Estimating the Marginal Social Value of Agriculturally Driven Nitrate Concentrations in an Aquifer: A Combined Theoretical-Applied Approach

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Abstract

We combine a theoretical model and a quantitative modeling chain based on a bio-economic model and a hydrological model in order to assess the marginal damage related to the nitrate concentration in an aquifer. The fundamental concept is to take the steady state level resulting from a social planner's optimization program as the target level of nitrate concentration. The interest of doing this is threefold: (i) we characterize the social value of damage related to the targeted nitrate concentration; which (ii) 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.

1 Introduction

In recent years, scientists and environmental agencies have been reporting increasing nitrate concentrations in water supplies, notably in the United States and Europe.

This pollution is mostly due to agricultural activities (European Commission, 2010; Parris, 2011)

To move beyond the handling of such water management problems according to administrative borders, the European Water Framework Directive (WFD; 2000/60/EC) has been designed to tackle them according to 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, interested parties, and researchers, increasing the demand for new tools including models to analyze, interpret, and display spatial information for river basin planning.

If we are to assess management strategies according to the WFD, modeling tools can help to test several policies and/or derive a theoretical best policy. Such tools are based on a hydro-economic model applied to given basins. From a hydrological point of view, these models must 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 both space and time.

Hydro-economic modeling 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 cases of non-point source pollution. However, most of these studies focus on salinity and river pollution rather than on aquifer pollution (Connor, 2008; Aftab et al., 2010, 2007). In order to better understand the impact of policies dealing with water pollution, hydro-economic modeling integrates three components: hydrological, economic and agronomic. 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 that 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) with input tax and set-aside management (Aftab et al., 2010), and (iii) broader 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 related to the environmental damage function which remain difficult to bring to light.

To overcome the limits of the applied and theoretical approaches, we set out to combine a theoretical model and a quantitative modeling chain based on a bio-economic model and a hydrological model. Such models are able to estimate the impacts of a farming system down to the aquifer water quality. Our study focused on the largest of the three aquifers of the Seine basin in the Northern France. The Seine basin intersects more than 6 millions agricultural hectares and provides the most part of drinking water for 16 million people..

The fundamental concept is to take the targeted nitrate concentration as the steady state level resulting from a social planner's optimization program. By doing this,the social planner is assumed to have enough insight into the environmental and human health damage to be able to estimate the sustainable concentration threshold. In doing so, he/she can access the monetary damage related to the pollution by using inverse modeling such as in this study. In that way, the social planner is in possession of information on monetary damage related to the pollution when the policy is chosen. To sum up, we aim at assessing the marginal value parameter as one that would be consistent with a social welfare maximizing planner and the targets set for the aquifer. Then, instead of computing an annual maximum nitrate stock as a function of damage assuming the monetary cost to be known, as is usually done in spite of the risks of not achieving the target concentration on time because of the time lag between the source (the nitrogen leaching flow) and its consequences (the aquifer contamination) associated with the hydrological transfer, the method we propose theoretically optimizes the policy for a given long-term goal, with year-on-year reduced cost of damage and contamination patterns that achieve the targeted time limit. In the aim of inferring the monetary damage parameter, we base an NO_3-loss tax on the theoretical implicit price of the damage in the long run. when a set of constant-rate taxes is implemented over time in appropriate agro-economic and hydrological models, and this set is taken in a sufficiently wide range, we can asses the tax level that will allow the target to be reached. then this tax provides the values of the damage parameter. Once the damage parameter has been estimated, we can design the benchmark tax path over time related to the solution of the optimal control problem including the inferred damage function. Although the present study includes more physics, the numerous hypotheses mean that it is still a theoretical approach, and the resulting tax path should be seen as a benchmark policy. A part of this idea was originally put forward in the work of Lee and Kim (2002). By minimizing the intertemporal cost to private farmers achieving a given target at a given date, they deduced an 'optimal tax'. However, as they did not use a damage function, they could neither assess the social value of the damage related to the nitrate concentration nor provide a welfare analysis in order to compare different tax policies

The interest of our methodology is the threefold: (i) we characterize the social value of damage related to the targeted concentration level of nitrate, with an explicit and physical estimation of the time lag between the source of the problem and the damage; which (ii) 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. Estimating the marginal damage of nitrate concentration, in monetary terms, and comparing various scenarios in terms of welfare, including the optimal tax path resulting from an optimal control program combined with quantitative simulation models dedicated to agricultural supply and hydrological transfers are two key points of our work. Although the question of the cost of reaching the WFD's or EPA's objectives has been extensively covered in the literature, it is still of interest and indeed necessary to assess the impacts of more and more stringent environmental policies, which are regularly announced and still not implemented. 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 the inherent variability of physical parameters across the aquifer which leads to a level of uncertainty that increases 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 to used to meet 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 a given nitrate concentration.

2 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.

2.1 Theoretical framework

The theoretical framework is based on a dynamic control model proposed by Bourgeois and Jayet (2015). 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 targeted in our analysis.

Let us 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 the soil characteristics summarized by a one-dimensional θ parameter. The individual farm profit function $\pi(x, \theta)$ is defined for any positive x and for any θ over the interval $\Theta = [\underline{\theta}, \bar{\theta}]$. 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 global profit per unit time 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 for pollutant accumulation problems applies. The state of the aquifer is characterized by the nitrate concentration assumed to be uniform in this theoretical part of the study, and 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 exponential characterized by the decline rate τ. On the other hand, nitrates accumulate in the aquifer due to N-fertilizer leached 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, β, characterizing the nitrate transfer between the top soil and the groundwater. At the root zone level, nitrate losses depend on the fertilizers consumed by any θ farm at ant time t, and the related emission function is denoted by $e(x, \theta)$ (let us recall that x depends on t and θ). All root zone nitrates reach the aquifer, and the related nitrate accumulation depends on the aquifer thickness. For the sake of simplicity, the time lag is assumed not to 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 (1).

$$\dot{z}(t) = -\tau z(t) + a \int_{\Theta} e(x(\theta, t-\beta), \theta)\gamma(\theta)d\theta \tag{1}$$

In the social planner's objective, the aquifer nitrate concentration is expressed 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 (2).

$$W = \int_0^{\infty} [\int_{\Theta} \pi(x(\theta, t), \theta)\gamma(\theta)d\theta - D(z(t))]e^{-\delta t}dt \tag{2}$$

Accordingly, his/her program refers to (3).

$$\max_{x(\theta,t)} W \quad \text{subject to (1) and to boundary conditions} \tag{3}$$

The problem is solved following the Hamiltonian approach (fully detailed in Bourgeois and Jayet (2015)), where λ denotes the current shadow price of the environmental variable z thanks to the boundary conditions (i.e., known values of x when $t \in [-\beta, 0]$ and the transversality condition satisfied). Let us summarize the set of equations useful for the present analysis by (R1).

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

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 $\dot{\lambda} = 0$. For interpretations, we note that when the delay is neglected, the optimal consumption of fertilizers is overestimated (Kim et al., 1993). Moreover, the shadow price and the corresponding optimal tax depend on the delay parameter, which implies that a decreasing tax path can be required even if the initial stock of pollutant is too low (Bourgeois and Jayet, 2015).

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 (MILP) 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 program 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)

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

restrictions concerning quasi-fixed production factors (land and livestock).[2] Farm groups are assumed to represent the agricultural sector at the regional level given and the Farm Accountancy Data Network (FADN) is the basis 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 avoid problems of data confidentiality and account for farm heterogeneity (e.g., Aftab et al. (2010, 2007)). The AROPAj calibration process is addressed by the re-estimation of the model parameter subset through a combination of Monte Carlo and gradient methods, in order to minimize the deviation between FADN estimated values and model results at farm group level (De Cara and Jayet, 2000). The AROPAj calibration process keeps the production set large, so AROPAj findings allow land allocation among crops, as is seen, for instance, in cases of change in CAP and in price or tax (Jayet and Petsakos, 2013).

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 throughout 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 make use 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 changes in economic policies to be taken into account, such as a tax on agricultural pollutants

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

A combination of the economic model AROPAj and the crop model STICS (Brisson et al., 1998, 2003) increases the model capacity to seize the adaptation of the farming system to price and to capture the farming system heterogeneity because soil characteristics and, more generally, pedo-climatic conditions are taken into consideration. Following the work of Godard et al. (2008), nitrogen input is calibrated to yield functions for the different farm groups and for most of the crops significant in terms of area and production (It accounts for soft wheat, durum wheat, barley, maize, sunflower, rapeseed, sugar beet, and potatoes.)

The crop model enables us to calibrate Mitscherlich-like exponential functions, through an accurate process of inferred physical parameter and curve selection, which requires a huge number of crop model runs (Godard et al., 2008). The N-yield functions replace fixed N-yield points and, consequently, changes in price or tax lead to changes in N- intakes and yields as optimal responses at the plot level.[3] The model becomes non linear, but this potential difficulty is easily overcome by a two-step optimization procedure. The first-step makes the per hectare gross margins optimal by computing N-input and yield for any crop and any farm-group. The second step uses these yields and N-inputs to start linear programming models. It should 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 occur at the root zone. Similarly to Aftab et al. (2010), we derive the leaching functions from STICS outputs when nitrogen input varies within a reasonable range. Linear regression applied to STICS outputs allows us to relate N-losses and fertilizer amounts (through affine functions). Consequently,

[3]This two-step process leads to the optimal solution when at least a part of each crop production is marketed. This is the case for almost all farming systems involved on the Seine river basin. When all production is on-farm used (for feed), we need shadow prices, instead of market prices, possibly computed by an iterative process. Without use of this iterative process, we obtain a slightly sub-optimal solution.

N-loss in the NO_3, related to crop j and farm group n depends on the N-input through the relation (4), A and B are estimated for a given plant on a given soil in the form

$$[NO_3]_{j,n}(N) = A_{j,n}N + B_{j,n} \tag{4}$$

More details on the N-loss estimation procedure in accordance with the selection and calibration of N-yield functions are given in Jayet and Petsakos (2013). To sum up, in terms of outputs, for any given tax scenario, AROPAj provides, among other things, the optimal amount of fertilizers for each crop, the optimal surface area to be devoted to each crop and the NO_3 losses at soil-root level.

2.3 Coupling with the hydrological model

The Seine river basin is characterized by the presence of several overlayed aquifer layers. The MODCOU application represents the whole Seine basin (95,560 km^2) and the three major aquifers designated accordingly to their geology (figure 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).

The use of the hydrological model MODCOU allows the nitrate transfer from the root zone toward the groundwaters of the Seine river basin. The hydrological model MOD COU (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, and also the flow to, in, and between aquifer layers and the interactions between rivers and each layer of the free part of the aquifer. It is also able to compute the convective transfer of solute to, in, and between those layers.

Regarding the MODCOU model, the initial condition (in 2002) can affect the simulation over several decades due to transfer time through the unsaturated zone (UZ) and aquifer nitrogen accumulation. Thus, special attention was paid to obtaining realistic initial conditions for the AROPAj-MODCOU simulation. Initialization is based on the MODCOU-STICS simulations (as described in Ledoux et al. (2007) and in Viennot P. (2007)), and is aimed at correctly reproducing the nitrate stock in the UZ, among other things, at the starting date of simulation (2002). In such simulations, STICS provides estimates of the nitrate flux entering in the UZ over the period 1971-2002 and reaching the aquifer from 2003. Moreover, MODCOU-STICS provides the NO3 reference flux as the mean value of 2002-2004 fluxes (fig 2).

Coupling of AROPAj and MODCOU comes into play from the simulation start date (2002). In this respect, we use the AROPAj spatialization process which geographically distributes AROPAj output at a fine level of resolution The three-step spatialization process starts with the spatial econometrics model developed by Chakir (2009). Firstly, crop location is estimated in relation to physical data at a very fine resolution. 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 spatialization method provides the contribution of each farm group on each cell of the grid to the regional (FADN) agricultural activity. This enables any AROPAj output to be distributed 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. Following the temporal tax path, spatialized simulated NO3 fluxes provided by the AROPAj model feed MODCOU by topsoil nitrate changes according to the formula:

$$\begin{aligned} NO_{3\ \text{input}} &= Mean[NO_{3\ \text{MODCOU-STICS}}(2001-2003)] \\ &\quad *[1+\tfrac{NO_{3\ \text{AROPAj}}(year)-NO_{3\ \text{AROPAj}}(2002)}{NO_{3\ \text{AROPAj}}(2002)}] \end{aligned} \tag{5}$$

For each year, the signal transmitted to MODCOU is based on the average emission provided by the coupled physical models (MODCOU-STICS) on the period [2001,2003], augmented with the annual variation provided by AROPAj and related to temporal tax path.

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 an NO_3 lixiviated flux which is transfered to the groundwater by the dynamic hydrological model MODCOU. More accurately, a scenario consists of a fertilizer tax path combined with one level of exogenous livestock adjustment. Scenarios may differ according to the tax path and to the level of livestock adjustment. In terms of CAP context, we assume that the Agenda 2000 scheme holds up to and including 2006 and that the "decoupled Luxembourg scheme" applies as of 2007. Figure (fig2) summarizes the integration process. In this integrated modeling application, we focus on the largest of the three aquifers within the Seine river basin, the chalk aquifer, and, more precisely, the free part of the aquifer, which is both the most affected by nitrate contamination and the most subject to water withdrawal. For the economic analysis, the NO_3 concentration in the aquifer is calculated as the annual median value for the entire chalk aquifer.

3 The integrated approach

The theoretical model and the quantitative integrated model match when the damage function is set explicitly. To do this, the linear derivative of the function addresses the need for parsimony and for an easily tractable approximation when damage varies within a relatively narrow range of values. Consequently we adopt the quadratic form (6).

$$D(z) = \frac{k}{2} z^2, \ k > 0 \tag{6}$$

The advantage here is that the social marginal damage is represented through the use of a single parameter. At this step, this parameter is still unknown.

By 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 system (R2) leads to relation (7).

$$\bar{\lambda} = \frac{k \; \bar{z}}{(\tau+\delta)} \tag{7}$$

Let us reverse the roles of z and k and consider the concentration level $\bar{z}$ to be the target set by the social planner. Making this assumption that the social planner maximizes the social welfare reveals the marginal social value of the damage $k\bar{z}$ through relation (7). Obviously this value depends implicitly on the structure of both the agricultural production, $\pi(x,\theta)$, and the pollutant emission, $e(x,\theta)$). It also depends on the discount rate (δ) and the physical parameters of the hydrological system (τ, β, a).

Finally, we implement the tax, μ, on the pollutant emissions e. For any θ-farm at any time t, the private optimal choice is obtained by solving the equation $\pi_x(x,\theta) = \mu \; e_x(x,\theta)$. Through the first equation of system (R1), this leads to the important but simple equation (8) linking this tax to the implicit price λ associated to with the state variable z.

$$\mu(t) = ae^{-\delta\beta}\lambda(t+\beta) \tag{8}$$

In the long term, this equation becomes $\bar{\mu} = ae^{-\delta\beta}\bar{\lambda}$. This long-term nitrate emission tax equation can be transformed to highlight the link between the tax and the social marginal damage $k\bar{z}$, as in relation (9).

$$\bar{\mu} = \frac{ae^{-\delta\beta}}{\tau + \delta} k\bar{z} \tag{9}$$

A tax set on soil-root nitrate losses can be implemented easily 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 root zone nitrate losses provided by AROPAj are applied on an annual basis and the simulation is run over a 100-year period. If we 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 that is the social planner's target. Finally, let us consider that economic and physical parameters δ, τ, β, and a are given or estimated. Equation (9) provides estimates of the damage parameter,k, and the social marginal value, $k\bar{z}$, of the damage associated with the nitrate concentration in the aquifer.

We choose to set the discount rate δ to 0.04.[4]

3.1 Physical parameters on average and variance

The natural decline rate, τ, and the mean time lag, β, of nitrate transfer between the root zone and the groundwater are approximated by the MODCOU model. Estimates are $\bar{\tau} = 0.02\ y^{-1}$ 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.

[4] Most economists use and recommend discount rates ranging from 3 to 5% when trying to put a price tag on future damage (Evans and Sezer, 2004). The E.P.A uses a rate of 3%, and the European commission suggests a rate of 5% since 2008 (European Commission, 2008). Since 2004, France, where our case study is located, has been following Lebègue et al. (2005) who recommend a discount rate of 4%

The Chalk aquifer is characterized by strong heterogeneity. Moreover, errors in the measurement of phenomena, which are inherent in quantitative models, lead us to randomize these two physical parameters. We distribute the 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, thus ensuring that the parameters values are always positive. 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. Unlike these two parameters, the clearing rate, τ, is assumed to remain constant and homogeneous in the aquifer. These elements are summed up in Table (1).

Let us consider given values of the steady state $\hat{z}$ and the tax level $\hat{\mu}$ which allows the aquifer to reach the state $\hat{z}$ when the tax $\hat{\mu}$ is constantly implemented over time. We take this tax to be the first-best long-term tax considering 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 (10).

$$\hat{k}\hat{z} = \underset{a,\beta}{E}[k]\hat{z} = \hat{\mu}(\tau + \delta)\underset{a,\beta}{E}[\frac{e^{\delta\beta}}{a}] \tag{10}$$

3.2 From nitrate target to social marginal damage value

Let us consider a set of I constant tax scenarios over 100 years together with runs of the AROPAj and MODCOU models. 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 made 10,000 calculations of k, reflecting the heterogeneity of the physical parameters. Figure (3) displays the monetary marginal damage ($k{*}z$) related to the concentration taken to be the optimal level. A 1 mgNO_3/l decrease in a target means that the marginal damage increases by 1.7 € $(mgNO_3/l\ ha\ year)^{-1}$.

Focusing on the two targets given, 50 and 38 mg/l[5], we estimate the marginal damage by 1.1 and 21.9 € $(mgNO_3/l\ ha\ year)^{-1}$respectively; in other words, 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 at 9.1 and 6.5 € $(mgNO_3/l\ ha\ year)^{-1}$ when the parameter distributions follow uniform densities and lognormal densities respectively.

4 Welfare Impact

4.1 Welfare loss induced by non-optimal regulation

In this investigation, welfare can be understood by including the net farmers' profit and the environmental damage caused by nitrate in the aquifer (by assuming that tax is lump-sum monetary transfers between the farmers and the rest of the society, we can suppress it from the welfare computation). Once the target, $\bar{z}$, has been set and the marginal damage, $k * \bar{z}$, is estimated, the k estimate allows the welfare to be calculated explicitly. We compare welfare according to scenarios differing in terms of tax path and AROPAj livestock adjustment.

As a benchmark, the theoretical model leads to an assessment of the optimal time path of the tax when the targeted NO_3 concentration is close to the steady state after 100 years and when the parameters are set at their average values as mentioned

[5]According to Directive 2006/118/EC of the European Parliament and the Council of 12 December 2006, 50 mg/l is the standard imposed and 38 mg/l the standard recommended to reverse significant and sustained upward trends

above. Other scenarios are designed to be more acceptable to farmers. In other words, the time path of the tax is different. In addition, we consider that livestock numbers may be adjusted because of the tax on all root zone nitrate lixiviation (mineral and organic sources). In 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 is reached 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 related to the aquifer NO_3 concentration of 50 and 38 mg/l respectively. 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 toward the same nitrate concentration in the aquifer in the long term 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 numbers leads to significant changes in tax levels when the target is set at 50 $mgNO_3/l$. If the adjustment increases livestock numbers, it is necessary to decrease the tax levels to reach the target. Different tax paths are represented in figure (4).

Figure (5), on the left, displays the tax refunded profit over time for each scenario that differs in terms of tax path and livestock adjustment when the targeted nitrate concentration is 50 $mgNO_3/l$. As the livestock adjustment leads to a dramatic decrease in the long-term tax required to reach the target, not surprisingly, the sum of farmers' profit and tax refunds is higher when the livestock adjustment increases. Nevertheless, focusing on economic effort, there is only a very slight difference between the scenarios and the path does not impact farmers' profit to any great extent. This situation occurs thanks to the low level of taxes. Whatever the scenario chosen, the evolutions of the aquifer nitrate concentration over time are very close. (see Figure (6)).

To sum up, when the target is not difficult to reach, an adjustment at farm level such as a livestock adjustment offers the possibility of strongly attenuating the harshness of the policy tool, i.e., the tax on pollutant emission. Table (2) displays estimates of discounted welfare according to scenarios and targets.

Let us now consider the more stringent target, 38 $mgNO_3/l$. Figure (5), 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. Compared to the $50 mgNO_3/l$ target, time-curve shapes and scenarios now differ significantly.

Figure (6) displays the nitrate concentration paths converging on either the 50 or 38 $mgNO_3/l$ target. 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 result in a 2060 nitrate concentration 4 $mgNO_3/l$ higher than the optimal one. The time lag strengthens the impact of a non-optimal scenario. 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). If the social self-imposes intermediate targets for himself, for example 42 mg/l, we can see that the optimal scenario reaches this intermediate target ten years before the constant scenario and twenty years before the scenario that is the most accommodating for farmers. Contrary to the situation for instantaneous pollution, when lagged pollution is considered, the social planner cannot rely on observation to manage developments. The longer the time delay on the pollution, the more important it is to act early and get as close as possible to the optimal policy. We note that the business as usual scenario, reproducing the actual agricultural practices with no livestock adjustment, leads to an NO_3 concentration stabilization at a slightly higher level of 50 mg/l. Figures (7) and (8) show the welfare induced over time by each tax scenario and livestock adjustment. For a given target of 50 mg/l, the welfare loss occurring in the case of sub-optimal scenarios is close to zero and mainly due to livestock adjustment. In the case of 38 mg/l target, welfare induced by the optimal taxation scenario is lower than for other scenarios during the first twenty years. The optimal regulation impacts the gross margin more strongly but, because of the lag, does not impact $N0_3$ concentration. Later, the opposite occurs.In addition, livestock adjustment supplies flexibility. Consequently, scenarios including high levels of adjustment may be more efficient than the optimal one when adjustment is excluded. This highlights the importance of not managing the mineral fertilizers alone.

Table (2) shows the discounted welfare in 2003 with a discount rate of 4%. 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 consideration in publications (e.g., Lacroix et al. (2005); Kosenius (2010)), leads to a welfare loss of 0.5 %.

It should be noted that the business as usual (BAU) estimate of discounted welfare leads to 0.02% (respectively 8.72%) loss in the 50mg/l (respectively 38mg/l) case compared to welfare induced by the optimal tax path in the 50 mg/l (respectively 38 mg/l) case (with 0% livestock adjustment).

4.2 Impact of discount rate

The discount rate impacts the calculations of the damage parameter (see Equation (10)) and accordingly the marginal damage. We illustrate this in table 3, which provides the marginal damage values related to two target concentrations of 38 and 50 mg/l and to values of discount rate of 3, 4 and 5%. To consider an increase of discount rate from 3 to 4% (resp. from 4 to 5) implies an increase of marginal damage of 50% (resp. 45%), for any target. The impact is clearly strong and it is strengthened by the lag. It is the term $\exp^{-\delta*\beta}$ in equation (10) that mainly determines the impact of discount rate on marginal damage. When there is no time lag, the impact of increasing discount rate from 3 to 4% (respectively 4 to 5%) on the marginal damage is only 20% (respectively 15%) (see Table 3). Commonly, increasing discount rate leads to a decrease in the value of environmental damage arising in the long term. When we focus on a given target (long term nitrate concentration),t he same effect of discount rate means that increasing the discount rate is related to an increasing value of the damage estimated by the social planner. Regarding the welfare evaluation, the discount rate has a double-sided effect. In addition to the effect of the discount rate on the marginal damage explained above, there is the standard effect on the present versus future costs and benefits. Consequently, change in cumulated farm profit over time is amplified when policies differ (in terms of tax path over time), and, due to converging tax paths, differences are visible in the early period. On the damage side, the differences are attenuated (the effect of policies on nitrate concentration is visible only after a period equivalent to the time lag). This complex double-sided effect is itself impacted by the level of the target. In the 38 mg/l case, the difference in discounted welfare cumulated over time between the benchmark policy (i.e. the optimal tax path) and the policy reaching 90% of the long-term tax after 10 years is respectively 0.5, 1.4, or 4.1% when the discount rate is 3, 4, or 5%. The difference between the benchmark and the business as usual (BAU) case is respectively 4.8, 8.7,and 20% when the discount is of 3, 4, and 5%. In other words, higher the discount rate, higher the difference between the optimal case and the alternative case. The discount rate effect has no significant impact in the 50 mg/l case. Regarding the comparison between the optimal case and the BAU, difference in cumulated discounted welfare decreases when discount rate increases. This difference is respectively 0, 0.02 or 0.01% when the discount rate is 3, 4 or 5% (see Table 4).

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 the nitrate concentration in the aquifer, and the damage parameter. We consider the nitrate concentration target as the steady-state level resulting from a social planner's program in which the lag is taken into account. To this end, we combine a theoretical model and a quantitative modeling chain based on bio-economic and hydrological models.

We show that decreasing the target by 1 $mgNO_3/l$ is equivalent to assessing an increase in marginal damage of 1.7 €.$(mg/l.ha.year)^{-1}$. When targets are 50 and 38 mg/l, the related marginal damages are 1.1 and 21.9 €.$(mg/l.ha.year)^{-1}$, i.e., differs by a factor of 20.6. Moreover, the estimated damage parameter allows us to design the optimal path consistent with the target and to 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 significant discounted welfare loss when the aquifer target NO_3 concentration increases in stringency, up to 0.5% in the case of 38 $mgNO_3/l$.In this case, the dis- counted welfare loss can reach 2.3% when a tax path that is more favorable to farmers during early years meets this target Although the present study draws benefit from hydrological modeling of the aquifer nitrogen concentration and from a well calibrated agro-economic model, it should be seen partly as a theoretical approach as well as being considered an applied study because of the assumptions made. Strong assumptions are introduced via the questionable value of the discounted rate and mainly via the long-run concentration target resulting from an optimal choice by the policy maker. It should be noted that the selected target, i.e., the mean aquifer concentration, might not be the strongest constraint, as part of the aquifer will, of course, still have a concentration above the threshold even when the target is reached. Moreover, the mean aquifer concentration cannot be observed. However, another type of target, better suited to the management issue, can easily be fixed, e.g. the average concentration in several wells, or its maximal value. In the same line of thought, the time when the target should be reached could also be an adjusted variable, although the hydro-geological condition will not make it possible to reach all the objectives at all times. To return to models, it is supposed that both AROPAj-STICS and MODCOU perform well enough for us to have confidence in their simulation. As stated before, these models were assessed in several contexts and, although they are not perfect, they were found suitable for such a theoretical study. Other important assumptions are the stability of the climatic and economic conditions. It is quite probable that economic conditions will change according to the outcome of agricultural policy, changes in world trade and climatic conditions. Such changes will be taken into account in a following study

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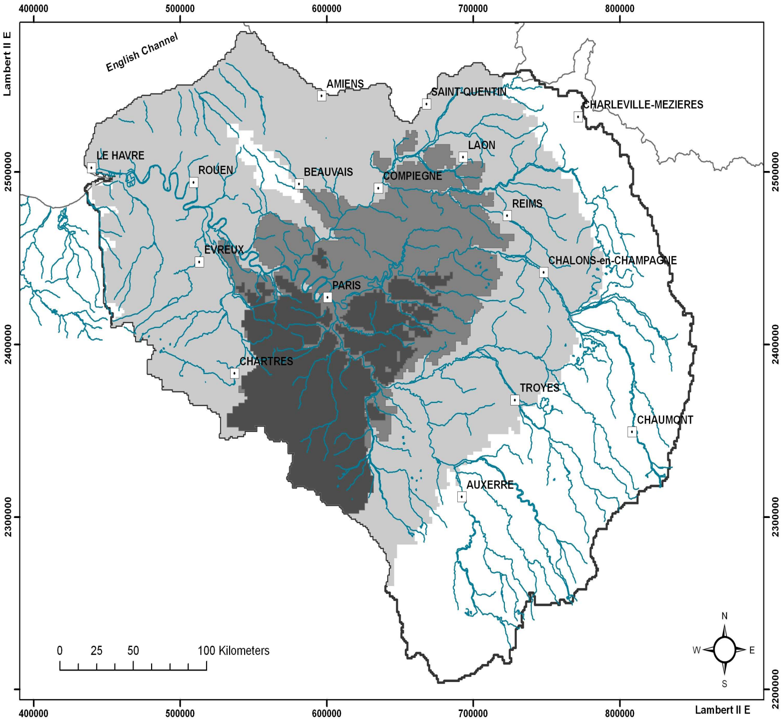

Figure 1: The Seine basin (shape) and the three main overlaid aquifer layers, from top to bottom: the Oligocene (dark gray), the Eocene (gray) and the Chalk (light gray).

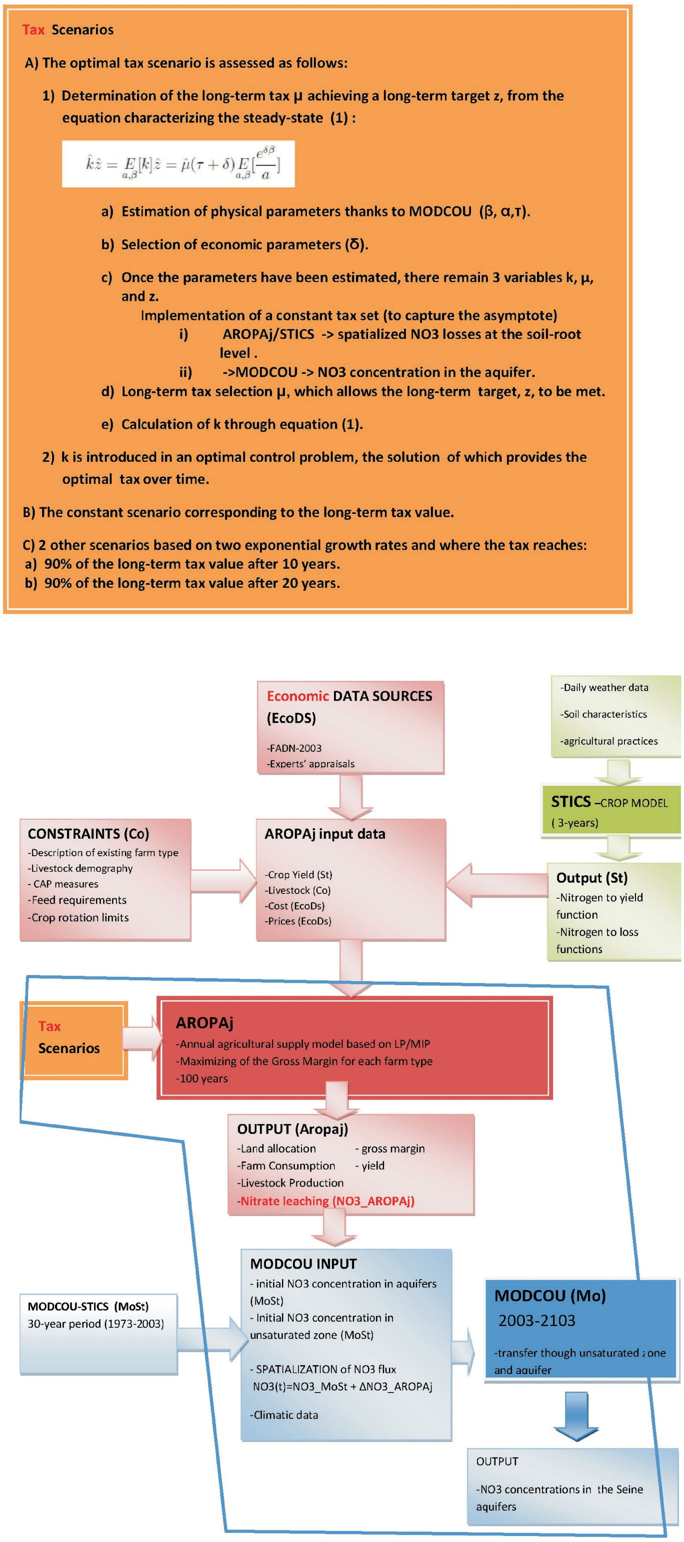

Figure 2: Schematic representation of the coupling methodology: 1st, comparison of the AROPAj STICS and STICS MODCOU annual root zone nitrogen fluxes (red and green boxes) for the reference year 2002 shows a good match 2nd, the tax scenario and associated annual root zone nitrogen flux (orange to purple boxes) simulated by AROPAj were then disaggregated on time according to the 2002 STICS daily evolution, and the daily fluxes transmitted to MODCOU the temporal evolution of the aquifer concentration analyzed on a annual basis (blue circles).

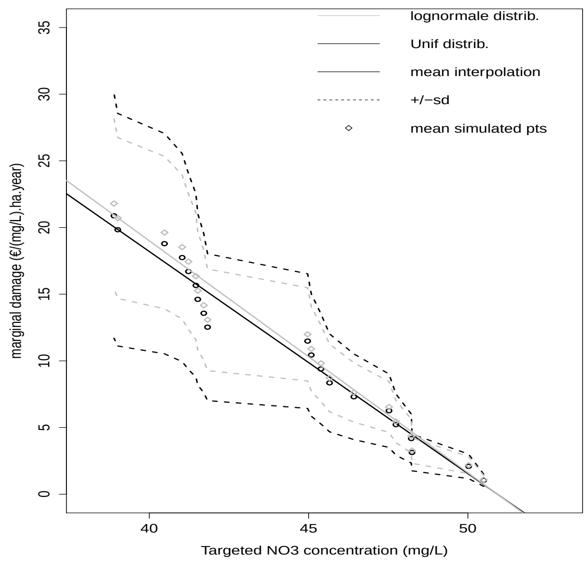

Figure 3: Monetary marginal damage as a function of targeted NO_3 concentration level (in bold: a and $beta$ following uniform distributions; in gray: a and β following lognormal distributions).

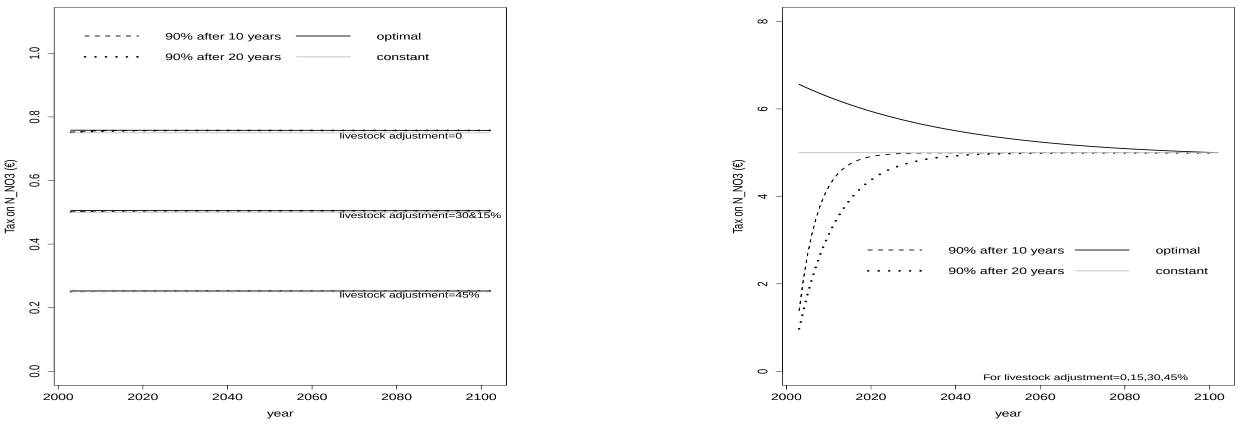

Figure 4: Tax paths over time, for 50 mg/l, on the left, and 38 mg/l, on the right, as steady state NO_3 concentration

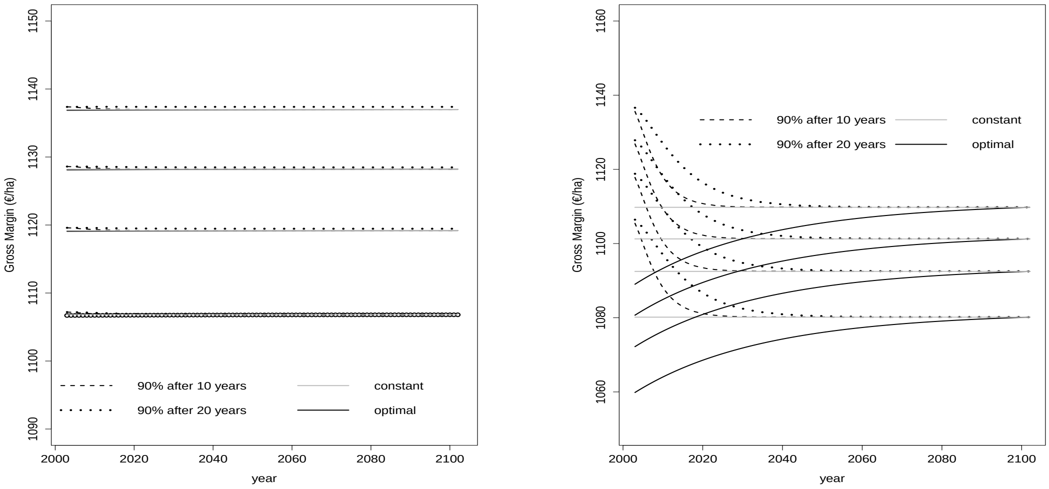

Figure 5: Profit plus tax receipt over time related to the 50 $mgNO_3/l$ (left) and the 38 $mgNO_3/l$ (right) target, when the tax path is optimal (solid black), constant (solid grey), reaches 90% of the steady state tax exponentially in 20 years (dashed) and in 10 years (dotted), and when the amplitude of livestock adjustment changes from 0 (down) to 45% (top) of initial livestock by increment of 15%.)

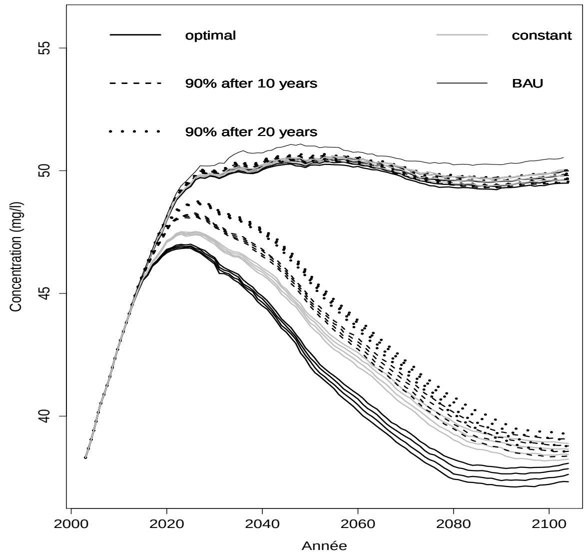

Figure 6: Nitrate concentration path related to scenarios differing in tax path and livestock adjustement, regarding the two targets (long-term value of 50 and 38 $mgNO_3/l$ in the aquifer), when the tax path is optimal (solid black), constant (solid gray), reaches 90% of the steady state tax exponentially in 20 years (dashed) and in 10 years (dotted), and when the amplitude of livestock adjustment changes from 0 (bottom) to 45% (top) of initial livestock by increments of 15%.

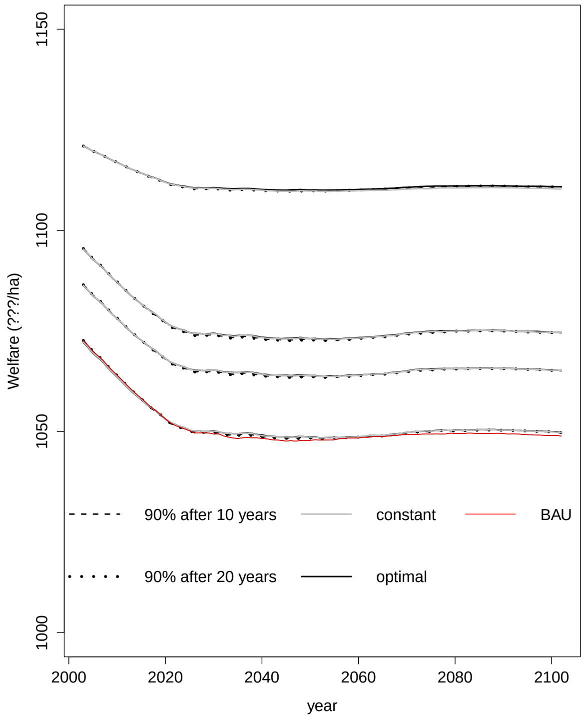

Figure 7: Welfare in the case of 50 mgNO_3 /l nitrate concentration target,when the tax path is optimal (solid black), constant (solid gray), reaches 90% of the steady state tax exponentially in 20 years (dashed) and in 10 years (dotted), and when the amplitude of livestock adjustment changes from 0 (bottom) to 45% (top) of initial livestock by increments of 15%.

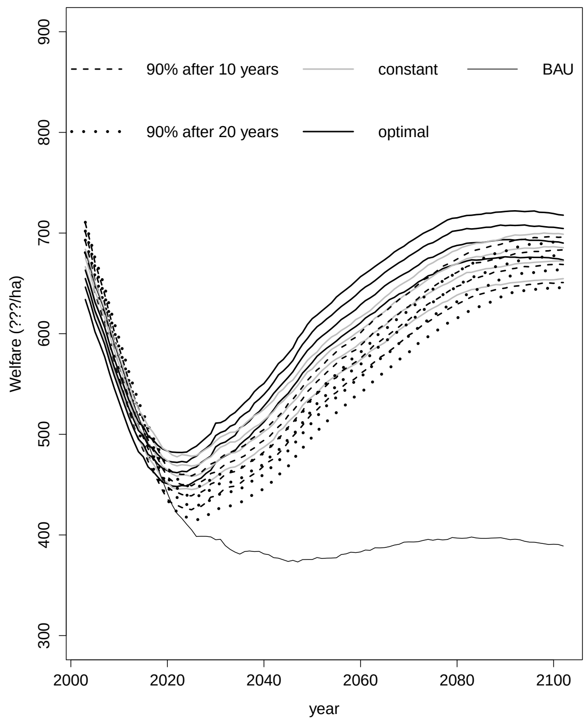

Figure 8: Welfare in the case of 38 mgNO_3 /l nitrate concentration target,when the tax path is optimal (solid black), constant (solid gray), reaches 90% of the steady state tax exponentially in 20 years (dashed) and in 10 years (dotted), and when the amplitude of livestock adjustment changes from 0 (bottom) to 45% (top) of initial livestock by increments of 15 %.

Table 1: Distributions of physical parameter values (aquifer thickness $1/a$ and time lag, β).

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 2: Variation and decomposition of discounted welfare in scenarios differing by livestock adjustment (α, amplitude in % of the initial value) compared to the optimal tax scenario without livestock adjustment, for two NO_3 concentration targets (mg/l).

Target(mg/l)	50				38					
alpha /scenario	BAU	optimal	constant	10 years	20 years	BAU	optimal	constant	10 years	20 years
0%										
Welfare	25906	25910	25911	25910	25910	11834	12958	12892	12775	12661
Variation	-0.02	0.00	0.00	0.00	-0.01	-8.72	0.00	-0.51	-1.41	-2.29
Net Gross Margin	27150	27141	27142	27144	27146	27154	26212	26489	26613	26717
Tax	0	602	571	506	447	0	2870	2684	2470	2252
Damage	1244	1230	1231	1234	1235	15331	13254	13596	13837	14506
15%										
Welfare		26223	26233	26221	26219		13300	13327	13105	12990
Variation		3.83	3.82	3.82	3.81		2.64	2.08	1.14	0.24
Net Gross Margin		27452	27452	27453	27454		26515	26791	26916	27020
Tax		404	383	338	298		2823	2644	2435	2222
Damage		1228	1229	1232	1234		13214	13564	13810	14030
30%										
Welfare		26446	26446	26444	26442		13551	13477	13355	13233
Variation		4.71	4.71	4.70	4.69		4.57	4.00	3.06	2.12
Net Gross Margin		27674	27674	27675	27675		26727	27007	27133	27238
Tax		402	381	336	296		2783	2613	2409	2199
Damage		1227	1228	1231	1233		13175	13529	13778	14005
45%										
Welfare		26667	26661	26663	26660		13797	13720	13594	13471
Variation		7.92	7.91	7.91	7.91		6.47	5.88	4.90	3.95
Net Gross Margin		27892	27892	27892	27890		26933	27215	27343	27449
Tax		202	191	168	148		2754	2591	2390	2183
Damage		1225	1231	1230	1232		13136	13495	13748	13978

Table 3: Sensitivity of long-term marginal damage to discount rate, for two NO_3 concentration targets (mg/l).

β /Discount rate	3 %	4%	5%
		Target 50 (mg/l)	
30%	1.47	2.19	3.15
0%	0.78	0.94	1.09
		Target 38 (mg/l)	
30%	14.68	21.5	31.80
0%	7.80	9.36	10.92

Table 4: Sensitivity of cumulated discounted welfare and its decomposition to discount rate for two NO_3 concentration targets (mg/l).

Target(mg/l)	50					38				
alpha /scenario	BAU	optimal	constant	10 years	20 years	BAU	optimal	constant	10 years	20 years
Discount rate		3%								
Welfare	33954	33955	33954	33954	33954	21398	22482	22445	22349	22249
Net Gross Margin	35017	35000	35001	35004	35005	35017	33839	34158	34288	34406
Tax	0	775	734	669	604	0	3680	3462	3241	3003
Damage	1061	1045	1047	1049	1051	13618	11358	11713	11939	12157
Discount rate		4%								
Welfare	25906	25910	25911	25910	25910	11834	12958	12892	12775	12661
Net Gross Margin	27150	27141	27142	27144	27146	27154	26212	26489	26613	26717
Tax	0	602	571	506	447	0	2870	2684	2470	2252
Damage	1244	1230	1231	1234	1235	15331	13254	13596	13837	14506
Discount rate		5%								
Welfare	20620	20622	20622	20622	20622	4384	5545	5545	5317	5190
Net Gross Margin	21989	21978	21979	21981	21989	21982	21206	21450	21569	21663
Tax	0	490	463	400	347	0	2336	2174	1967	1766
Damage	1369	1356	1357	1359	1361	17605	15661	15995	16252	16472