Evaluating the efficiency of a N-input tax under different policy scenarios at different scales

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Abstract

Nitrate pollution from agriculture is an important environmental problem, caused by the excessive use of inorganic fertilizers. The internalization of this externality, via a tax on mineral nitrogen, could lead to a second best solution, reducing nitrate emissions. Several authors suggest that a reduction in agricultural support could produce similar results. In this paper we examine the effects of a nitrogen levy on nitrate pollution from agriculture in northern France under two different policy scenarios corresponding to (i) the Agenda 2000 and (ii) the Luxembourg reform of 2003, including the 2006 arrangement. The analysis aims at revealing what synergies or conflicts are created between a fertilizer levy and the policy scenarios, with respect to nitrate pollution mitigation. The applied methodology is based on the coupling of the economic model AROPAj with the crop model STICS. For each policy scenario, a nitrogen tax is simulated, involving different tax levels up to 100% the input price. Results reveal that at higher tax levels the reformed CAP can lead to slightly greater nitrate reductions than Agenda 2000, while the opposite applies when the tax is low. A down-scaling method is then used for the spatial distribution of the outputs, allowing for a more detailed representation of the nitrate abatement effects of the N-tax at different geographical levels.

Keywords: Crop model; mathematical programming; nitrogen response curves; nitrate pollution; nitrogen tax

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1 Introduction

The evolution of the agricultural sector in the European Union (EU) during the second half of the twentieth century has been accompanied by numerous environmental problems, one of which is the pollution of underground and surface waters with nitrates, due to the excessive use of nitrogen and phosphorous fertilizers. The existence of nitrates in soils reveals a trade-off effect: fertilizers serve as one of the main production factors for agriculture, but at the same time they are pollutants, producing a negative externality, which leads to a social welfare loss that is not compensated for and prevents the attainment of a Pareto optimal allocation of resources.

When emissions are measurable, the implementation of either a Pigouvian tax on the effluent itself or individual emission quotas can set pollution at the socially desired level and lead the economy to a first best optimal position, eliminating this externality. In the case of nitrates, which are a typical case of nonpoint source pollution, emissions from individual polluters are diffused and cannot be measured precisely, due to technical or economic limitations, thus hindering the use of the traditional emission-based policy instruments proposed by the literature. More specifically, nitrates also occur naturally in the soil as a product of the N-cycle (nitrification process), but to a different extent among the various geographic locations within the same region; this spatial heterogeneity of pollution production means that the same management for the same crop in different fields will not necessarily lead to similar nitrate losses. For example, shallow soils with a lower yield potential may contribute greatly to leaching even when fertilization is carefully planned (Beaudoin et al., 2005). Additionally, leaching does not depend only on the soil type but also on the weather conditions, which implies that random factors influence farmers' contribution to pollution. Hanley (1990) also stresses the great time lag between nitrate losses from a farmer's field and the consequent pollution of water bodies, which adds to the uncertainty about the true costs and benefits of pollution control. In order to implement emission based policy instruments, precise monitoring of nitrate leaching from each field through soil analyses is therefore required. Obviously, such an option is not feasible since the involved costs will outweigh any environmental benefits.

Due to the above problems, literature proposes indirect methods of nonpoint pollution control that focus on regulating or taxing input use. Concerning which polluting input should be taxed, inorganic nitrogen fertilizers are always the first that come in mind when dealing with nitrate pollution. This approach has the disadvantage of neglecting other sources of nitrate pollution, like animal wastes, but on the other hand it allows for better fertilization management and lower monitoring costs because the amount of nitrogen applied is known with accuracy. Fertilizer taxes change the value of marginal product of nitrogen and can therefore modify the outputs and the fertilization patterns of a profit maximizing farmer. On the other hand, regulating nitrogen fertilizer use is usually proposed within the best management practices framework that involves optimizing the application schedule and reducing the amount of fertilizer applied through per hectare quotas.

Other authors also suggest that the elimination of agricultural price support could help nitrate pollution abatement, since this normally leads to a reduction in the use of chemical fertilizers, due to the lower marginal value product of the input. An interesting setup to examine this claim is provided by the 2003 reform of the Common Agricultural Policy (CAP), which, unlike most previous reforms until Agenda 2000 that involved a piecemeal decrease in the support of most agricultural products, constitutes an unprecedented and radical shift of support regimes, by introducing full or partial decoupling of subsidies from production, thus posing a challenge for modelers attempting to analyze its impact on European agriculture and the environment.

In this paper we analyze the efficiency of a tax on mineral nitrogen, simulated in the Seine river basin region in France, under changes in the agricultural policy regime. More specifically, we examine whether the 2003 mid-term review reform provides by itself an adequate policy context for a decrease in nitrate emissions, as opposed to the previous CAP regime (described by Agenda 2000) and what kind of synergies or conflicts are created between the nitrogen tax and the two policy scenarios, with respect to nitrate pollution mitigation. For this we use a bioeconomic approach by coupling the agricultural supply model AROPAj and the crop growth model STICS. Initially, exponential nitrogen response functions are produced with STICS for each farm-group in AROPAj and are used as agronomic information for the latter. Nitrate production in soils is also modeled through the corresponding STICS module, allowing for the estimation of linear functions which relate nitrogen input and pollutants. A nitrogen tax is then simulated for both CAP regimes, followed by the analysis of its effects on fertilizer use and nitrogen transformations to nitrates at the root level. For every policy scenario, 21 simulations concerning continuously increasing the tax level up to 100% the input price are examined. Finally, the spatial distribution of the results is performed with a three-step downscaling method in combination with a Geographical Information System (GIS) package allowing for the visualization of the combined effects of the N-tax and the policy scenarios on nitrate losses.

2 Nitrogen, the nitrate problem and instruments for pollution control

2.1 Nitrates in the N-cycle and polluting effects

Nitrates occur naturally in the soil as products of the nitrogen cycle, but also have anthropogenic origins, the most important of which is the application of inorganic or organic fertilizers that directly add both ammonium (NH_4^+) and nitrate (NO_3^-) ions to the soil nitrogen pool. According to Bronson (2008), the nitrification process is strongly related with the soil concentration of ammonium ions, the largest source of which are the ammoniacal nitrogen fertilizers. The increase of livestock density, resulting in considerable production of liquid manure waste per area unit of cultivated land, has also contributed to the increase of nitrate concentrations in crop soils.

Nitrate ions not taken up by the crop may be leached from the soil and enter water bodies, polluting them. The consequences of nitrate pollution on the environment and human health are examined by numerous authors (Scholten et al., 2005; Ward et al., 2005; L'Hirondel et al., 2006) and environmental agencies have defined standards on water nitrate concentration. For example, the World Health Organization has proposed a concentration of 50 mg/l as the maximum contaminant level of nitrates in drinking water, while the EPA (Environmental Protection Agency) in the US has adopted the even stricter standard of 10 mg/l.

The first legislation text of the EU concerning nitrates was the Nitrate Directive, adopted in 1991, which aims at reducing and preventing water pollution by nitrates from agricultural sources. The Directive dictates that member States are responsible for identifying pollution sources, designating "vulnerable" zones in their territories and designing the appropriate action programs either at national level or specifically for each zone. Such zones are defined as areas that drain into waters which contain a nitrate concentration of more than 50 mg/l, or are likely to contain such concentrations if measures are not taken. Member States are also responsible for monitoring their action programs, in order to assess their effectiveness. For nitrate vulnerable zones, the Directive sets a maximum annual limit of 170 kg of nitrogen from livestock manure that can be applied per hectare. After the 2003 reform of the CAP, all provisions of the nitrate Directive have been included in the cross-compliance obligation for farmers, as a prerequisite for receiving the single payment.

Reducing nitrate concentration in ground waters is also an integral part of the new Water Framework Directive (WFD) of the EU. In fact, action programs under the Nitrates Directive are one of the basic measures of the WFD, which ranks nitrates among the basic pollutant sources of ground waters. In this context, the WFD defines that nitrates are one of the five core parameters to be monitored in order to ensure that their concentration does not surpass the 50 mg/l standard defined by the Ground Water Directive.

2.2 The least-cost framework for nitrate pollution control

An important aspect of any policy instrument designed for nitrate pollution control is the social cost involved in it, which includes, besides any measurable monetary values, the increase in social welfare (utility) due to pollution reduction, or equivalently, the decrease in the social damage brought about by nitrates. The problem is that the corresponding social damage function is usually either impossible to construct or very costly and difficult to estimate. Because of the problems associated with estimating potential costs and benefits of nitrate pollution control, the goal is not social optimality but rather to find ways of efficiently achieving desired environmental standards at least cost (Hanley, 1990). Such an approach, which does not necessarily lead to a Pareto optimal solution but will realize the emission reduction target in the least-cost way, was first presented by Baumol and Oates (1971) called the "Environmental Pricing and Standards" (EPS) system. It involves initially the establishment of a socially acceptable standard of environmental quality and then an iterative set of uniform taxes on either inputs or effluents that affect both outputs and pollution and are continuously readjusted through a "trial and error" procedure, until the environmental target is achieved. The EPS scheme has been widely discussed by numerous authors who identified its shortcomings and potential uses (Tietenberg, 1973; Abrams and Barr, 1974).

The question concerning the choice of an efficient policy measure to control nonpoint pollution has long been studied by economists and theoretical work during the last decades has shed light on the issue. Griffin and Bromley (1982) utilize the least-cost framework of the EPS approach to examine four different abatement policies of input regulation and input management incentives, under the assumption of complete information on emissions and polluters' returns. They conclude that, when properly specified, all policies are equally efficient, attaining the goal of least-cost pollution control. Shortle and Dunn (1986) build on the work of Griffin and Bromley and relaxe the strong assumptions of certainty by treating the runoff process and the weather as random variables. They examine four policy options for achieving agricultural nonpoint pollution abatement, namely standards and economic incentives on management practices, standards on emissions and economic incentives on estimated runoff. They argue that although neither policy achieves a first-best solution, management practice incentives are preferable under incomplete information on the runoff process, since they permit the farmer to fully utilize his experience and knowledge of his own farm operations.

An important issue concerning the selected nonpoint pollution control policy is whether its implementation should be differentiated or uniform. For example, uniformity of taxes requires that input use leads to the same marginal damages across all firms (Helfand et al., 2003), but in the case of nitrates this is not possible due to the different intensity of input use among farmers and the spatial heterogeneity of pollution production. Claassen and Horan (2001) also raise the issue of equity related to uniform second-best policies. Most authors agree that the selected nonpoint abatement policy will be optimal only if it is implemented in a non-uniform way (e.g. Griffin and Bromley, 1982; Shortle et al., 1998). This means that the optimal solution would be to introduce a policy measure regulating or taxing the use of every input for each of the concerned farmers. Obviously, this option may not be feasible, since imposing numerous different charges or quotas implies increased monitoring and enforcement costs. The adoption or not of such a policy instrument depends on the comparison of the welfare losses of a uniform implementation with the increased costs of the differentiated policy. Helfand and House (1995) find that the former leads only to minor welfare losses compared to their non-uniform implementation, a result which Shortle et al. (1998) find "unusual". Yet, empirical evidence provided by Martínez and Albiac (2006) verify the findings of Helfand and House.

Ambient taxes/subsidies have also received considerable attention in the literature. Their conceptual debut is attributed to Segerson (1988) who proposed an ambient incentive scheme that can achieve an efficient solution for controlling water quality, based on a Cournot-Nash equilibrium. This approach involves the continuous monitoring of the ambient pollution levels and the implementation of a uniform tax (penalty) for all producers, if the ambient pollution level exceeds the desired standard and a uniform subsidy (credit) when it is lower. Further contributions have examined the moral hazard issue related to ambient pollution based schemes (Xepapadeas, 1991) and the role of asymmetric information concerning the fate of effluents in designing such an instrument (Cabe and Herriges, 1992). Shortle et al. (1998) also identify equity problems that may limit its political acceptability, since firms that pollute the most may profit from the abatement efforts of others.

Empirical work on the economic performance of alternative instruments for nitrate nonpoint pollution control has produced ambiguous results. Gallego-Ayala and Gómez-Limón (2009) find that a quota in nitrogen fertilizer is the most efficient instrument for nitrate emissions control, while Semaan et al. (2007) find that incentives for better management constitute the most cost-effective method of reducing nitrate losses. Taylor et al. (1992) argue that no policy is optimal and actual results can vary even among farms within the same region and under the same weather conditions. Similar conclusions are drawn from the paper of Wu and Babcock (2001), while Goetz et al. (2006) propose that input taxes should be complemented by land use taxes for greater cost efficiency. Finally, as far as we know, there is no empirical work on ambient taxes/subsidies, only experimental designs (Spraggon, 2002; Cochard et al., 2005) that yield different results concerning the efficiency of the ambient instrument.

Reducing the level of agricultural prices support is considered an alternative indirect way of nitrate pollution abatement. For example, De Haen (1982) discusses the advantages and disadvantages of various nitrate pollution abatement policies, one of which is the reduction in support for agricultural products. He argues that this reduction should be significant in order to lead to lower fertilizer intensity. Abler and Shortle (1992), who use a partial equilibrium model to also examine the linkages between regulation scenarios of chemical inputs and farm commodity programs, show that the elimination of the latter can lead to a significant reduction in inputs.

Although a policy regime change can affect the use of chemical inputs, its impact on nitrate pollution may be less significant than initially expected. Wier et al. (2002) examine the environmental effects of the Agenda 2000 and find that, even though the reform leads to a decrease in inorganic fertilizer use, such a reduction in commodity prices has only a minimal effect on nitrogen losses, due to the combined effect of increase in animal production (more manure), crop mix change and a change in fertilization patterns.

The environmental implications of the 2003 Luxembourg reform of the CAP are still examined. Schmid et al. (2007) address this issue in their impact analysis of the reformed CAP in Austria and conclude that it will have a positive environmental effect, mostly due to land use changes and production output decline, which will in turn lead to a decrease in both animal wastes and in chemical inputs use and consequently to nitrogen losses. They also suggest that the cross compliance measures will have no effect in EU countries already applying agri-environmental programs with stricter standards. On the contrary, Mosnier et al. (2009) who examine the environmental impact of the reformed CAP on two arable farms in France, argue that decoupling can have positive effect on the environment only when accompanied by the cross-compliance measures. Finally, Gallego-Ayala and Gómez-Limón (2009) consider the 2003 reform of the CAP to be an important way of solving the nitrate problem, but it may be complemented with other policy instruments when the reduction in N emissions is not deemed sufficient.

2.3 A comment on input taxes

The main problem concerning the implementation of an input tax is the uncertainty on the abatement results achieved through limiting input use due to its increased cost. Although for a single crop the effect of an input tax is straightforward, when a farmer faces multiple crop production possibilities where each one is represented by different patterns of input use (production functions) and contributes differently to nitrate pollution, an input tax could actually lead to completely opposite results than the ones expected in theory. This can be demonstrated by using simple calculus techniques.

Let's assume a farmer's profit function:

$$Z\left(X_i\right) = \sum_{i=1}^{I} \left\{X_i \left[p_i Y_i - c\left(w, \boldsymbol{\beta}_i\right)\right]\right\}$$

In the above equation, Z denotes farmer's gross margin, x_i the surface allocated to crop i, p_i is the crop's price, Y_i is the yield of crop i and c represents the crop's variable cost function, which depends on the price of nitrogen, denoted by w and considered exogenous, and the vector of the other cost parameters, denoted by $\boldsymbol{\beta}_i$. Crop area, X_i, can therefore be written as a function of w, while p_i and vector $\boldsymbol{\beta}_i$ remain constant:

$$X_i = x\left(w, p_i, \boldsymbol{\beta}_i\right)$$

Y_i can also be expressed as a function of nitrogen used (N_i) and the vector of parameters $\boldsymbol{\alpha}_i$ describing the physical conditions that affect crop production.

$$Y_i = y\left(N_i\left(w\right), \boldsymbol{\alpha}_i\right)$$

The yield function is assumed to be concave with a diminishing marginal product, as is required by both crop science and economic theory. Under the hypothesis of a rational economic behavior and for a given w and p_i, the farmer uses for each crop i such a quantity of nitrogen (N_i^*) that equates its value marginal product to its price:

$$p_i \frac{\partial Y_i}{\partial N_i} (N_i^*) = w$$

This means that as w increases (e.g. as a result of a tax) and p_i remains constant, per hectare use of nitrogen decreases ($\partial N_i / \partial w < 0$) because the concavity of the yield function calls for reducing input use in order to achieve a higher marginal product.

Production of each crop i is also associated with a function E_i of per hectare nitrate emissions, which depends on the amount of nitrogen used and can be mathematically expressed as:

$$E_i = e\left(N_i\left(w\right), \boldsymbol{\gamma}_i\right)$$

with $\partial E_i / \partial N_i > 0$. Symbol $\boldsymbol{\gamma}_i$ denotes the vector of soil and climatic parameters that affect nitrate losses from crop i. In a specific region, ambient pollution, L, is equal to the sum of nitrate losses from every activity and is calculated as the product of E_i and the area X_i occupied by each crop:

$$L = \sum_{i=1}^{I} X_i E_i \tag{1}$$

Thus, nitrate emissions depend on the decisions taken both at the extensive margin (crop area, X_i) and the intensive margin (nitrogen used, N_i). However, although at field level an increase in w will always lead to a decrease in the amount of nitrogen, this may not be the case for the area, X_i, allocated to each crop. In fact, due to the different degree of change in crops' relative profitability, brought about by the increase in the price of nitrogen, an activity substitution may take place. This means that the derivative $\partial X_i / \partial w$ can take any sign, depending on the direction of changes at the extensive margin for crop i. To show this, we set for simplicity reasons $i = 1, 2$ and differentiate (1) with respect to w in order to get:

$$\frac{dL}{dw} = \left[\left(\frac{\partial X_1}{\partial w}\right) E_1 + X_1 \left(\frac{\partial E_1}{\partial N_1}\frac{\partial N_1}{\partial w}\right)\right] + \left[\left(\frac{\partial X_2}{\partial w}\right) E_2 + X_2 \left(\frac{\partial E_2}{\partial N_2}\frac{\partial N_2}{\partial w}\right)\right]$$

By rearranging terms and assuming an activity substitution by considering the decrease of X_1 to be equal to the increase of X_2 ($\partial X_1 / \partial w + \partial X_2 / \partial w = 0$):

$$\frac{dL}{dw} = \left[X_1 \left(\frac{\partial E_1}{\partial N_1}\frac{\partial N_1}{\partial w}\right) + X_2 \left(\frac{\partial E_2}{\partial N_2}\frac{\partial N_2}{\partial w}\right)\right] + \left[\frac{\partial X_2}{\partial w}\left(E_2 - E_1\right)\right] \tag{2}$$

The first part in brackets of equation (2) represents the results in nitrate pollution caused by intensive margin changes and the second the results due to extensive margin changes. The former is always negative, since $X_i > 0$, $\partial E_i / \partial N_i > 0$ and $\partial N_i / \partial w < 0$, implying that an increase in the price of nitrogen will always result in lower nitrate emissions from a single field, when no activity substitution is taken into account. However, the sign of the second bracketed part cannot be defined, as it depends on the per hectare nitrate emissions of each crop at the optimal nitrogen use level. If crop 2 pollutes more than crop 1 ($E_2 > E_1$), the sign is positive (we have assumed that $\partial X_2 / \partial w > 0$) and this activity substitution leads to an increase in nitrate losses. Consequently, the changes in the ambient pollution level will depend on whether the changes at the intensive margin are more significant than the ones at the extensive margin: In the first case, $dL/dw < 0$, while in the second $dL/dw > 0$.

3 Modeling nitrate pollution control policies

3.1 Bio-economic models

The majority of empirical work on modeling instruments for nitrate pollution control relies on farm-level, regional or social welfare bio-economic models, based on mathematical programming (MP) and includes static linear programming (LP), nonlinear programming (NLP) or dynamic programming specifications. MP models constitute an approach consistent with microeconomic theory, which is the maximization of income under constraints concerning the availability of fixed inputs, while at the same time they offer a quantitative representation of production technology. According to Janssen and van Ittersum (2007), a bio-economic model is "*a model that links formulations describing farmers' resource management decisions to formulations that describe current and alternative production possibilities in terms of required inputs to achieve certain outputs and associated externalities*".

The biophysical requirements of a bio-economic model are covered by specialized crop growth models that can relate crop yields, soil characteristics and input usage and are the result of the latest advances in agronomy, soil and crop science. However, when the main concern is to infer agronomic results to a regional level, such models seem too unwieldy to handle and are site-specific, depending on the microclimatic conditions and soil characteristics of specific fields. This means that they lack an economic dimension, as they cannot be applied at a larger geographical scale without serious assumptions on the physical data used. To overcome this problem and to link an agronomic with an economic model, two approaches can be identified in the literature:

The first approach uses the simulation results as inputs for the economic model. This takes the form of estimating technical and/or biophysical coefficients in order to either form appropriate constraints or to improve the specification of the objective function. Examples of this approach applied to nitrate pollution reduction are provided by Johnson et al. (1991), Taylor et al. (1992) and Semaan et al. (2007).

The second approach also uses the simulation results from the agronomic model in order to estimate response functions that relate yields and runoffs with the factors of production under consideration, *ceteris paribus*. These kinds of analytical expressions treat the former as dependent (endogenous) variables and consider all factors affecting them (climate, soil, cultivar etc.) as constants, except for a number of agronomic inputs, which are used as independent (or exogenous) variables. The derived response functions can then be directly incorporated inside a programming model. Concerning the study of nitrate pollution, this kind of coupling agronomic and economic models that are based on mathematical programming, allows for a more realistic and detailed representation of the functional relation between input use and effluent emissions and improves the analysis of the implementation results of any of the policy measures previously described. Examples of this approach include among others Helfand and House (1995), Larson et al. (1996) and Martínez and Albiac (2006). In this paper we follow the same approach and use of the crop growth model STICS in order to estimate nitrogen yield response and nitrate emission functions to be incorporated in the economic model AROPAj.

3.2 The economic model AROPAj

AROPAj is a short-term agricultural supply model, based on linear and mixed-integer programming, developed by the INRA[1] Agricultural Research Center at Grignon, France, in order to study the effects of the CAP reforms in French and European agriculture at different scales, from the farm to the EU level. An example of AROPAj use is provided by Jayet and Labonne (2005) who assess the impacts of the 2003 mid-term review of the CAP in France. AROPAj utilizes the Farm Accountancy Data Network (FADN) which is an EU database with micro-economic accountancy data from a sample of agricultural holdings, collected every year from national surveys. This EU-wide coverage of FADN gives the possibility to expand the utilization of AROPAj in order to include all EU member states.

Although FADN contains a sample of representative farms for each administrative region at national level, AROPAj maximizes an objective function for farm group types rather than individual farms, each having its own constraint set. This means that AROPAj actually consists of a set of independent models that describe the economic behavior of the corresponding farm type and represent the wide diversity of farming systems encountered in European agriculture, covering most annual crops, grasslands, and major animal production activities found throughout the EU. This farm typology is performed for each administrative region and involves a two-step aggregation (clustering) procedure that utilizes non-hierarchical methods and is based on three farm characteristics: (i) Farming type (14 types of farming activities, according to FADN nomenclature), (ii) location altitude (<300m, 300–600m, >600m) and (iii) economic size, as defined by the Economic Size Unit variable. One important remark is that for reasons of private data protection, each farm group should be associated with at least 15 farms in the FADN database. The derived farm group types (1074 in total for EU-15[2]) represent "average" farms of the same type and can be considered as homogeneous in terms of farming type, geographical location and altitude.

The objective function of AROPAj, the activities, the set of constraints and the shortcomings related to FADN use are described in detail by De Cara et al. (2005). The model maximizes gross margin for each farm group type and the activity set concerns crop area and output, animal numbers, animal production (milk and meat) and the quantity of the purchased animal feeds. The constraint set includes (i) crop rotation and agronomic constraints, (ii) restrictions concerning animal demography and nutritional requirements, (iii) restrictions concerning quasi-fixed production factors (land and livestock) and (iv) restrictions related to CAP measures. More specifically, crop rotations and agronomic constraints concern limited area allocation and average input use, while for animals, a demographic and a nutritional equilibrium are always in effect. CAP restrictions include production quotas or area limitations, while mutually exclusive discrete choices faced by farmers are modeled with the use of binary or integer variables.

Before an MP model, like AROPAj, is used for policy analysis, it must be validated in order to ensure that it is suitable for performing the task for which it was constructed. Hazell and Norton (1986) propose six different methods of validating a model, the most common of which is the production test, where model results are compared with observed values of the variables. Various calibration techniques are encountered throughout the literature, ranging from *ad hoc* solutions such as "flexibility" constraints that bound the variables to their observed level (Day, 1963), to even more robust methods like Positive Mathematical Programming (Howitt, 1995) that is

[1]The acronym stands for "Institut National de la Recherche Agronomique".

[2]In this study the V2 version of the model is used, which concerns the version developed for the GENEDEC program and is associated with FADN data for 2002 and EU-15.

specially designed to exactly calibrate programming models. All the above solutions consist on adding either nonlinear terms in the objective function or extra constraints. In the contrary, the calibration procedure in AROPAj is based on the re-estimation of one of the model's parameters subset through a combination of Monte Carlo and gradient methods, in order to minimize the difference between actual observations and model results for each farm group (De Cara and Jayet, 2000). The calibrated parameters include animal feeding requirements, grassland yields and maximal crop area shares.

3.3 Coupling AROPAj with STICS

One important characteristic of AROPAj is its modular structure that allows for the selective use of various modules, including a large range of policy tools, like quotas and taxes on inputs or outputs, as well as technical modules that take into account environmental considerations. The first technical module to be included in AROPAj concerned the estimation of greenhouse gas emissions (De Cara and Jayet, 2000; De Cara et al., 2005). The coupling of STICS with AROPAj is thus performed with a specific module that involves the replacement of average point yields for crops in each farm group type with a response function of nitrogen. This leads to the transformation of AROPAj from an LP to a NLP model, with respect to nitrogen, since the latter is now regarded as a variable and its optimal use is calculated endogenously, along with the corresponding crop yields.

STICS[3] (Brisson et al., 2003) is a crop growth simulation model, based on water and nitrogen balances and driven by daily climatic data, while utilizing soil characteristics and management practices as inputs. It consists of a number of modules, each dealing with a different set of biophysical functions, either above the soil (e.g. yield and biomass formation) or beneath it (e.g. water and nitrogen balances). A last module is dedicated to the simulation of management practices (irrigation, fertilization). For the present work, the nitrogen balance module is of great importance, as it gives the opportunity to estimate both nitrogen uptake by crops and nitrogen losses that ultimately lead to nitrate formation in the soil.

The objective of coupling STICS with AROPAj is to estimate a nitrogen response function to be incorporated in AROPAj. Following the methodology presented by Godard et al. (2008), which is based on the innovative work of Godard (2005) who first introduced the coupling of the two models, the response function selected is of exponential specification. Our contribution lies in the additional estimation of a linear function relating nitrogen applications and nitrate emissions for every [farm group type-crop] combination in AROPAj.

It has to be noted that, besides the exponential one, various specifications of yield functions can also be found in the literature, the most common of which are polynomial ones (Helfand and House, 1995; Larson et al., 1996; Martínez and Albiac, 2006). All of these functions are concave and increasing in the feasible region of production, implying diminishing marginal returns, thus abiding by the characteristics imposed by economic theory. The selection of an appropriate functional form has been the main subject of numerous scientific studies but it is suffice to say that the choice is still controversial (Berck and Helfand, 1990; Llewelyn and Featherstone, 1997). In fact, Frank et al. (1990) conclude that no functional form should be assumed a priori as there are situations where one would seem preferable over another. On the other hand, nitrate emission functions found in the literature include polynomial (square-root

[3]The acronym stands for "Simulateur mulTIdisciplinaire pour les Cultures Standard".

and quadratic) (Larson et al., 1996; Martínez and Albiac, 2006) or exponential specifications (Abrams and Barr, 1974).

Beginning with the exponential yield response function, it can be expressed as:

$$Y(N) = Y_{max} - (Y_{max} - Y_{min})\, e^{-tN} \tag{3}$$

In the above expression, Y denotes the estimated crop yield, Y_{max} the maximum attainable yield (under no nitrogen stress), Y_{min} the minimum yield (with no fertilization), t represents the curvature of the response function and N the quantity of nitrogen applied to the crop. The advantage of the selected exponential form is that it has all the necessary attributes of a well defined production function and at the same time it includes parameters that allow for agronomic interpretation.

The process for estimating nitrogen response functions comprises two steps, each performed for every crop of each farm group type. The first step involves providing various options for soil, weather and management data for STICS, which leads to a number of data combinations and consequently to an equal number of possible N-response curves. The second step consists on selecting a single appropriate response function from the previous set.

Concerning the first step, the main problem encountered when linking crop and economic models is the inconsistency of the data used in each of them. AROPAj utilizes aggregated data for farm group types, including costs (but not quantity) of fertilization, with no geographical reference (apart from the knowledge of the FADN region that they belong). On the contrary, STICS requires field-level information on soil, weather and management practices for each crop, which cannot be found in AROPAj. To overcome this problem, Godard et al. (2008) opted for the combined use of a number of European-level databases concerning soil and climate information, in addition to phenology and other crop characteristics :

- Daily weather data for the year 2002 corresponding to version V2 of AROPAj were retrieved from the MARS[4] project database; To associate climate data with each AROPAj farm group type, every weather cell was assigned an altitude value by overlaying the grid of the MARS database and that of the Digital Elevation Model of Europe[5] (DEM);
- Soil data for STICS were provided by the European Soil Database[6] (ESDB); To identify lands where the modeled crops can be cultivated, the map grid of the ESDB was overlaid with the CORINE[7] Land Cover (CLC) database. For every FADN region, the five most common soil types from the ESDB were chosen;
- Management options for the simulated crops included cultivar type, timing of the crop cycle, irrigation and nitrogen fertilization. For maize and sunflower whose cultivars vary greatly with respect to earliness, three varieties and one sowing date were chosen. For the other crops, one variety and three sowing dates were considered, since their cultivars share practically the same timing crop cycle;
- Two cases of irrigation were opted, namely fully irrigated or rain fed crop;

[4] The acronym stands for "Monitoring Agriculture from Remote Sensing". For more information: http://www.marsop.info/marsop3/.

[5] For more information: http://www.eea.europa.eu/data-and-maps/data/digital-elevation-model-of-europe.

[6] For more information: http://eusoils.jrc.ec.europa.eu/.

[7] The acronym stands for "Coordination of Information on the Environment". For more information: http://www.eea.europa.eu/publications/COR0-landcover.

- Concerning nitrogen fertilization, the types of chemical fertilizer, the total quantity used and the number of applications for each crop were based on expert knowledge. Furthermore, application dates were translated into phenological stages instead of calendar dates, in order to ensure an adequate fertilization schedule. In farms with livestock, organic nitrogen in soil was estimated from FADN, using a per animal head parameter of nitrogen loss and was simulated through an option in STICS which concerns complementary nitrogen sources (besides chemical fertilizers);
- Pea crop and a winter wheat were the two possible crops in the preceding year.

The sum up of all possible combinations involving data and management options produced 30 or 60 response curves[8] for each farm type and crop. These curves were produced by performing, for each possible data combination, 31 simulations with STICS that involved continuously changing the total quantity of nitrogen applied, from zero to a maximum level of 600 kg per hectare using a 20 kg step. Every set of points in the Yield-Nitrogen space produced by these 31 iterative simulations in STICS, was adjusted to the exponential function represented by equation (3) and parameters Y_{max}, Y_{min} and t were estimated.

The second step in the process of constructing N-response functions involves the selection of the appropriate response curve among the set of the 30 or 60 possible candidates. Initially, the derived curves for each crop were compared with the horizontal line, representing the average yield (constant) of the same crop in the selected farm group type; the curves that were below this reference yield were excluded. From the remaining ones, the curve that best satisfied the marginal condition of nitrogen use was finally selected: at the intersection point with the FADN reference yield, the value marginal product of nitrogen should be equal to its price.

$$\frac{\partial Y_j}{\partial N}\left(N_{intersection}\right) = \frac{w}{p_j}$$

In the above equation, p_j denotes the price of the crop j, $\partial Y_j / \partial N$ is the marginal product of nitrogen (the slope of the yield function at the intersection point) and w the price of nitrogen. In other words, the slope of the response function at the intersection point should be equal to the price ratio of nitrogen and crop j.

The estimation of the nitrate emission function follows that of the yield function. For each of the 31 simulations concerning increasing doses of applied nitrogen, STICS produces not only a yield value, but also values corresponding to nitrogen losses in the form of nitrates (NO_3-N), ammonia (NH_3-N) and nitrous oxide (N_2O-N) that result from the simulated cropping activity. These emissions are calculated by the nitrogen-balance module in STICS, which is able to simulate the physical processes of nitrification, volatilization and denitrification that occur in the soil-root system and produce each of these pollutants respectively. Concerning NO_3-N losses, the set of points produced by STICS that corresponds to increasing nitrogen doses can be adjusted to the following linear function:

$$e\left(N\right) = AN + B \tag{4}$$

The estimated parameters in (4) are A and B. The former represents the slope of the pollution function, i.e. the marginal contribution of the specified crop to NO_3-N emissions. Parameter B expresses the quantity of NO_3-N that is produced in a specific soil through the physical process

[8] For rain-fed crops, two irrigation options were considered (full irrigation and rain), while for crops with high water requirements only full irrigation was opted. Hence, 60 curves were created for the former and 30 for the latter.

of nitrification and independently of any kind of farming activities (no anthropogenic origins). Figure 1 presents the relation between yield and the nitrate pollution functions estimated from STICS.

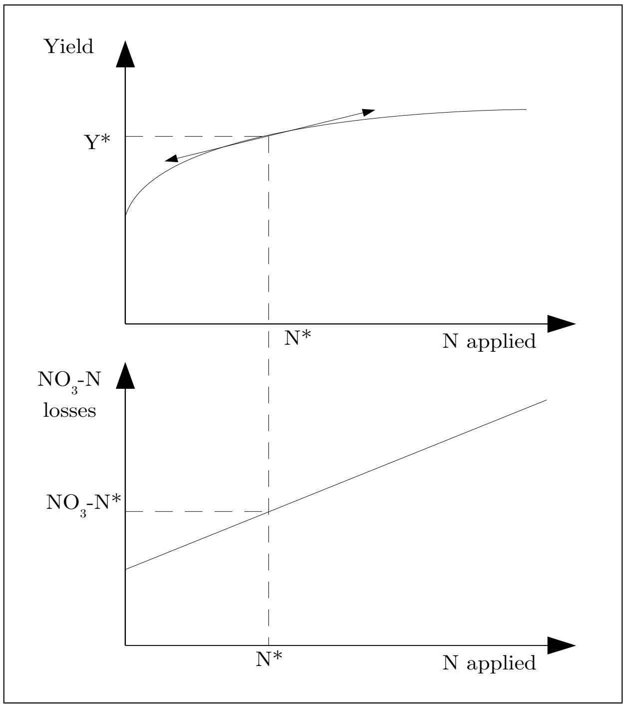


Figure 1: N-yield and nitrate emission functions estimated by STICS

3.4 Visualizing the results

Although the AROPAj farm typology allows for the representation of the diversity of European agriculture, it lacks a geographical dimension that would allow visualizing model's results, a feature that could be of great importance when trying to evaluate the impacts of environmental policies. For example, the spatial distribution of NO_3-N losses could possibly allow for more decentralized policies that take into account local geographical characteristics, climate and farming systems. This shortcoming is addressed by a method of a spatial disaggregation of the results produced by AROPAj, with the use of the GIS package ARCGIS. The method is based on the work of Chakir (2009) and follows a three-step procedure:

1. In the first step, the CLC grid is associated with the LUCAS[9] database, in order to estimate the probability $\bar{p}_{ij}$ that a j land use category from LUCAS exists in an i spatial unit (or map pixel) of the CLC grid. For this, a multinomial logit model was used along with additional variables concerning climate, soil and altitude.

2. In the second step, the probability $\bar{p}_{ij}$, estimated previously, provides prior information that permits the estimation of the posterior probability p_{ij} of locating the j FADN activity on the pixel i, with the use of a Generalized Cross Entropy model (Golan et al., 1996).

[9]The acronym stands for "Land Use/Cover Area frame Statistical Survey". For more information: http://eusoils.jrc.ec. europa.eu/projects/LUCAS/.

3. The third step involves the spatial distribution of the AROPAj farm groups, by estimating the probability π_{ik} of locating the k farm group on the i pixel, with the help of p_{ij} estimated previously and as conditional probability of the altitude class.

4 Case study

4.1 Description of the study area

This empirical work focuses on the Seine river basin region in northern France, which covers a surface of about 78,600 km^2 and totally encompasses the FADN regions of Île de France and Haute-Normandie and partially those of Champagne-Ardenne, Picardie, Centre, Basse-Normandie, Bourgogne and Lorraine. Its population amounts about 16 million inhabitants, representing more than a quarter of France's total population.

According to French national statistics for year 2000, 52% of the Seine river basin surface concerns arable lands, 12% permanent grasslands, 24% forest and bush areas and 12% urban areas and other types of land cover. About 15% of French farms are situated in the region, using 23% of the total available agricultural land in the country and having an average size of 68 hectares per farm. In terms of cropping activities, cereals and protein crops are the dominant cultures, taking up about 43% of the total agricultural surface in the region, while animal production concerns mostly cattle (Schott et al., 2009).

This information shows that agriculture is the principal activity of the Seine river basin in terms of land use and appears as a major source of pollution of local water bodies with nitrates. In fact, the Seine river basin has been designated a vulnerable zone, as defined by the Nitrate Directive, portraying an average annual increase of nitrate concentration in ground waters of about 0.64 mg/l over the last thirty years (Viennot et al., 2009). For this reason, since 1989, researchers from various disciplines participating in the project PIREN have studied the ecosystem of the Seine river basin in an attempt to better understand how its special characteristics (geology, vegetation, climate and human activities) affect the quality of water streams in the region.

4.2 Simulating an N-tax in the Seine river basin

Starting from the policy design examined in section 2.3, we consider a scheme based on an input tax. In our study, a tax on mineral nitrogen is simulated in the study region, under two policy scenarios, namely the Agenda 2000 and the Luxembourg (mid-term review) CAP regimes, in order to examine how the levy impacts on NO_3-N losses and whether the CAP reform can actually provide an adequate context of achieving pollution reduction. For every policy scenario, 21 simulations concerning continuously increasing the tax level up to 100% the input price are examined, using a 5% increase step. The tax is modeled as an increase in nitrogen price in AROPAj, which, due to its non-linear form after its coupling with STICS, is able to provide results in both the intensive margin (input use) and the extensive margin (activity levels).

As explained previously, a linear nitrate emission function was estimated for every crop in each farm group type, making per hectare NO_3-N losses from crops an endogenous variable. For other types of land use (e.g. fallow and grasslands), AROPAj takes account of low intensity nitrogen fertilizer applications and the corresponding nitrate losses through the use of appropriate parameters. On the contrary, for the "subsidized set aside" activity, appearing only in the Agenda 2000 scenario, no nitrogen applications were considered and thus no NO_3-N losses were calculated.

Since the tax is imposed on the nitrogen content of the fertilizer and not on the fertilizer itself, the actual monetary value of nitrogen needs to be estimated. In the simulations performed with STICS, different kinds of fertilizer were used for each crop, which, although desirable from an agronomic point of view, cause problems in the modeling of a nitrogen tax: the use of different fertilizers with different prices and different nitrogen contents does not allow for a single answer on the actual value of nitrogen. Therefore, for simplicity reasons, we considered a reference fertilizer per crop and as the market of nitrogen is hypothetical, the estimation was based on the price and content of each reference fertilizer type. More specifically, the price of the fertilizer was divided between the relative content of its basic components (N, P and K), yielding an average nitrogen value of about 1 €/kg, depending on the crop.

Finally, an important note is that the nitrogen-balance module in STICS calculates losses of NO_3-N only at the root level of the soil-crop system. However, the pollution of water bodies from nitrates constitutes a dynamic procedure that involves a significant time lag between the emission and its polluting effect and, in addition, it is difficult to predict and simulate the actual fate of the nitrate ion after it leaves the upper soil layers, due to the various random parameters that affect it (most commonly weather conditions and the variability of soils). In the PIREN project, these simulations are produced by the hydro-geological model MODCOU that allows for the reproduction of the hydro-dynamic behavior of the river basin and simulates the transfer of pollutants on the various soil components. However, such kinds of simulations are beyond the scope of this work, which has a pure economic orientation and concerns the efficiency of an input tax to control NO_3-N losses. The methodology used for this purpose provides a static image of the nitrates that are produced in the root zone, as a result of farming activities and the N-cycle, implying that the estimated NO_3-N losses concern only the geographical location that they were produced. Therefore, no predictions can be made about the consequent pollution of surface or ground waters in the region.

5 Results and discussion

5.1 Results on nitrogen fertilizer use

The implementation of a tax on mineral nitrogen is anticipated to have an impact on both the intensive and the extensive margin. For the former, Figure 2 describes the change in fertilizer use as the N-tax increases, revealing that a tax level of 100% the input price leads to an almost similar reduction in inorganic nitrogen fertilizer use in both policy scenarios (50.4% and 51.2% for the Agenda 2000 and the Luxembourg scenario respectively). The above result implies that the demand for fertilizer with respect to the price of its nitrogen content is inelastic, since a percentile increase in the price of nitrogen leads to a lower percentile decrease in fertilizer consumption. The result is generally in line with the existing literature, although the actual fertilizer demand changes due to a tax may vary: For example, Berntsen et al. (2003) find that a similar N-tax will lead to a reduction of 23-28%, depending on soil type. However, this difference can be explained by the fact that their estimated price of nitrogen is 0.67 €/kg, which is significantly lower than our 1 €/kg. On the contrary, Schou et al. (2000) find that a 100%

N-tax will result in a significant reduction of fertilizer consumption, ranging from 40% to 80%, depending on the soil type and the farm system examined, although their estimation of the value of mineral nitrogen is even lower than that of Berntsen et al., reaching only 0,42 €/kg (3.15 DKK).

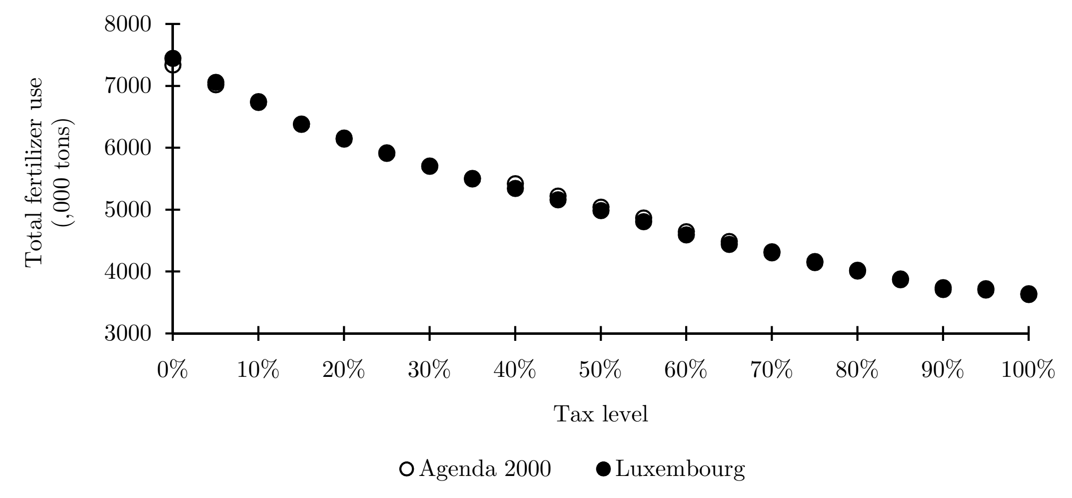
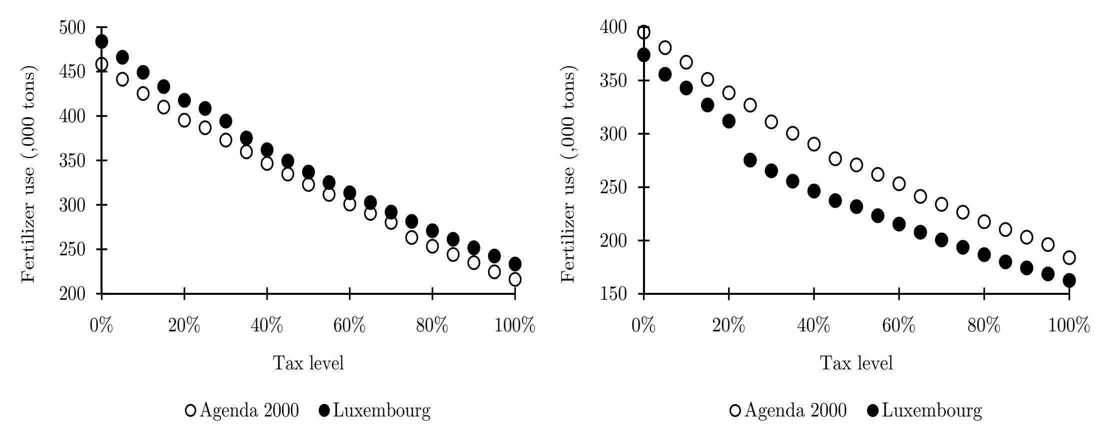


Figure 2: Total fertilizer use under different N-tax levels

What is interesting is that our model predicts only a weak change in total fertilizer use for passing from one policy scenario to another and that both policies have a similar reaction to the N-tax. Obviously, this contradicts the existing literature which suggests a reduction in input use due to the abolishment of agricultural support programs. However, this aggregate picture can be misleading since each region of the Seine river basin differs greatly from the other in terms of fertilizer use. This is clearly shown in Figure 3 that presents the effects of decoupling and of the N-tax in two FADN regions within the Seine river basin area, namely Île de France (FADN region 121) and Basse-Normandie (FADN region 135).



(a) Île de France (b) Basse-Normandie

Figure 3: Regional fertilizer use under different N-tax levels

The two FADN regions presented in the above figures represent different changes in fertilizer use when passing from one policy regime to the other. More specifically, fertilizer use in Île de

France increases slightly with decoupling, while in Basse-Normandie decreases up to a maximum of about 16% (at an N-tax level of 25%). This difference can be explained by examining the production orientation of the farm group types comprising each region: Île de France produces mostly arable crops, with soft wheat, maize, barley, rapeseed and sugar beet taking up about 85% of the region's total agricultural land. For a zero tax level, the passing on to the Luxembourg scenario leads to a slight increase in the surface allocated to all these crops, which now take up about 93-94% of total land, while the yields remain practically unchanged. Most of the extra hectares devoted to these crops come from the abolition of the set aside (subsidized) regime that existed under the Agenda 2000. Finally, the implementation of the N-tax results in a similar percentile fall of crop yields in both policy scenarios, thus explaining the increased use of fertilizer under the new CAP.

On the other hand, Basse-Normandie produces mainly livestock and at the same time 35% of total agricultural land is covered by permanent meadows. The Luxembourg scenario leads to a slight increase in the livestock units raised and in land allocated to permanent meadows, while fallow lands are doubled. This result combination leads to a reduction in fertilizer use under the Luxembourg scenario for all N-tax levels. The above presentation clearly shows that decoupling does not guarantee a reduction in input use. Most importantly, the type of farm group (production orientation) is the most important factor affecting fertilizer use, possibly neutralizing any decoupling effects

5.2 Results on NO_3-N losses

As anticipated, NO_3-N losses decrease with the increase of the N-tax in both policy scenarios, but Agenda 2000 initially leads to slightly lower NO_3-N losses compared to the Luxembourg scenario (Figure 4). Past the point of about 10% tax level, the Luxembourg scenario outperforms Agenda 2000, without however showing any important percentage performance difference, even at a tax level of 100% the N price. At this tax level, the reduction in nitrate production is estimated at 17.5% for the Agenda 2000 and 20.5% for the Luxembourg scenario.

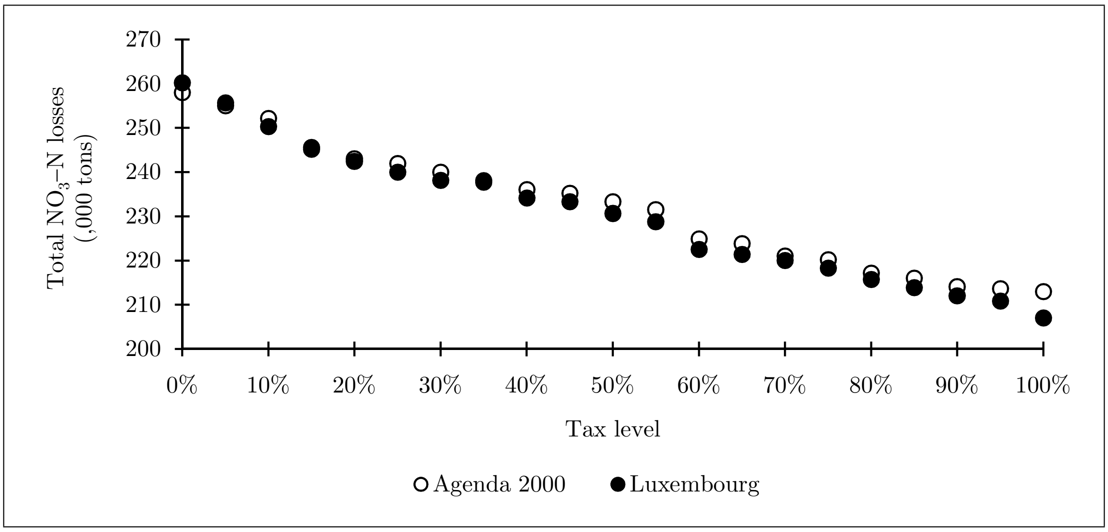


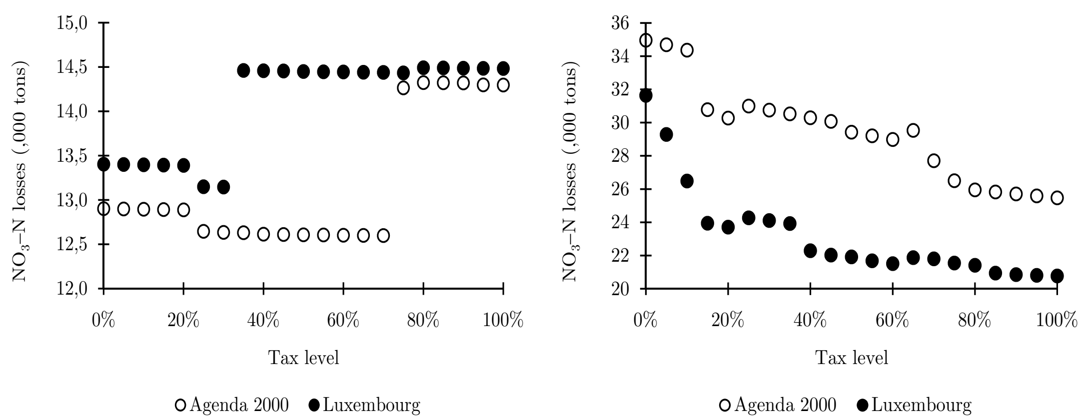
Figure 4: Total NO_3-N losses under different N-tax levels

These results show that nitrate response to an N-tax is inelastic, which is in accordance with previous studies; for example, Schou et al. (2000) find that a similar tax will result in a 20-22%

reduction in nitrate leaching, depending on farm type. These results indicate that decoupling contradicts the objectives of an N-tax when the latter is kept at rather low levels, while synergistic effects between the two measures appear at higher tax levels.

Nitrate emissions portrayed in Figure 4 can be viewed in more detail in the maps presented in figures I to IV, which can be found in the Appendix and were produced with the spatial disaggregation procedure described in the methodology section. More specifically, figures I and II portray NO_3-N losses under a zero tax level for the Agenda 2000 and the Luxembourg scenario respectively, where it is evident that the effect of policy change is negligible. Similar remarks can be made about figures III and IV that present NO_3-N losses for the two policy scenarios under a tax level of 100%.

As in the case of fertilizer use, this aggregated picture can be better explained by examining the variability of NO_3-N losses observed between FADN regions, due to the changes in the extensive and the intensive margin, brought about by the N-tax at different geographical levels. Disaggregated results therefore indicate, once again, that the observed nitrate reduction in the Seine river basin is the sum up result of these changes, which can vary both in magnitude and direction. For example, in Lorraine (FADN region 151), the implementation of the N-tax leads to a decrease in nitrate emissions (although not monotonic), yet this decrease is greater under the Luxembourg scenario, which implies that there is a synergistic effect between the N-tax and decoupling. On the contrary, in Île de France (FADN 121), NO_3-N losses under Agenda 2000 are lower than under the Luxembourg scenario and most importantly, for both policy scenarios, after an initial slight decrease, the N-tax actually leads to a significant increase in nitrate emissions (Figure 5).



(a) Île de France (b) Lorraine

Figure 5: Regional NO_3-N losses under different N-tax levels

The reason for this unexpected increase in NO_3-N losses lies in the changes brought about by the N-tax in the extensive margin, i.e. the surface allocated to each crop, and can be explained by the theoretical point discussed in section 2.3. Examining these changes in both policy scenarios (Figure 6), it is clear that the sudden increase in NO_3-N losses occurs simultaneously with the significant increase in the area allocated to soft wheat and the reduction in that of maize. At the same time, rapeseed area also increases slightly, while sugar beet remains constant, but still takes up about 20,000 hectares more than in the Agenda 2000 scenario.

This means that the increase in nitrate emissions in both scenarios, despite the implementation of the N-tax, is the result of the crop substitution effect, caused by the change in the relative profitability of the farming activities in the region. More specifically, as the N-tax increases, and due to the different yield functions of each crop, the changes in crops' gross margins lead to an increase in area allocated to the more profitable ones, which, however, contribute more to nitrate emissions than their predecessors. For a single crop, the tax always reduces emissions at field level (per hectare losses). However, the substitution of activities with more polluting ones will result in higher NO_3-N losses from the same field under a similar tax level. At an aggregated level, the tax-induced reduction in nitrogen use in these crops is not enough to compensate for the increase in the surface of crop land, resulting in the subsequent increase in nitrate emissions. The above discussion shows that at the local level, crop substitution, caused by an input tax, may have a more significant effect on NO_3-N losses than the reduction of input use itself.

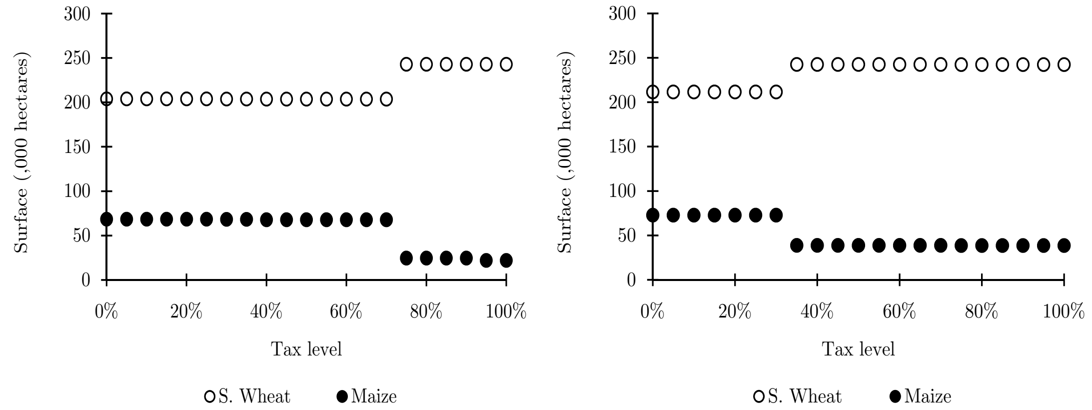


Figure 6: Crop surface area changes under different N-tax levels in Île de France

One solution for eliminating this substitution effect is the land use tax proposed by Goetz et al. (2006) as a complement to the N-tax. The former could be imposed on the crops that contribute the most to nitrate emissions, so that the N-tax would actually lead to a monotonous decrease of nitrate losses, regardless of the geographical level examined.

5.3 Assessing the cost-effectiveness of the N-tax

An important aspect of any policy instrument designed for nitrate pollution control is its social cost, which, for the case of the N-tax, is defined as the difference between foregone income from the producers' side and the amount of money received by the policy maker to implement the measure, plus the increase in the social welfare (nitrate pollution reduction). Transaction, administration, control and enforcement costs are also important, but are not considered in the analysis. Since this increase of social welfare cannot be measured, any cost comparison between the two policy scenarios must follow the least-cost framework of Baumol and Oates (1971): The most cost-effective policy regime will be the one with the lowest unitary nitrate abatement cost at any N-tax level, for a given social value of the nitrate damage, or given the pollution reduction target (i.e. local or regional nitrate reduction).

In empirical studies found in the literature of nitrate pollution economics, the examined policy instruments (or regimes in our case) are often compared with a monetary value that refers to the unitary cost of water treatment for reducing nitrate concentration in water at the desired levels (Horner, 1975; Goetz et al., 2006). Since water treatment constitutes an alternative to field-level pollution control, we can make the strong hypothesis that this cost represents the value that the society attributes to the nitrate problem, i.e. it represents the society's marginal damage function. Furthermore, it provides a good proxy of the costs involved in off-site abatement choices and allows the comparison with on-site policy instruments of nitrate pollution control.

In our case, total nitrate emissions are calculated at the root level locally and not at a single catchment area or an aquifer. This means that the "treatment-cost" approach is rendered inapplicable, since we cannot estimate the actual water pollution that will result from the calculated NO_3-N losses. On the other hand, the least-cost framework of Baumol and Oates requires the establishment of a socially acceptable pollution standard, which commonly refers to nitrate concentration in drinking or underground waters and therefore the same caveat applies as above. As a result, the assessment of the cost-efficiency of the two policy scenarios should be simply based on the comparison of the marginal fiscal abatement cost with respect to specific different tax levels, i.e. the difference between producers' foregone income and the policy maker's fiscal receipts.

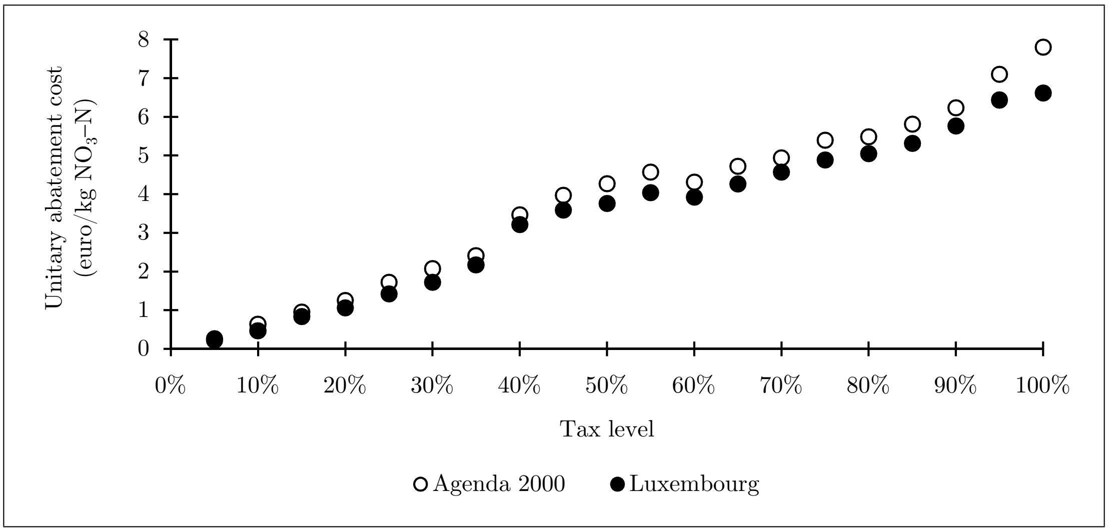


Figure 7: Unitary abatement cost under different N-tax levels

Figure 7 clearly shows that the Luxembourg scenario is most cost-effective than Agenda 2000, leading to a lower unitary abatement cost, even at the low tax levels, where NO_3-N losses under Agenda 2000 are lower (Figure 1). More importantly, the performance difference between the two scenarios increases with the N-tax, reaching almost 1.2 €/kg NO_3-N at a 100% tax level.

6 Conclusions

In this paper we study the effects of different levels of an N-tax on nitrate emissions from agriculture in the Seine river basin region in France and examine how the reformed CAP scenario affects the efficiency of the tax with respect to its preceding regime, the Agenda 2000 scenario. The methodology is based on the coupling of the economic model AROPAj and the crop model STICS and takes into account the spatial heterogeneity of nitrate emissions. Due to the nonpoint source nature of nitrate pollution, input taxes are often used as instruments of nitrate pollution regulation, even though they constitute a second best solution that includes considerable uncertainty on the true costs and benefits of the corresponding abatement effort. This is shown in this paper, using very simple calculus and it is empirically verified by the simulations performed with AROPAj, which involve continuously increasing the N-tax up to a level of 100% the input price.

At the aggregate regional level, fertilizer use and nitrate losses decrease as the N-tax increases in both scenarios. Examining the effect of policy change, we observe that the passing on to the Luxembourg scenario reduces total fertilizer use only marginally. On the other hand, nitrate losses are more affected by policy change, since Agenda 2000 leads to slightly fewer losses at low tax levels, while the Luxembourg scenario is more efficient at higher levels. This shows that the N-tax policy and the CAP reform lead to synergistic effects on nitrate reductions as the former increases.

An important conclusion derived from the simulations is that the effects of the N-tax on fertilizer applications and nitrate emissions differ locally and depend on the prevailing farming activity in the examined region: Results indicate that in regions with arable crops it is more difficult to define the exact direction of change in nitrate losses, since the decrease in nitrogen use is sometimes overlapped by extensive margin changes that involve an increase in areas allocated to more profitable, but at the same time more polluting, crops. This is clearly shown in the case of Île de France where a paradoxical increase in NO_3-N losses occurs exactly at the tax level that a partial substitution of maize with soft wheat is observed. On the other hand, in regions with an important animal sector, the N-tax leads to reduced nitrate emissions, since the crop substitution effect is less significant and mostly involves replacement of arable crops with fallow land or meadows.

Although our model operates at the regional (FADN) level, results suggest that the passing on to more disaggregated levels will probably reveal an even greater variability on NO_3-N losses. A similar variability will be observed in the cost effectiveness of the N-tax at different geographical scales, since a farm level implementation implies a more precise abatement policy at increased cost, while a uniform one may be less effective, yet less costly. Hence it is evident that the uniformity option should be ruled out, as it fails completely to capture any geographical nitrate emissions variability. Additionally, applying a different N-tax in each examined region may increase its cost-efficiency and possibly limit the impact of crop substitution in NO_3-N losses. Similarly, a land use tax on the most polluting crops is proposed for eliminating the crop substitution effect. Examining the relative efficiency of these two options presented above is beyond the scope of the present work; however both merit further study since they provide a promising framework for increasing the efficiency of an input tax as a means to achieve nitrate pollution abatement.

References

Abler, D. G. and Shortle, J. S. (1992). Environmental and farm commodity policy linkages in the US and the EC. *European Review of Agricultural Economics*, 19:197–217.

Abrams, L. W. and Barr, J. L. (1974). Corrective taxes for pollution control. an application of the environmental pricing ond standards systems to agriculture. *Journal of Environmental Economics and Management*, 1:296–318.

Baumol, W. J. and Oates, W. E. (1971). The use of Standards and Prices for protection of the environment. *The Swedish Journal of Economics*, 73(1):42–54.

Beaudoin, N., Saad, J. K., Laethem, C. V., Machet, J. M., Maucorps, J., and Mary, B. (2005). Nitrate leaching in intensive agriculture in Northern France: Effect of farming practices, soils and crop rotations. *Agriculture, Ecosystems and Environment*, 111:292–310.

Berck, P. and Helfand, G. (1990). Reconciling the von Liebig and differentiable crop production functions. *American Journal of Agricultural Economics*, 72(4):985–996.

Berntsen, J., M., B., Petersen, Jacobsen, B. H., Olesen, J. E., and Hutchings, N. J. (2003). Evaluating nitrogen taxation scenarios using the dynamic whole farm simulation model FASSET. *Agricultural Systems*, 76(3):817–839.

Brisson, N., Gary, C., Justes, E., Roche, R., Mary, B., Ripoche, D., Zimmer, D., Sierra, J., Bertuzzi, P., Burger, P., Bussière, F., Cabidoche, Y. M., Cellier, P., Debaeke, P., Gaudillère, J. P., Hénault, C., Maraux, F., Seguin, B., and Sinoquet, H. (2003). An overview of the crop model STICS. *European Journal of Agronomy*, 18:309–332.

Bronson, K. F. (2008). Forms of inorganic Nitrogen in soil. In Schepers, J. S. and Raun, W. R., editors, *Nitrogen in Agricultural Systems*, volume 49 of *Agronomy Monographs*, pages 31–55. American Society of Agronomy, Maddison, Wisconcin.

Cabe, R. and Herriges, J. A. (1992). The regulation of non-point-source pollution under imperfect and asymmetric information. *Journal of Environmental Economics and Management*, 22(2):134–146.

Chakir, R. (2009). Spatial downscaling of agricultural land-use data: An econometric approach using cross entropy. *Land Economics*, 85(2):238–251.

Claassen, R. and Horan, R. D. (2001). Uniform and non-uniform second-best input taxes. *Environmental and Resource Economics*, 19:1–22.

Cochard, F., Willinger, M., and Xepapadeas, A. (2005). Efficiency of nonpoint source pollution instruments: An experimental study. *Environmental and Resource Economics*, 30:393–422.

Day, R. H. (1963). On aggregating linear programming models of production. *Journal of Farm Economics*, 45(4):797–813.

De Cara, S., Houzé, M., and Jayet, P. A. (2005). Methane and Nitrous Oxide Emissions from Agriculture in the EU: A Spatial Assessment of Sources and Abatement Costs. *Environmental and Resource Economics*, 32:551–583.

De Cara, S. and Jayet, P. A. (2000). Emissions of greenhouse gases from agriculture: the heterogeneity of abatement costs in France. *European Review of Agricultural Economics*, 27(3):281–303.

De Haen, H. (1982). Economic aspects of policies to control nitrate contamination resulting from agricultural production. *European Review of Agricultural Economics*, 9:443–465.

Frank, M. D., Beattie, B. R., and Embleton, M. E. (1990). A comparison of alternative crop response models. *American Journal of Agricultural Economics*, 72(3):597–603.

Gallego-Ayala, J. and Gómez-Limón, J. A. (2009). Analysis of policy instruments for control of nitrate pollution in irrigated agriculture in Castilla y Léon, Spain. *Spanish Journal of Agricultural Research*, 27(1):24–40.

Godard, C. (2005). *Modélization de la réponse à l' azote du rendement des grandes cultures et intégration dans un modèle économique d'offre agricole à l' échelle européenne: Application à l' évaluation des impacts du changement climatique*. PhD thesis, UMR Economie Publique, INRA Paris-Grignon, France.

Godard, C., Roger-Estrade, J., Jayet, P. A., Brisson, N., and Bas, C. L. (2008). Use of available information at a european level to construct crop nitrogen response curves for the regions of the EU. *Agricultural Systems*, 97:68–82.

Goetz, R. U., Schmid, H., and Lehmann, B. (2006). Determining the economic gains from regulation at the extensive and intensive margins. *European Review of Agricultural Economics*, 33(1):1–30.

Golan, E., Judge, G., and Miller, D. (1996). *Maximum Entropy Econometrics*. Wiley, Chichester, UK.

Griffin, R. C. and Bromley, D. W. (1982). Agricultural runoff as a nonpoint externality: A theoretical development. *American Journal of Agricultural Economics*, 64(3):547–552.

Hanley, N. (1990). The economics of nitrate pollution. *European Review of Agricultural Economics*, 17:129–151.

Hazell, P. B. R. and Norton, R. D. (1986). *Mathematical programming for economic analysis in agriculture*. MacMillan, New York.

Helfand, G. E., Berck, P., and Maull, T. (2003). The theory of pollution policy. In Mäler, K. G. and Vincent, J. R., editors, *Environmental Degradation and Institutional Respones*, volume 1 of *Handbook of Environmental Economics*, pages 249–303. Elsevier, Amsterdam, The Netherlands.

Helfand, G. W. and House, B. W. (1995). Regulating nonpoint source pollution under heterogeneous conditions. *American Journal of Agricultural Economics*, 77(4):1024–1032.

Horner, G. L. (1975). Internalizing agricultural nitrogen pollution externalities: A case study. *American Journal of Agricultural Economics*, 57(1):33–39.

Howitt, R. E. (1995). Positive Mathematical Programming. *American Journal of Agricultural Economics*, 77:329–342.

Janssen, M. and van Ittersum, M. K. (2007). Assessing farm innovations and responses to policies: A review of bio-economic farm models. *Agricultural Systems*, 94:622–636.

Jayet, P. A. and Labonne, J. (2005). Impact d' une réforme de la Politique Agricole Commune par le découplage. *Économie et Prévision*, 167(1):101–116.

Johnson, S. L., Adams, R. M., and Perry, G. M. (1991). The on-farm costs of reducing groundwater pollution. *American Journal of Agricultural Economics*, 73(4):1063–1073.

Larson, D. M., Helfand, G. E., and House, B. W. (1996). Second-Best Tax Policies to Reduce Nonpoint Source Pollution. *American Journal of Agricultural Economics*, 78(4):1108–1117.

L'Hirondel, J. L., Avery, A. A., and Addiscot, T. (2006). Dietary nitrate: Where is the risk? *Environmental Health Perspectives*, 11(8):458–459.

Llewelyn, R. V. and Featherstone, A. M. (1997). A comparison of crop production functions using simulated data for irrigated corn in western kansas. *Agricultural Systems*, 21(4):521–538.

Martínez, Y. and Albiac, J. (2006). Nitrate pollution control under soil heterogeneity. *Land Use Policy*, 23(4):521–532.

Mosnier, C., Ridier, A., Kphaliacos, C., and Carpy-Goulard, F. (2009). Economic and environmental impact of the CAP mid-term review on arable crop farming in South-western France. *Ecological Economics*, 68(5):1408–1416.

Schmid, E., Sinabell, F., and Hofreither, M. F. (2007). Phasing out of environmentally harmful subsidies: Consequences of the 2003 CAP reform. *Ecological Economics*, 60(3):596–604.

Scholten, M. C. T., Foekema, E. M., VanDokkum, H. P., Kaag, N. H. B. M., and Jak, R. G. (2005). *Eutrophication Management and Ecotoxicology*. Springer Berlin Heidelberg, Berlin, Germany.

Schott, C., Mignolet, C., and Benoît, M. (2009). *Agriculture du bassin de la Seine*, volume 5 of *La Collection du Programme PIREN-SEINE*. Agence de l' eau Seine-Normandie, Nanterre, France. Available from http://www.sisyphe.upmc.fr/ piren/fascicules.

Schou, J. S., Skop, E., and Jensen, J. D. (2000). Integrated agri-environmental modelling: A cost-effectiveness analysis of two nitrogen tax instruments in the Vejle Fjord watershed, Denmark. *Journal of Environmental Management*, 58(3):199–212.

Semaan, J., Flichman, G., Scardigno, A., and Steduto, P. (2007). Analysis of nitrate pollution control policies in the irrigated agriculture of Apulia region (southern Italy): A bio-economic modelling approach. *Agricultural Systems*, 94(2):357–367.

Shortle, J. S. and Dunn, J. W. (1986). The relative efficiency of agricultural source water pollution control policies. *American Journal of Agricultural Economics*, 68(3):668–677.

Shortle, J. S., Horan, R. D., and Abler, D. G. (1998). Research issues in nonpoint pollution control. *Environmental and Resource Economics*, 11(3-4):571–585.

Spraggon, J. (2002). Exogeneous targeting instruments as a solution to group moral hazards. *Journal of Public Economics 84*, 84:427–456.

Taylor, M. L., Adams, R. M., and Miller, S. (1992). Farm-level response to agricultural effluent control strategies: The case of the Willamette valley. *Journal of Agricultural and Resource Economics*, 17(1):173–185.

Tietenberg, T. H. (1973). Controlling pollution by Price and Standard systems: A general equilibrium analysis. *The Swedish Journal of Economics*, 75(2):193–203.

Viennot, P., Ledoux, E., Monget, J. M., Schott, C., Gernier, C., and Beaudoin, N. (2009). *La pollution du bassin de la Seine par les nitrates*, volume 3 of *La Collection du Programme PIREN-SEINE*. Agence de l' eau Seine-Normandie, Nanterre, France. Available from http://www.sisyphe.upmc.fr/piren/fascicules.

Ward, M. H., DeKok, T. M., Levallois, P., Brender, J., Gulis, G., Nolan, B. T., and Vanderslice, J. (2005). Workgroup report: Drinking-water nitrate and health–recent findings and research needs. *Environmental Health Perspectives*, 113(11):1607–1614.

Wier, M., Andersen, J. M., Jensen, J. D., and Jensen, T. C. (2002). The EU's Agenda 2000 reform for the agricultural sector: environmental and economic effects in Denmark. *Ecological Economics*, 41(2):345–359.

Wu, J. and Babcock, B. (2001). Spatial heterogeneity and the choice of instruments to control nonpoint pollution. *Environmental and Resource Economics*, 18:173–192.

Xepapadeas, A. (1991). Environmental policy under imperfect information: Incentives and moral hazard. *Journal of Environmental Economics and Management*, 20(2):113–126.

Appendix

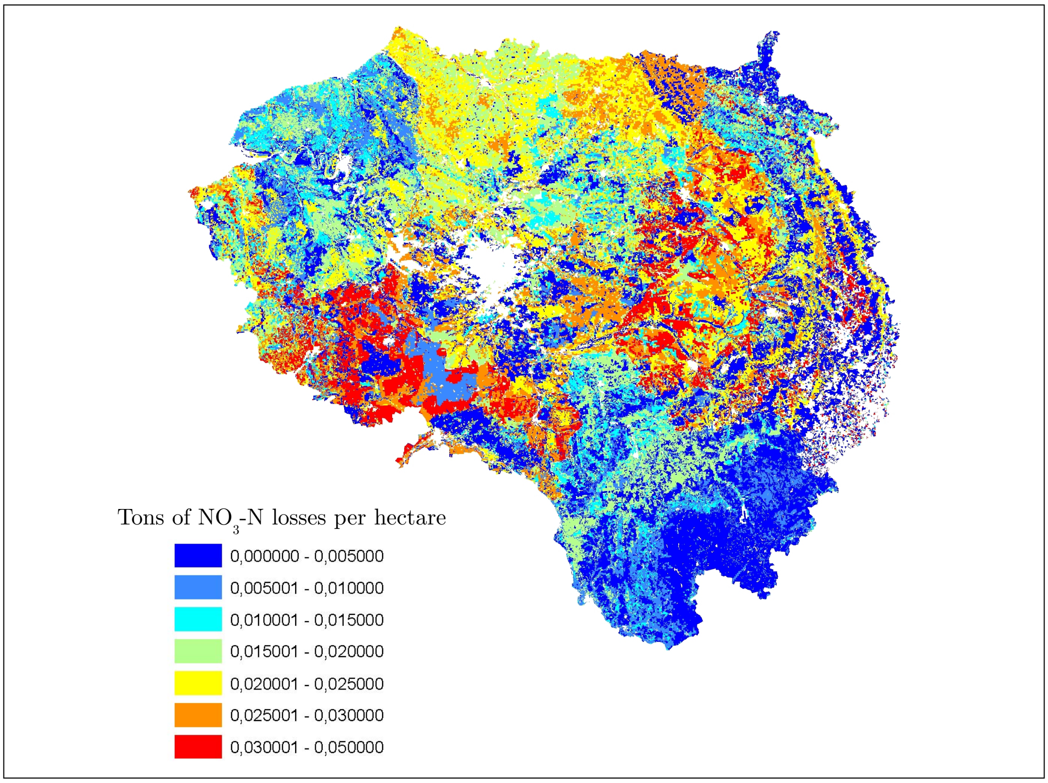


Figure I: NO_3-N losses per hectare under the Agenda 2000 scenario and no N-tax

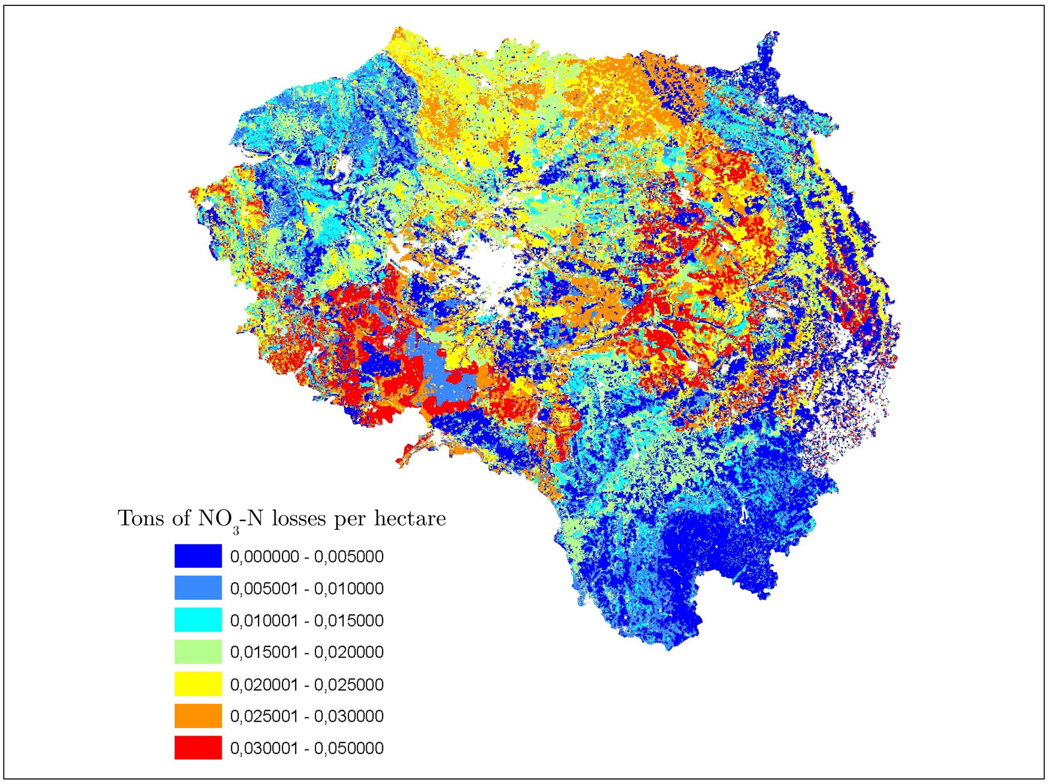


Figure II: NO_3-N losses per hectare under the Luxembourg scenario and no N-tax

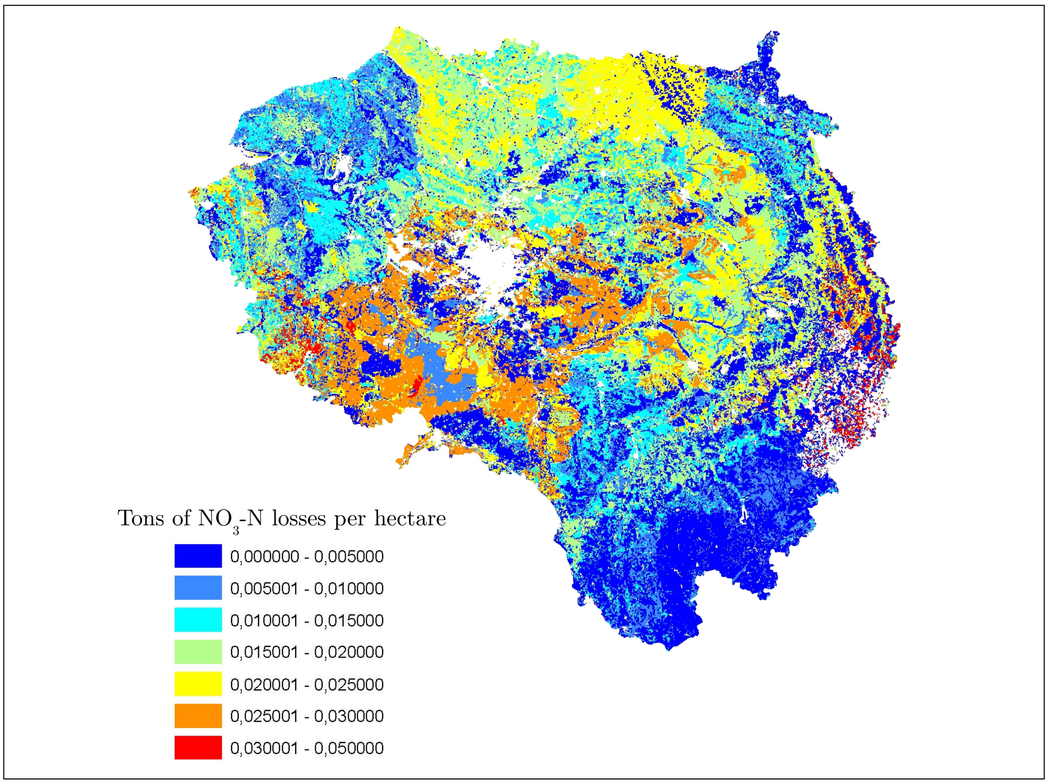


Figure III: NO_3-N losses per hectare under the Agenda 2000 scenario and a 100% N-tax

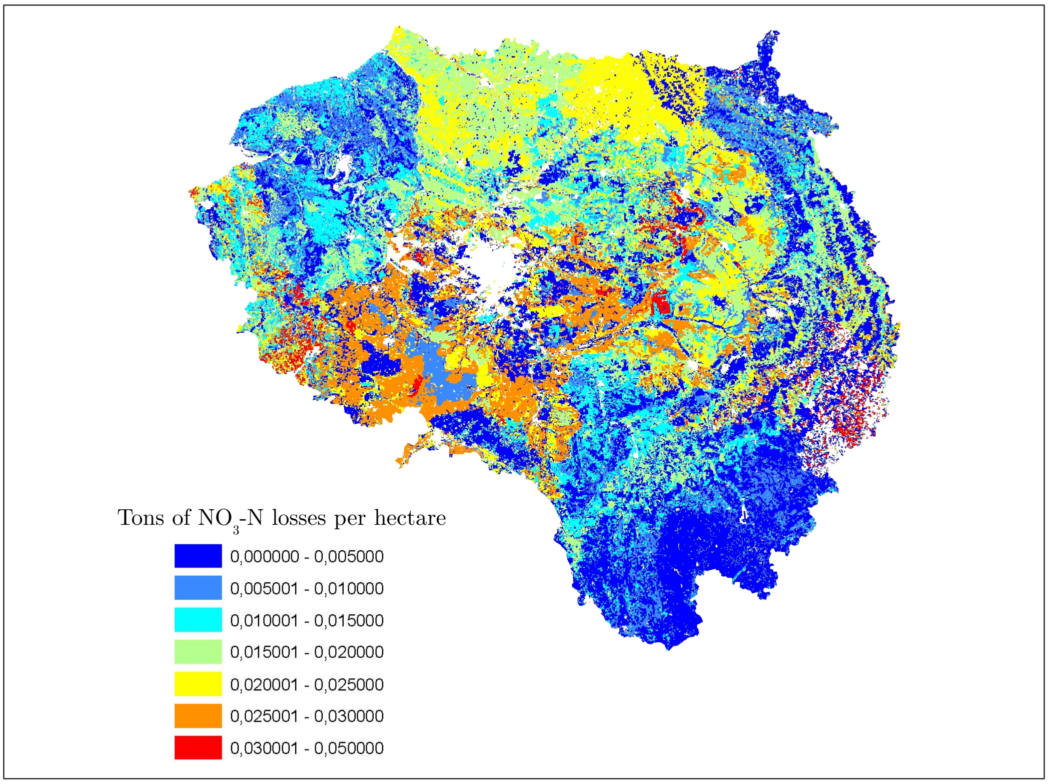


Figure IV: NO_3-N losses per hectare under the Luxembourg scenario and a 100% N-tax