	17.4061

Chapitre 6

Farms, aquifer, and social value of nitrate damage : a modelling integrated approach

6.1 Introduction

Over the last several years, scientists and environmental agencies have been reporting increasing amounts of nitrate concentrations in water supplies, notably in the Unites States and Europe. For the most part, this pollution is due to agricultural activities, especially those within the farming sector (European Environment Agency, 2001 ; US Geological Survey, 1999).

To move beyond the handling of such water management problem according to administrative borders, the European Water Framework Directive (WFD, 2002) has been designed to tackle them as a function of natural river basin boundaries. This represents a paradigm shift towards integrated European water management and policy. As a result, the implementation of the WFD poses challenges to water managers, planning authorities, stakeholders, and researchers, increasing the demand for new tools including models that analyze, interpret, and display spatial information for river basin planning.

If we are to asses management strategies according to the WFD, hydro-economic models based on river basins must be implemented. From a hydrological point of view, these models properly reproduce the physical behavior of the systems with a realistic representation of the different surface and groundwater resources, including their interaction, the spatial and temporal variability of resource availability, and their impacts on nitrate concentration in the river basin, including the aquifer, in space and over time.

Hydro-economic modelling has been used for many decades to investigate water management in terms of surface and groundwater (Young and Bredehoeft 1972, Noel and Howitt 1982). More recently, significant advances have been made by applying this kind of model to problems arising in a case of non-point source pollution. However, most of these studies focus on river pollution rather than on aquifer pollution. (Aftab et al., 2010, 2007).

Hydro-economic modelling integrates different components (hydrologic, economic and agronomic) in order to better understand the impact of policies dealing with water pollution. However, when all aspects of water management are under investigation, the integrated approach leads to strong assumptions related to future water demand and irrigation. Moreover, among the empirical studies which deal with nitrate contamination, one of the three components is often neglected. To focus on different scenarios and compare their cost-effectiveness, these studies have relied on different methodologies. They may refer to (i) a percentage of emission abatement reached before a given date (Dellink et al., 2011; Brouwer et al., 2008), (ii) a mixed approach including conversion of intensive arable land into extensive grassland and livestock reduction (Volk et al., 2008) and input tax and set-aside management (Aftab et al., 2010), and (iii) enlarged approaches based on applied general equilibrium modelling including emission permit markets (Dellink et al., 2011; Brouwer et al., 2008), or (iv) the "Bayesian belief network" approach (Barton et al., 2008).

Within the framework of an applied study, the comparison of scenarios allows us to pinpoint the most cost effective one, but it is not the optimal result in that it does not arise from the maximization of a social planner's objective function. A theoretical approach based on over time trade-offs between farm profits and environmental damage makes it possible to determine the optimal tax path over time. However, this optimal path depends on many parameters, especially ones related to the environmental damage function which remain unknown.

To overcome the limits of the applied and theoretical approaches, we have set out to combine a theoretical model and a quantitative modeling chain based on a bio-economic model and a hydrological model. The applied models account for the set of farming systems and the impacted aquifer. Our study focused on the largest of the three aquifers within the Seine river basin.

The basic idea is to take the nitrate concentration target to be the resulting steady state level of a social planner's optimization programme. The interest of doing so is threefold : (i) we characterize the social value of damage related to the targeted concentration level of nitrate ; (ii) which leads us to design the optimal path consistent with the target ; and (iii) we can in turn assess "welfare losses" arising when the tax path deviates from the optimal one. To sum up, the marginal social value of damage in terms of nitrate concentration in the largest aquifer of the Seine river basin is assessed to be 1.1 € $mgNO_3/l\ (ha\ an)^{-1}$ when the concentration target is set at 50 $mgNO_3$/l. It is assessed to be 21.9 € $mgNO_3/l\ (ha\ an)^{-1}$ when the target is set at 38 $\mathrm{mgNO_3}$/l. In addition, we estimate the impact of taking into account the inherent variability of physical parameters across the aquifer which leads to a level of uncertainty increasing with the stringency of the target. We also show that applying a constant tax path instead of the optimal one leads to a discounted welfare loss of 0.5% for a concentration target of 38 mg/l. The discounted welfare loss can be as much as 2.3% when a tax path less objectionable to farmers meets the same target.

The paper is organized as follows. The modeling methodology is presented in section 2. The integrated approach and the damage parameter assessment are explained in section 3. In section 4, we compare estimates of welfare loss induced by various suboptimal scenarios. We conclude by highlighting how combining an applied and a theoretical approach makes it possible to estimate the marginal social value of damage related to given nitrate concentration.

6.2 The models

The modeling of our case study is presented in two parts, firstly the theoretical framework, and then the applied aspect involving a bio-economic model linked to a hydrological model.

6.2.1 The theoretical framework

The theoretical framework is based on a dynamic control model proposed by Bourgeois and Jayet (2010). A social planner maximizes the discounted sum of agricultural profit minus the damage value related to the apply of nitrogen fertilizers. Soil nitrogen losses play a role in the accumulation of nitrate in aquifers which is the pollution taken into account in our analysis.

Consider the set of farmers contributing to the nitrate pollution. Farming activity is represented by the demand for N-fertilizer denoted by x. Activity depends on performance characteristics summarized by a one-dimension θ parameter. The individual farm profit function $\pi(x, \theta)$ is defined for any feasible x and for any θ over the interval $\Theta = [\theta, \bar{\theta}]$. The probability density function denoted by $\gamma(\theta)$ is assumed to be strictly positive at any θ within the interval. The related cumulative function is denoted by $\Gamma(\theta)$. Accordingly, the per time total profit is expressed by $\int_\Theta \pi(x(\theta, t), \theta)\gamma(\theta)d\theta$. Usual assumptions apply to π, which is assumed to be twice continuously differentiable, increasing and concave with respect to the fertilizer input x.

Regarding the environmental impact and related damage, the standard framework dedicated to pollutant accumulation problems applies. The state of the aquifer is characterized by the nitrate concentration which is denoted by z. The dynamic evolution over time is the result of a double-side effect. On the one hand, the clearing effect takes the form of an usual exponential decline characterized by the rate τ. On the other hand, nitrates accumulate in the aquifer due to N-fertilizer consumed by any θ farm at any time t. At this point, a key aspect of our analysis comes into play : the introduction of a time lag characterizing the nitrate transfer between the top soil and the groundwater. At the top soil level, nitrate losses depend on fertilizers consumed by any θ farm at time t, and the related emission function is denoted by $e(x, \theta)$ (let us recall that x depends on t and θ). All top soil nitrates reach the aquifer, and the related nitrate accumulation depends on the aquifer thickness. For simplification, the time lag does not depend on θ and the aquifer thickness $1/a$ is assumed to be constant over the whole physical domain. The time evolution of the environmental system is described by equation (6.1).

$$\dot{z}(t) = -\tau z(t) + a \int_{\Theta} e(x(\theta, t-\beta), \theta)\gamma(\theta) d\theta \tag{6.1}$$

The aquifer nitrate concentration is expressed in the social planners objective through a damage function $D(z)$ which is assumed to be increasing and convex. The social planner is assumed to be perfectly informed about the aquifer characteristics and the transfer process as well as all individual farm activity at any time. His/her objective over time W is expressed by (6.2).

$$W = \int_0^{\infty} [\int_{\Theta} \pi(x(\theta, t), \theta)\gamma(\theta) d\theta - D(z(t))] e^{-\delta t} dt \tag{6.2}$$

Accordingly, his/her programme refers to (6.3).

$$\max_{x(\theta,t)} W \quad \text{subject to (6.1) and to boundary conditions} \tag{6.3}$$

Boundary conditions (i.e., known values of z when $t \in [-\beta, 0]$ and infinite time value of the shadow λ-price) and the problem solving are detailed in Bourgeois and Jayet (2010). Let us summarize by (R1) the set of equations useful for the present analysis.

$$\begin{aligned}
&\forall\theta,\ \forall t > 0\ :\ \pi_x(x^*(\theta,t),\theta) = a\ e_x(x^*(\theta,t),\theta)\lambda(t+\beta)e^{-\delta\beta} \\
&\dot{z}^*(t) = -\tau z^*(t) + a\int_\Theta e(x^*(\theta,t-\beta),\theta)\gamma(\theta)d\theta \\
&\dot{\lambda}^*(t) - (\tau+\delta)\lambda^*(t) = -D_z(z^*(t))
\end{aligned} \tag{R1}$$

The shadow price λ refers to the pollution state, i.e., z. The steady-state solution $(\bar{z}, \bar{\lambda})$ comes easily when $\dot{z} = 0$ and $\dot{\lambda} = 0$.

6.2.2 The agro-economic model

The investigation undertaken here relies on an updated version of the economic model, AROPAj, presented by De Cara et al. (2005) and updated by Galko and Jayet (2011). The model consists of a set of independent [1], mixed integer linear-programming models. Each model describes the economic behavior of a representative farmer (or a 'farm group') with respect to eligible crops, crop area allocation, animal numbers, and animal feeding. The farm group's programme is to maximize its gross margin with respect to a set of constraints. The set of constraints includes (i) crop rotation and agronomic constraints ; (ii) CAP-related constraints ; (iii) restrictions concerning animal demography and nutritional requirements ; and (iv) restrictions concerning quasi-fixed production factors (land and livestock). [2] Farm groups are assumed to represent the

1. Regarding France and the V2 version, used in this paper, 54 MILPs act for the Seine river basin.

2. Following De Cara et al. (2005), we assume in our central set of simulations that livestock numbers are allowed to vary within +/- 15 % of the values reported in the FADN

agricultural sector at the regional level given at the Farm Accountancy Data Network (FADN) is the grounds for calibration. In comparison to many bioeconomic models, the fact that we use a FADN sample of farms clustered into farm groups makes it possible to overcome data confidentiality and account for farm heterogeneity (e.g.,Aftab et al. (2010, 2007).

Changes occurring in the Common Agricultural Policy (CAP) from 2003 to 2007 have been taken into account. Among them, decoupling schemes are known to trigger significant changes in the European agricultural sector (Galko and Jayet, 2011). As is the case through France, the Seine river basin is obviously strongly impacted by changes in the CAP.

Balana et al. (2011) note that a large number of agri-environmental WFD-related studies are based on 'stylized' farms and consequently fail to capture the inherent heterogeneity of real-world farms, thereby sending wrong signals upstream in the decision making process. They suggest using actual farm data instead of 'stylized' farm design. Our methodology retains the advantages of these two approaches. On the one hand, farm groups recreate the agricultural sector at the regional scale thanks to the use of FADN data. On the other hand, the functional form model related to any farm group allows taking into account changes in economic policies such a tax on agricultural pollutants.

Coupling the economic model AROPAj with the crop model STICS (Brisson et al., 1998, 2003) captures the heterogeneity because soil characteristics and, more generally, pedo-climatic conditions are taken into account. Following the work by **?**, we calibrate nitrogen input to yield functions for the different farm groups and for most of the crop significants in terms of area and production. [3]

database.

3. It accounts for soft wheat, durum wheat, barley, maize, sunflower, rapeseed, sugar beet, and potatoes.

This functions are adjusted on a Misterliech-like exponential form. This allows the economic model to gain in realism when price change impacts the crop productivity at the plot level. As a counterpart, the economic model becomes non linear, but this potential difficulty is easily overcome by a two-step optimization procedure. It has to be noted that animal manure management is a source of nitrogen which accounts for a part of crop $N-$input in the model.

Finally, in addition to nitrogen input to yield functions, the crop model provides nitrogen input to nitrogen-loss functions. We focus here on nitrogen to nitrate functions when nitrate losses stand at the top soil level (root zone). Similarly to Aftab et al. (2010), the leaching functions are derived from STICS outputs when nitrogen inputs varies within a reasonable range. Nitrogen loss functions are adjusted on an affine base. In other words, the N-loss under the NO_3 form related to the crop j and the farm group k depends on the N-input through the relation (6.4). A and B being estimating for a given plant on a given soil .

$$[NO_3]_{j,k}(N) = A_{j,k}N + B_{j,k} \tag{6.4}$$

6.2.3 Coupling with the hydrological model

The use of the hydrological model MODCOU allows us to estimate the nitrate transfer of top soil toward the groundwaters at the Seine river basin level. The hydrological model MODCOU (Ledoux, 1980; Ledoux et al., 1984, 1989) computes the daily water balance using climatic data (rainfall, PET), the water flow to and in the river network, as well as the flow to, in, and between aquifer layers and the interactions between rivers and different layers of the aquifer. It is also able to compute the convective transfer of solute to, in, and between those layers.

The Seine river basin is characterized by the presence of several overlayed aquifers layers. The MODCOU application represents the whole Seine basin (95,560 km^2) and the three major aquifers designated in terms of the sediment the waters flow through (figure 6.1) : the Oligocene (sands and limestone), the Eocene (sands and limestone), and Chalk (Cretaceous chalk), with a spatial resolution varying from 1 to 8 km (Gomez, 2002; Gomez et al., 2003; Ledoux et al., 2007).

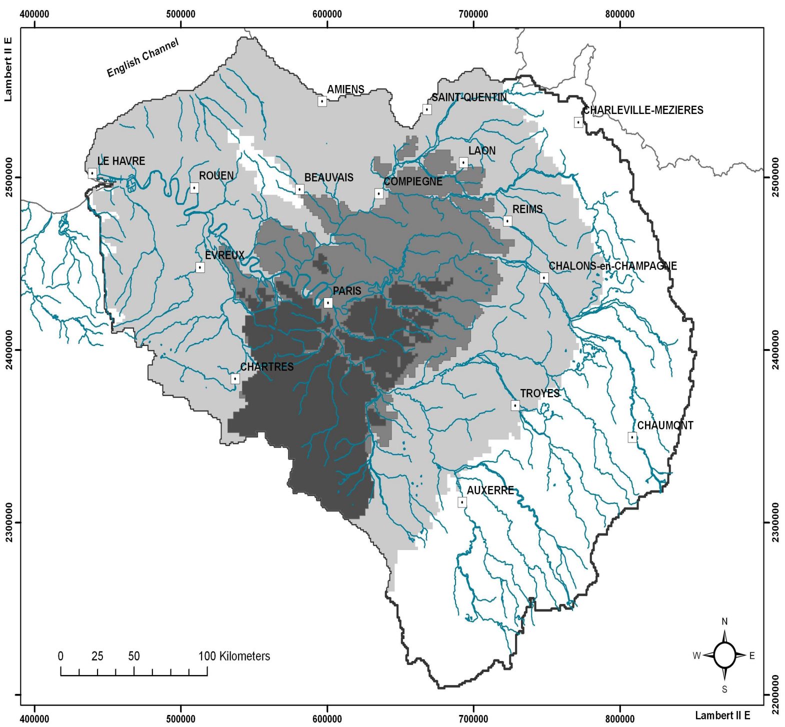


FIGURE 6.1 – The Seine basin (shape) and the main three overlayed aquifer layers, from top to bottom : the Oligocene (dark grey), the Eocene (grey) and the Chalk (light grey).

To estimate the nitrate contamination of the aquifer, the nitrate lixiviation flux from the agronomic model STICS (Brisson et al., 1998) is used. To do so, the STICS agronomic model is applied to the whole basin, and not only to on specific farm type as in AROPAj. The spatial variability of the agronomic fluxes is based on 150 Small Agricultural Regions (SAR) according to the database on agricultural practices established by Mignolet et al. (2007), as well as on the spatial variability of the pedologic and atmospheric conditions. The STICS-MODCOU simulation is described in Ledoux et al. (2007), and the update is available in Viennot P. (2007).

Coupling of the economic model AROPAj with the hydrological model requires that model input-output sets go through the probabilistic "spatialisation" of the AROPAj farm groups. The three-step spatialization process starts with the spatial econometrics model developed by Chakir (2009). Firstly, crop location is estimated in relationship to physical data at a very fine resolution level. Secondly, a cross entropy method refines crop location probabilities.In the third step, the FADN is linked up to the high resolution crop location, allowing us to estimate the location of farm groups on the same resolution grid. This three-step process is detailed in Cantelaube et al. (2012). The process is slightly refined when the data network related to the "Small Agricultural Regions" (SARs, which are partitions of FADN regions) is used in the last two steps.

The spatialization method provides the contribution to the regional (FADN or SAR) agricultural activity of each farm group on each cell of the grid. We are therefore able to distribute any AROPAj output over the geographical area, i.e., the Seine river basin. Soil-root nitrate losses related to fertilizer use are then distributed on a grid compatible with the spatial resolution of the hydrological model. Additional adjustments are needed to deal with the different time steps used by the models, i.e., the year in AROPAj and the day in MODCOU. One year denoted by the "reference" year ensures consistency across the models. The 2002-FADN based AROPAj calibration year has been selected. The nitrate losses estimated by AROPAj and STICS MODCOU in 2002 compare fairly well, therefore, the spatial and temporal evolution of the nitrate losses estimated by STICS-MODCOU in 2002 are used as a reference. Any other AROPAj run acts as an annual variation from the reference run, and this variation is homogeneously distributed within the year and over the entire basin to feed the MODCOU model.

Finally a “scenario” is defined as a set of time-ordered annual AROPAj inputs from the date 0 up to a given horizon T. Any of t-annual AROPAj runs provides a NO_3 lixiviated flux which is transfered to the groundwater thanks to the dynamic hydrological model MODCOU. More accurately, a scenario consists in a fertilizer tax path combined with a range of exogenous livestock adjustments. Scenarios may differ according to tax path as well as to the range of livestock adjustments. In terms of CAP context, we assume that the so-called Agenda 2000 holds up to and including 2006 and that the “decoupled Luxembourg scheme” holds as of 2007.

In this application of an integrated modelling approach, we focus on the largest of the three aquifers within the Seine river basin, the Chalk aquifer, and more precisely, only the free part of the aquifer, which is both the most affected by nitrate contamination and subject to water withdrawal. For our economic analysis, the NO_3 concentration in the aquifer is calculated as the annual median value for the entire Chalk aquifer.

6.3 The integrated approach

The theoretical model and the quantitative integrated model match when the damage function is explicitly set. To do so, we adopt the quadratic form (6.5).

$$D(z) = \frac{k}{2} z^2, \ k > 0 \tag{6.5}$$

The advantage is to represent the social marginal damage through the use of an unique parameter. At this step, this parameter is still unknown.

Thanks to considering the dynamic system at the steady state (R1), we can now relate the damage parameter k and the steady-state concentration $\bar{z}$ through the following system (R2).

$$\begin{aligned}
&\forall \theta \ : \ \pi_x(\bar{x}(\theta), \theta) = a \ e_x(\bar{x}(\theta), \theta)\bar{\lambda} e^{-\delta\beta} \\
&\tau \bar{z} = a \int_\Theta e(\bar{x}(\theta), \theta)\gamma(\theta) d\theta \\
&(\tau + \delta)\bar{\lambda} = k \ \bar{z}
\end{aligned} \tag{R2}$$

Rearranging the last equation of the system (R2) leads to the relation (6.6).

$$\bar{\lambda} = \frac{k \ \bar{z}}{(\tau + \delta)} \tag{6.6}$$

Let us reverse the roles of z and k and let us consider the concentration level $\bar{z}$ as the target set by the social planner. Assuming that the social planner as maximizes the social welfare leads to revealing the marginal social value of the damage $k\bar{z}$ through relation 6.6. Obviously this value implicitly depends on the structure of both the agricultural production (i.e., $\pi(x, \theta)$) and the pollutant emission (i.e., $e(x, \theta)$). This value also depends on the discount rate (δ) and the physical parameters of the hydrological system (τ, β, a).

Finally, let us implement the tax μ set on the pollutant emissions e. For any θ farm at any time t, the private optimal choice leads to solving the equation $\pi_x(x, \theta) = \mu\, e_x(x, \theta)$. Thanks to the first equation of the system (R1), this leads to the important but simple equation (6.7) linking this tax to the implicit price λ associated to the state variable z.

$$\mu(t) = ae^{-\delta\beta}\lambda(t + \beta) \tag{6.7}$$

Over the long term, it becomes immediately $\bar{\mu} = ae^{-\delta\beta}\bar{\lambda}$. This long-term nitrate emission tax equation can be transformed in order to highlight the link between the tax and the social marginal damage $k\bar{z}$, as shown by the relation (6.8).

$$\bar{\mu} = \frac{ae^{-\delta\beta}}{\tau + \delta} k\bar{z} \tag{6.8}$$

A tax set on soil-root nitrate losses can be easily implemented in the AROPAj model. Let us consider the scenario in which the tax is constant over time. The MODCOU model reaches the steady-state when top soil nitrate losses provided by AROPAj are applied on an annual basis and the simulation is run over a 100-year period. Let us repeat the simulation process (AROPAj and MODCOU) for a set of taxes. We obtain a set of steady state values, $\bar{z}$. We assume that these simulations include the steady state claimed to be the social planner's target. Finally, let us consider that economic and physical parameters δ, τ, β, and a are given or estimated. Equation 6.8 provides estimates of the k damage parameter as well as the social marginal value $k\bar{z}$ of the damage associated with the nitrate concentration in the aquifer.

We select the discount rate δ equal to 0.04.

6.3.1 Physical parameters on average and variance

The natural decline rate τ and the mean time lag β of nitrates transfered between the top soil and the groundwater are approximated thanks to the MODCOU model. Estimates are respectively $\bar{\tau} = 0.02\ y^{-1}$ and $\bar{\beta} = 20$ which means that around 2 % of the water and solute in the aquifer are renewed each year, and that there is a 20-year time lag before the pollution reaches the free part of the Chalk aquifer (Philippe et al., 2011). The average water column height acts as $1/a$ in the theoretical economic model. It is estimated to be $1/\bar{a} = 13.5\ m$ without porosity.

Indeed, the Chalk aquifer is characterized by strong heterogeneity. Moreover, errors in phenomena measurement, which are inherent in the case of quantitative models, lead us to randomize these two physical parameters. We distribute parameter values according to two probability distributions in order to delimit the k value. We select uniform and lognormal distributions, centered around the previous approximate values, ensuring that parameters retain positive values. The $1/a$ parameter follows the lognormal distribution such that $ln(\bar{a}/a) \mapsto \mathcal{N}(0, 0.18)$, and the β time lag follows the lognormal distribution such that $ln(\beta/\bar{\beta}) \mapsto \mathcal{N}(0, 0.25)$. Regarding the uniform distributions, the effective thickness of the aquifer, $1/a$, is distributed in the interval $[5, 22]$ meters and β is distributed in the interval $[10, 30]$ years. In total 10,000 simulations were run for the two sets of probability distributions. In contrast to these two parameters, the clearing rate τ is assumed to remain constant and homogeneous in the aquifer. These elements are summed up in Table 6.3.1.

Let us consider a given value of the state $\hat{z}$ and the tax level $\hat{\mu}$ which allows

Parameter	Distribution	Min	1st Quartile.	Median	Mean	3rd Quartile	Max
$1/a$	$ln(\bar{a}/a) \mapsto \mathcal{N}(0, 0.18)$	6.6	12.0	13.6	13.8	15.4	25.8
$1/a$	$1/a \mapsto \mathcal{U}[5, 22.2]$	5.0	9.3	13.6	13.6	17.9	22.2
β	$ln(\beta/\bar{\beta}) \mapsto \mathcal{N}(0, 0.25)$	8.2	16.9	20.0	20.6	23.6	64.3
β	$\beta \mapsto \mathcal{U}[10, 30]$	10.0	15.0	20.0	20.0	24.9	30.0
τ	constant				0.02		
δ	constant				0.04		

TABLE 6.1 – Distributions of parameter values play a role in the marginal damage assessment.

the aquifer to reach the state $\hat{z}$ when the tax $\hat{\mu}$ is constantly implemented over time. Let us consider this tax as the first best long-term tax thanks to the value of the damage, characterized by k. The mean value of the marginal social damage $\hat{k}\hat{z}$ is assessed through the relation (6.9).

$$\hat{k}\hat{z} = \underset{a,\beta}{E}[k]\hat{z} = \hat{\mu}(\tau + \delta)\underset{a,\beta}{E}[\frac{e^{\delta\beta}}{a}] \tag{6.9}$$

6.3.2 From nitrate target to social marginal damage value

Let us consider a set of I constant tax scenarios over 100 years as well as runs of the models AROPAj and MODCOU. We keep the I 2-tuples $\{\hat{\mu}_i, \hat{z}_i\}_{i=1,I}$. The tax $\hat{\mu}_i$ is set equal to $0.25(i-1)$ (€per kg$N$ in NO_3) and $I = 21$.Then we estimate k when the corresponding target NO_3 concentration in the aquifer is considered as the optimal social level. For each tax value and each of the two distribution densities, we did 10,000 calculations of k reflecting the heterogeneity of the physical parameters.. Figure 6.2 displays the monetary marginal damage ($k * z$) related to the concentration level taken to be the optimal level. A decrease in a target by 1 mgNO_3/l means that the marginal damage increases by 1.7 € $(mgNO_3/l\ ha\ an)^{-1}$. Focusing on the two given targets, 50 and 38[4] mg/l, we estimate the marginal damage respectively by 1.1 and 21.9 € $(mgNO_3/l\ ha\ an)^{-1}$, i.e., a dramatic increase in the marginal damage. When the targeted concentration level is 38 mg/l, the standard deviation of the marginal damage is estimated respectively by 9.1 and 6.5 € $(mgNO_3/l\ ha\ an)^{-1}$ when the parameter distributions follow uniform densities and lognormal densities.

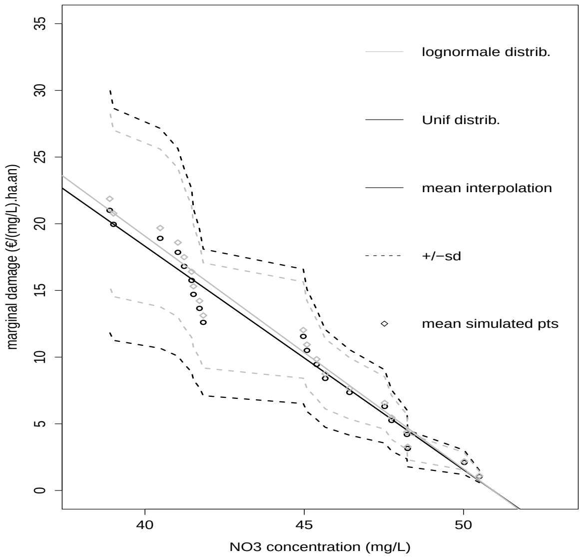


FIGURE 6.2 – Monetary marginal damage as a function of targeted NO_3 concentration level (in bold : a and $beta$ following uniform distributions ; in gray : a and β following lognormal distributions)

6.4 Welfare loss induced by non-optimal regulation

In this investigation, welfare is apprehended through including here farmers' profit, tax receipts and the environmental damage caused by nitrate in the

4.

aquifer. Once the z target is given and the $k * z$ marginal damage is estimated, the k estimate allows us to explicitly calculate the welfare. We compare welfare according to scenarios which differ in terms of time tax path and AROPAj livestock adjustment.

As a benchmark, the theoretical model leads us to assess the optimal time path of the tax when the targeted NO_3 concentration level is close to the steady state after 100 years and when the parameters of our problem are set at their average values as mentioned above. Other scenarios are designed to be more acceptable to farmers. In other words, time path of the tax differs. In addition, we consider that livestock may be adjusted, thanks to the tax on all top-soil nitrate lixiviation (mineral and organic source). As a consequence, we define four scenarios : (i) optimal nitrate tax path, (ii) constant tax over time, (iii) exponential tax path when the tax reaches 90 % of the long-term tax value after 10 years, and (iv) 90 % of the long-term tax value after 20 years. All policy scenarios are completed by a range of livestock adjustments from 0 to 15%, 30% and 45%.

The optimal tax path is achieved by solving the time differential system (R1) when the k-value is respectively related to the aquifer NO_3 concentration of 50 and 38 mg/l. The past path of the period $[-\beta, 0]$, required by the calculation, is provided by the MODCOU model. The 0-time $z(0)$ is the estimated 2002 NO_3 concentration in the aquifer. All policies converge over the long term toward the same nitrate concentration in the aquifer when the long-term tax is set at the steady state value and when the amplitude of the livestock adjustment is given. Offering the farmers the possibility of adjusting their livestock leads to significant changes in tax levels when the target is set at 50 mgNO_3/l. If the livestock adjustment increases, it is necessary to decrease the tax levels to reach the target. Different tax paths are represented in Figure 6.3.

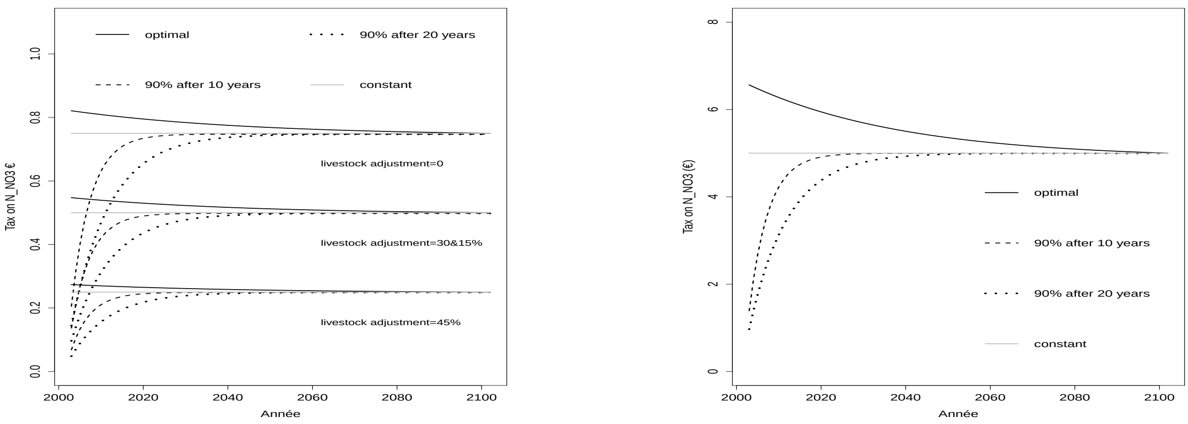


FIGURE 6.3 – Tax path over time, respectively for 50 mg/l and 38 mg/l as steady state NO_3 concentration

Figure 6.4, on the left, displays the tax refunded profit over time as regards our scenarios which differ in terms of tax path and livestock adjustment when the targeted nitrate concentration is $50mgNO_3/l$. As the livestock adjustment leads to a dramatic decrease in the long- term tax required to reach the target, the sum of farmers' profit and the refunded tax is higher when the livestock adjustment increases. Nevertheless, as regards economic effort, there is only a very slight difference between the scenarios and the path does not considerably impact farmers' profit. This is so thanks to the low level of taxes. Whatever, the given scenario, the evolutions of the aquifer nitrate concentration over time are very close. (Figure 6.5).

To sum up, when the target is not difficult to reach, an internal farm adjustment such as the livestock adjustment offers an opportunity to strongly attenuate the harshness of the policy tool, i.e., the tax on pollutant emission. Table 6.2 displays estimates of discounted welfare according to scenarios and targets.

Let us now consider the more stringent target : $38mgNO_3/l$. Figure 6.4, on the right, represents the farmers' profit path when the tax path and the range of livestock adjustment make it possible to reach this target.

Figure 6.5 displays the nitrate concentration paths which converge on either the 50 or 38 $mgNO_3/l$ target.

The implementation of a non-optimal tax path may lead to significant changes in profits, nitrate concentration and welfare over time. The worst of our scenarios would render the 2060 nitrate concentration $4mgNO_3/l$ higher than the optimal one. The lag strengthens the impact on a non-optimal scenario. Indeed, it takes about twenty years to see any impact of the regulation. However, if we compare the constant-tax scenario with scenario 3 (reaching 90 % of long-term value after ten years), we see that the profits induced by these two scenarios are equal after twenty years whereas the concentration levels are equal from 80 years onward (figure 6.5). Moreover, contrary to instantaneous pollution, in which case the social planner can manage 'at view', when lagged pollution is under consideration, the social planner cannot do so. The longer the pollution is delayed, the more important it is to act early and get as close as possible to optimal policy.

The calculation of welfare illustrates this point (figure 6.4). The figures display the welfare over time induced by each tax-scenario and livestock adjustment. For a given target of 50 mg/l, the difference of welfare due a non-optimal scenario is close to zero. The difference is due to a livestock adjustment. For a given target of 38 mg/l, we see that that the optimal taxation scenario is better after 2020 (in comparison to other scenarios with the same level of livestock adjustment). It is less efficient between 2003 and 2020 because profits are more heavily taxed whereas the tax does not have an impact on concentration levels (due to the lag). However, it is interesting to note that some non-optimal tax scenarios with high level of livestock adjustment are more efficient than the optimal tax scenario without livestock adjustment. This highlights the importance of not managing solely the mineral fertilizers.

Table 6.2 represents the discounted welfare in 2003 with a discount rate of 3%. The worst tax scenario shows a welfare loss of 2.3 % in comparison to the optimal tax scenario for a given target of 38 mg/l. A constant tax scenario, often under consideration in publications, leads to a welfare loss of 0.5 %.

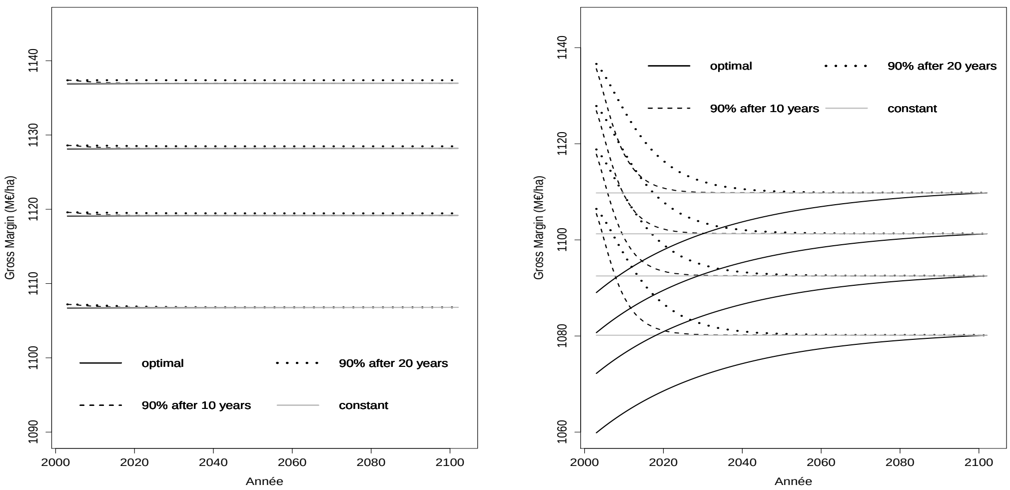


FIGURE 6.4 – Profit plus tax receipt over time related to the 50 mgNO_3/l (left) and the 38 mgNO_3/l (right) target, when the tax path is optimal (solid black), constant (solid grey), reaches exponentially 90% of the steady state tax in 20 years (dashed) and in 10 years (dotted), and when the amplitude of livestock adjustment changes from 0 (down) to 45% (top) of initial livestock by increment of 15%.)

6.5 Conclusion

We set out to assess the marginal social value of nitrate pollution in the Chalk aquifer in the Seine river Basin. The methodology involves reversing the role of z, the nitrate concentration in the aquifer, and k, the damage parameter. We

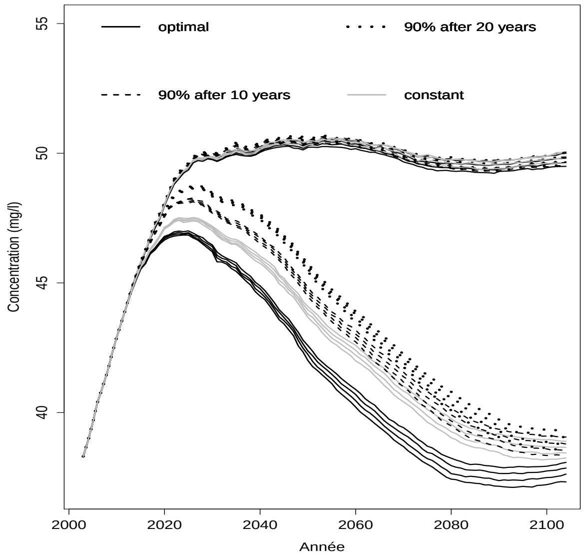


FIGURE 6.5 – Nitrate concentration path related to scenarios differing in tax path and livestock adjustement, regarding the two targets (long-term value of 50 and 38 $mgNO_3/l$ in the aquifer).

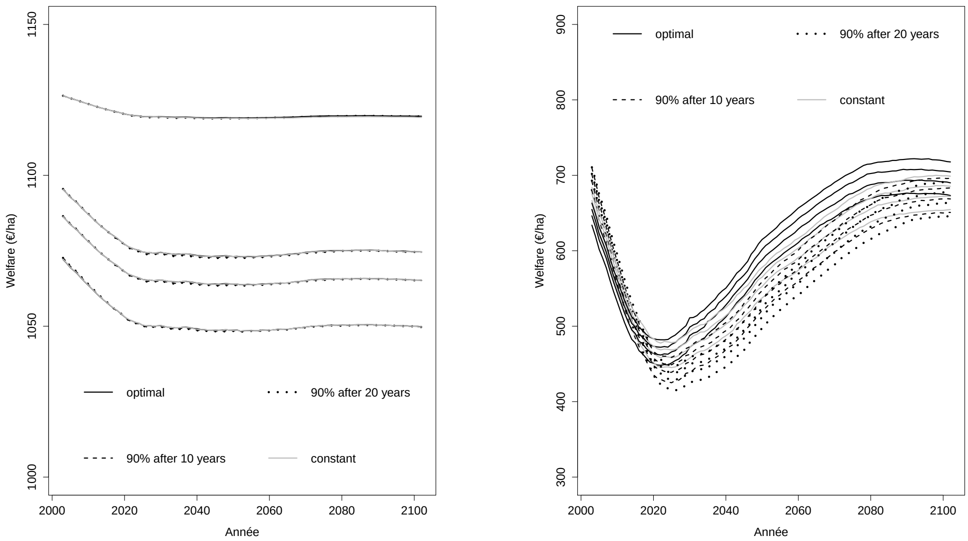


FIGURE 6.6 – Welfare when the nitrate concentration target is 50 (left) and 38 (right) $mgNO_3/l$

TABLE 6.2 – Loss of discounted welfare of each scenario compared to the optimal tax scenario without livestock adjustment for each NO_3 (mg/l) concentration target

Target(mg/l)	50				38			
kadj /scenario	optimal	constant	10ans	20ans	optimal	constant	10ans	20ans
0%	0.00	0.00	0.00	-0.01	0.00	-0.51	-1.41	-2.29
15%	3.83	3.82	3.82	3.81	2.64	2.08	1.14	0.24
30%	4.71	4.71	4.70	4.69	4.57	4.00	3.06	2.12
45%	7.92	7.91	7.91	7.91	6.47	5.88	4.90	3.95

consider the nitrate concentration target as the resulting steady state level of a social planner's program. To this end, we combine a theoretical model and a qualitative modeling chain based on bio-economic and hydrological models. We show that decreasing the target by 1 mg/l is equivalent to assessing an increase in marginal damage of 1.6 €.$(mg/l.ha.an)^{-1}$. When the targets are 50 and 38 mg/l, the associated marginal damage is respectively 1.1 and 21.9 €.$(mg/l.ha.an)^{-1}$, i.e., 16-fold increase. Moreover, the determination of k-value allows us to design the optimal path consistent with the target and assess the "welfare" losses arising when the tax path deviates from the optimal one. We show that applying the constant tax path instead of the optimal one leads to a discounted welfare loss of 0.5% for an aquifer NO_3 concentration target of 38 mg/l. The discounted welfare loss can reach 2.3% when a tax path more in favor of farmers meets the same target.

Conclusion

Les travaux présentés dans cette thèse ont pour cadre l'analyse micro-économique de la régulation des pollutions diffuses d'origine agricole. En particulier, la thèse a eu pour double objectif de définir les principaux déterminants permettant de définir une politique optimale de régulation des pollutions azotées puis à l'aide de modèles appliqués de quantifier et d'évaluer l'impact de ces politiques.

Nous avons d'abord définit un cadre permettant de mieux comprendre le rôle de l'allocation des terres dans la définition d'une politique optimale (Chapitre 2). Plus précisément, comme il s'avère impossible de taxer directement les émissions lorsque celles-ci sont diffuses, il convient de définir au mieux ce que doit être une politique de 'second-best'. Il s'avère alors que négliger les effets induits par une politique de taxation des intrants sur l'allocation des terres conduit à surestimer les bénéfices attendus d'une telle régulation et peut même augmenter la pollution. Si d'un point de vue théorique, une politique de taxe sur les fertilisants différentiée par culture couplée à une subvention sur le choix d'allocation des cultures rétablirait l'optimum, elle est clairement irréaliste sur le plan pratique. Une solution serait alors d'associer une taxe hétérogène à une subvention à des cultures à faible pertes azotées comme les cultures pérennes. Ces instruments sont alors implémentés dans le chapitre 5 où la subvention prend la forme d'une aide à la culture de miscanthus. L'intuition présente dans le chapitre 2, à savoir que ces instruments pourraient être coût-efficaces pour la pollution due aux nitrates, est alors clairement mise en évidence. Cependant, l'hétérogénéité spatiale des systèmes de production et des émissions associées étant relativement élevée, les gains d'une telle politique sont fortement variables. En vue d'une politique coût-efficace, celle-ci doit alors être adaptée par bassin versant.

Nous avons également approfondi l'étude de la pollution des aquifères due aux nitrates. Cette pollution entraîne des questions spécifiques provenant de la forte inertie du milieu physique considéré en raison du temps de latence pouvant être relativement important entre l'application d'engrais et la pollution de l'aquifère. Ce délai a un impact important sur la définition d'une politique optimale (Chapitre 3). Outre le stock de pollution optimale de long terme qui augmente avec le délai, un sentier de taxe plus fort est nécessaire pour atteindre ce stock. En terme de politique publique, cela peut conduire à un certain blocage, tant au niveau de l'acceptabilité de la mesure elle-même, car l'impact sur les dommages ne sera visible qu'une vingtaine d'année après le début de la régulation, qu'au niveau de la définition du sentier optimal.

Si l'on renverse le problème et que l'on qu'on considère que la concentration ne doit pas dépasser un certain seuil, comme c'est actuellement préconisé par la directive cadre de l'eau, cela signifie que la valeur que le régulateur accorde aux dommages est d'autant plus élevée que le délai est important. En outre, plus on veut atteindre un objectif ambitieux, plus il est coûteux en terme de bien-être de s'éloigner du profil optimal ou de retarder la régulation (Chapitre 6).

Les travaux présentés dans cette thèse permettent d'ouvrir des perspectives de recherche sur les thèmes d'utilisation des terres agricoles et des pollutions azotées associées. Si les chapitres 2 et 5 ont montré comment l'allocation des terres pouvait modifier la politique optimale visant à réduire la pollution due à un type de polluant azoté, la problématique relative à l'allocation optimale des cultures lorsque l'on considère l'ensemble des polluants azotés et notamment comment on résout l'arbitrage entre pollution atmosphérique et pollution de l'eau reste à définir. Dans le cadre de la pollution des aquifères, une problématique se pose relative à la différence entre les frontières administratives d'application d'une politique et la diversité du milieux. Dans notre travail, on a pris en compte les relations entre le secteur agricole et une masse d'eau représentée par un seul aquifère. En réalité, on devrait pouvoir étendre l'analyse au cas de plusieurs aquifère présent dans un même bassin versant, différenciée en terme de hauteur d'eau et de délais , ou temps de transferts, entre l'application de fertilisants et l'aquifère. Or, ce sont deux paramètres clefs dans la définition d'une politique optimale de régulation des nitrates (Chapitre 3). Ainsi une politique par bassin versant ne peut pas permettre d'atteindre un même seuil de concentration pour l'ensemble des aquifères et donc, leur valeur estimée par le régulateur diffère également. De même, le changement climatique en modifiant les hauteurs d'eau et le volume de précipitation peut avoir un impact significatif sur l'estimation des dommages associés aux nitrates.

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