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ANALYSIS

# Agricultural water nonpoint pollution control under uncertainty and climate variability

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**Abstract**

The objective of this paper is to study the probabilistic cost-effectiveness of the farm management practices supported by the European Union for reducing nitrate pollution. Our method consists in using a bio-physical model to evaluate the environmental and economic impacts of various scenarios characterized by a set of farm practices. The cost-effectiveness of each scenario is calculated for a catchment area located in the northeast of France, for various climatic years and under different assumptions of crop prices. The results show that it is not realistic to obtain a rapid reduction of nitrate concentrations by implementing the scenarios tested. In the long run and irrespective of the economic context simulated, the optimum scenario in the case studied is one that combines integrated fertilization with the introduction of catch crops. Our findings thus highlight the effectiveness of catch crops that are able to reduce variability of nitrate concentration and thus significantly reduce the risk of exceeding environmental constraints. They therefore provide some recommendations for policy-makers.


*Keywords:* Stochastic nitrate pollution; Interdisciplinary approach; Cost-effectiveness analysis; Cropping systems; Environmental policies; Environmental constraints

## 1. Introduction

For several years, scientists and environmental agencies have reported an increase in the water nitrate concentration, especially in Europe and the USA. It has been established that this pollution is due, for the most part, to agricultural activities especially to intensive farm management practices (European Environment Agency, 2001; US Geological Survey, 1999). Although the effect of high nitrate concentration on human health is still a controversial issue (Addiscott et al., 1991; Apfelbaum, 1998), several environmental policies have been defined in order to control agricultural nitrate pollution. Controlling nitrate pollution is usually considered as the first step towards a wider control of agricultural water pollution

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in the sense that a better management of the nitrogen cycle will initiate an essential process of technical and organizational learning-by-doing to control phosphorus and pesticide pollutions as well. These policies rely mainly on legal instruments (command and control approach), such as limitation on the authorized level of pollutants or the designation of protected areas, and rarely use economic instruments (incentive-based approach). Some economic incentives have nevertheless been implemented in order to encourage farmers to improve their practices. These schemes are voluntary and involve farmers entering into management agreements in return for financial compensation. Contractual obligations regulate such production practices as the 'best management practices' (Clean Water Act Amendments, 1987) in the USA, the 'agri-environmental measures' (Council Regulation No. 2078/92/EEC) in Europe and the 'Nitrate Sensitive Areas scheme' in the United Kingdom (Szoege et al., 1996). Similar environmental policies are likely to be more important in the near future, especially in European Union countries. Economic and environmental assessment of these policies is therefore of prime importance.

Contractual obligations are based on agronomic recommendations but their efficiency conditions have not been precisely assessed. The uncertainty and climate variability of their economic and environmental impacts have not been significantly explored. Can these standard practices have a significant environmental impact every year, in any agronomic and hydrologic system? Can they minimize the economic burden of pollution control in any context? This paper considers these questions. The aim is to provide useful assessments for policy makers in order to determine efficient management practices for controlling agricultural pollution.

Due to the complexity of interactions between economic, agronomic and hydrologic systems, to the stochastic nature of some factors (e.g. climate, soil, topographic conditions), and to the lack of knowledge, the consequences of management practices recommended by state authorities cannot be accurately predicted. Uncertainty is particularly high in the case of nitrate pollution because many factors are involved. For instance, climate and pests affect crop growth and nitrogen use efficiency, temperature influences the nitrogen cycle in the soil, especially nitrogen mineralization, and the nitrate runoff depends on rainfall and soil characteristics. Mc Sweeny and Shortle (1990) have developed the cost-effectiveness approach in order to take into account uncertainties in both the efficiency and the cost of the proposed management practices. In their approach, known as the probabilistic cost-effectiveness approach, the uncertainty in the runoff rate value is described by means of a random variable. Bystrom et al. (2000) used a similar approach to study the interest of wetlands for controlling nitrate pollution. The main limitation of these studies is that uncertainty is described very roughly by means of very simple probability functions. In these functions, uncertainty due to climate variability and uncertainty due to lack of knowledge are not distinguished. Moreover, the level of uncertainty (i.e. the variances of the random variables) is fixed arbitrarily.

An alternative solution would be to use bio-physical models. Such models can serve to predict crop yield, crop quality, water and nitrogen flows in relation to field characteristics and management practices (Wagenet and Hutson, 1996). Agronomists and economists have already explored this way (Ribaudo et al., 2001; Vatn et al., 1999; Weaver et al., 1996). However, they used a deterministic function for analyzing water-related environmental impacts of agriculture and did not study the influence of climate variability on the results of a pollution control policy. Most of the bio-physical models include climatic variables. Since they are linked with economic models, implemented at the catchment scale, such models can be interesting for studying the between-year variability of the cost-effectiveness of various farm management practices. Another advantage of bio-physical models is that, in some cases, the errors of the model have been extensively studied (Gorres and Gold, 1996). It is then possible to define realistic probability functions for describing model errors on the basis of large data sets. Such probability functions give a realistic representation of uncertainty due to the lack of knowledge.

In this paper we use a bio-physical model to assess the probabilistic cost-effectiveness of the farm management practices supported by the European Union for reducing nitrate pollution. Six nutrient management scenarios are examined. Each scenario is characterized by a set of farm practices defined for controlling nitrate pollution. The cost-effectiveness of each scenario is calculated for various climatic years under different assumptions of crop prices and for a catchment area located in the northeast of France. The methodological framework is described in detail in Section 2. The case study based on empirical data is presented in Section 3. The cost-effectiveness of the scenarios is analysed in Section 4. In Section 5, the implications of our findings are discussed for agri-environmental policy design.

## 2. The model

A model is defined here to evaluate the cost-effectiveness of a series of scenarios for reducing the nitrate concentration in drained water. The scenarios are evaluated for a catchment area covering several fields and for several years. Our approach consists of four steps. First, the farm practices associated with the different scenarios are determined for each field of the catchment area and for each year. Second, the consequences of the different farm practices on yield, grain quality and water nitrate content are simulated for each field and each year by means of crop models. Third, costs induced by the different scenarios are calculated at the catchment level for each year on the basis of crop model outputs. Finally, scenarios inducing low monetary costs for farmers and satisfying constraints on water nitrate content are identified. Details are given below.

### 2.1. Costs of scenarios

We show here how to calculate the costs of $I$ scenarios applied during $J$ years in a catchment area covering $K$ fields. We use $v_{ijk}$ to denote a vector whose elements describe the farm practices (amount of fertilizer applied, nature of the crop, date of sowing of the catch crop, etc.) for year $j$ and field $k$ when scenario $i$ is applied, $i$=1,. . .,$I$, $j$=1,. . .,$J$, $k$=1,. . .,$K$. As farm practices are year and field-dependent, the practices associated with a given scenario vary between years and between fields. For each value of $v_{ijk}$ we calculate a yield value denoted as $y_{ijk}$ and, when the crop is a cereal, a grain protein content value denoted as $p_{ijk}$. We consider that $y_{ijk}$ and $p_{ijk}$ are calculated by means of crop models. The advantage of using crop models is that these models take into account the soil characteristics of the different fields, the impact of soil nitrogen supply on the nitrogen plant uptake, the climate and the farm practices. Consequently, crop models are likely to give realistic values of yield and grain protein content. However, errors associated with crop model predictions can be substantial due to uncertainty in equations, parameter values and input variables. This leads us to define random variables for describing crop model errors. We denote $\varepsilon^{y}_{ijk}$ and $\varepsilon^{p}_{ijk}$ as the random error terms associated with the values of $y_{ijk}$ and $p_{ijk}$, respectively. The farmer's income per unit of area obtained for year $j$ and field $k$ when scenario $i$ is applied is denoted as $m_{ijk}$ and is calculated from crop characteristics (yield, grain protein content) and from the farm practices by using a function $f$:

$$m_{ijk} = f\left(v_{ijk}, y_{ijk} + \varepsilon^{y}_{ijk}, p_{ijk} + \varepsilon^{p}_{ijk}\right) \tag{1}$$

Various expressions of $f$ that differ on grain price values can be considered (see the case study). As yield and grain protein content are defined as stochastic variables, $m_{ijk}$ is also stochastic. Let $\pi_{ij}$ denote the income at the catchment area level obtained for scenario $i$ and year $j$. $\pi_{ij}$ is defined by

$$\pi_{ij} = \sum_{k=1}^{K} m_{ijk} x_k \tag{2}$$

where $x_k$ is the area of the $k$th field. Cost values are calculated relatively to a baseline scenario. Thus, the cost $C_{ij}$ obtained for scenario $i$ and year $j$ is defined by

$$C_{ij} = \pi_{1j} - \pi_{ij} = \sum_{k=1}^{K} m_{1jk} x_k - \sum_{k=1}^{K} m_{ijk} x_k \tag{3}$$

where $m_{1jk}$ and $\pi_{1j}$ are the incomes obtained for year $j$, for field $k$ and for the catchment area, respectively, when the baseline scenario ($i$=1) is applied. The cost averaged over years for scenario $i$ is then:

$$\bar{C}_i = \frac{1}{J} \sum_{j=1}^{J} C_{ij} \tag{4}$$

As farmers' incomes are defined as stochastic variables, $C_{ij}$ and $\bar{C}_i$ are also stochastic.

*2.2. Nitrate concentration in drained water*

We denote $E_{ij}$ as the nitrate concentration in drained water at the catchment level for year $j$ when scenario $i$ is applied. $E_{ij}$ is predicted in relation to climate characteristics and farm practices by means of crop models.

$$E_{ij} = \frac{\sum_{k=1}^{K}\left(l_{ijk} + \varepsilon_{ijk}^{l}\right)x_k}{\sum_{k=1}^{K} d_{ijk}x_k} \tag{5}$$

where $l_{ijk}$ and $d_{ijk}$ are respectively the nitrate leaching and the depth of drained water for year $j$ and field $k$ when scenario $i$ is applied. Here also, model predictions are expected to be realistic but not exactly equal to true nitrate leaching values. Errors of prediction of $l_{ijk}$ are described by a random term noted $\varepsilon_{ijk}^{l}$. Error of prediction of the drainage amount is neglected.

The nitrate concentration averaged over years is defined by

$$\overline{E}_i = \frac{1}{J}\sum_{j=1}^{J} E_{ij} \tag{6}$$

As the nitrate leaching values associated with the different fields are stochastic, the values of $E_{ij}$ and of $\overline{\mathrm{E}}_i$ are also stochastic.

*2.3. Optimal scenario*

An optimal scenario is defined here as a scenario that minimizes $E(\overline{\mathrm{C}}_i)$ (i.e. the expected cost value) subject to a constraint on the nitrate concentration of the drained water at the catchment level. Because of between-year variability and of uncertainty in crop model predictions, various constraints on nitrate concentration can be defined.

First, constraints can be defined for limiting the nitrate concentration averaged over years $\overline{\mathrm{E}}_i$. Such constraints are relevant in a long-term perspective. They therefore concern aquifers with a long water residence time, i.e. groundwater. As shown above, $\overline{\mathrm{E}}_i$ is stochastic because of the crop model errors. Two approaches can be considered to take into account uncertainty in $\overline{\mathrm{E}}_i$. The first approach consists simply in defining a constraint on the expected value of $\overline{\mathrm{E}}_i$:

$$E\left(\overline{E}_i\right) \leq 50 \text{ mg l}^{-1} \tag{7}$$

where $E(\overline{\mathrm{E}}_i)$ is the expected value of $\overline{\mathrm{E}}_i$. Constraint (7) ensures that the expected nitrate concentration is lower than the European Union limit for drinking water (Council Directive 80/778/EEC). However, this constraint does not take into account the risk due to errors of prediction. In the second approach we require that the environmental pollution constraint be achieved with a certain probability:

$$P\left[\overline{E}_i \leq 50 \text{ mg l}^{-1}\right] \geq \alpha \tag{8}$$

By specifying an acceptable probability level $\alpha$, it is possible to monitor the risk of violating pollution constraint.

With constraints (7) and (8), the nitrate concentration is not necessarily lower than 50 mg $\mathrm{l}^{-1}$ every year. We now present two constraints concerning annual nitrate concentrations that are relevant in the short term, i.e. when the aim is to secure rapid reductions in nitrate concentration. This temporal scale would concern shallow water resources.

The first constraint considers the expected values of the yearly nitrate concentrations:

$$E\left(E_{ij}|j\right) \leq 50 \text{ mg l}^{-1}, \forall j \in [1, \ldots, J]. \tag{9}$$

With constraint (9), the expected value of the nitrate concentration is lower than 50 mg $\mathrm{l}^{-1}$ for each of the $J$ years. Like constraint (7), constraint (9) does not take into account the risk due to crop model errors. The next constraint requires that the environmental pollution constraint be achieved with a certain probability every year:

$$P\left[E_{ij} \leq 50 \text{ mg l}^{-1}|j\right] \geq \gamma, \ \forall j \in [1, \ldots, J] \tag{10}$$

Minimizing the expected value of the average cost defined by Eq. (4) subject to one of the four constraints (7), (8), (9), or (10) gives an optimal scenario.

## 3. Illustration

To illustrate the framework presented above, we consider the Bruyères catchment area (Beaudoin et al., 1999; Mary et al., 1997). This site is located in the intensive cropping region of the Parisian Basin in the northeast of France. It is of moderate size (145 cultivated ha) and its economic activity is exclusively

Table 1
Climatic characteristics: rainfall, air temperature and simulated drainage

| Period | | Rainfall (mm $year^{-1}$) | Temperature (°C) | Simulated drainage (mm $year^{-1}$) |
|---|---|---|---|---|
| 1991–1992 | | 672 | 9.8 | 97 |
| 1992–1993 | | 599 | 9.5 | 217 |
| 1993–1994 | | 939 | 10.0 | 409 |
| 1994–1995 | | 788 | 10.7 | 364 |
| 1995–1996 | | 575 | 9.2 | 40 |
| 1996–1997 | | 680 | 9.6 | 177 |
| 1991–1997 | Average | 709 | 9.8 | 217 |
| 1961–1997 | Average | 695 | 9.7 | 226 |
| 1961–1997 | Minimum | 436 | 8.5 | 40 |
| 1961–1997 | Maximum | 939 | 11.0 | 420 |

agricultural. Crop intensification has been continuous since the 1950s and increased by clearance of grasslands. The nitrate content of water has therefore increased regularly by 1 mg/year since 1975,[1] even though best agricultural practices have been engaged in since 1989.[2]

The unsaturated zone, with a thickness of 20–35 m, is responsible for the slow response to changes in agricultural practices at the outlet. The mean residence time of water has been estimated at 20–25 years. To assess the impacts of changes in agricultural practices on water quality, simulations concerning the nitrate concentration in drained water are needed.

### 3.1. Data

The catchment area covers 36 fields which differ widely due to the variability of the underlying parent material. Four main soil types are present: loam, sand, marl and stones, and limestone. These soil characteristics induce considerable variability of potential yields that are modelled.

The climate is oceanic with a continental influence. The study period covers 6 years: 1991–1997. The averaged temperature and rainfall recorded on the site for these years are close to normal, in spite of a very high level of inter-annual variability (Table 1). The simulations of annual drainage[3] show that the years 1992–1993 were close to the mean recorded in 36 years; 1993–1994 was almost the highest (35 rank); 1995–1996 was the lowest. The study period is thus representative of the 36 years as regards the means and extreme values for drainage volumes.

Table 2
Type and frequency of crops grown in the Bruyères catchment area during the years 1991–1997 and mean sowing and harvest dates

| Crop | Frequency (% area) | Sowing period | Harvest date |
|---|---|---|---|
| Winter rapeseed | 6.1 | 20–31 August | 20 July |
| Winter barley | 12.3 | 20–30 September | 15 July |
| Winter wheat | 37.7 | 1–20 October | 5 August |
| Spring barley | 2.8 | 20–28 February | 31 July |
| Spring peas | 16.4 | 1–10 March | 05 August |
| Sugar beet | 17.8 | 20 March–10 April | 25 October |
| Maize | 1.5 | 20–30 April | 31 October |
| Sunflower | 3.8 | 25 April–5 May | 5 October |
| Covered set aside | 1.5 | | |

The crops and their rotations considered in this work are the ones really practiced by farmers during the 6 years (Table 2). Spring crops account for 41% of the agricultural surface; consequently, bare land is frequent during winter (33% of the surface has a 237-day inter-cropping interval). But the area devoted to spring crops varies with the year, ranging from 29% in 1996–1997 to 71% in 1994–1995.

### 3.2. Scenarios simulated

Six scenarios are considered. Four of them simulate farm management practices that have been suggested by the European Union with a view to reducing agricultural nonpoint pollution. The other two simulate agronomists' recommendations. Main characteristics of these scenarios are displayed in Table 3.

The first scenario comes under the 'code of good agricultural practice' (Council Directive 91/676/CEE).

[1] In 1975, the nitrate concentration of the water-catchment was less than 25 mg $NO_3$ $l^{-1}$, in 2000 it was around of 60 mg $l^{-1}$.

[2] At this date, farmers agreed to change their cropping practices in the frame of a voluntary agreement. They committed themselves to (i) apply the recommended fertilization; (ii) to sow a catch crop before any spring crop; and (iii) bury the crop residues that were poor in nitrogen. Since 1996, this agreement has been supported by Europe as an 'agri-environmental measure'.

[3] The drainage was calculated for a rotation winter wheat–winter barley–sugarbeet–winter wheat–peas, in a soil with a PAW equal to 175 mm (Beaudoin et al., 1998).

Table 3
Main characteristics of the simulated scenarios

| Scenario | Nitrogen fertilization doses | Catch crop | Set aside |
|---|---|---|---|
| *Conv* | Conventional | none | 1.5% of area |
| *Intfert* | Optimum level for yield and pollution | none | 1.5% of area |
| *IntfertC1* | Optimum level for yield and pollution | Sown in September | 1.5% of area |
| *IntfertC2* | Optimum level for yield and pollution | Sown in August | 1.5% of area |
| *RedinpC1* | Optimum level minus 20% | Sown in September | 1.5% of area |
| *RedinpC2* | Optimum level minus 20% | Sown in August | 1.5% of area |
| *Setas* | Conventional | none | 17% of area |

It consists only in limiting application of fertilizers on the land in vulnerable zones, and is not too demanding. This scenario, called 'integrated fertilization' (*Intfert*), aims to optimize yields and reduce the amount of mineral nitrogen present in the soil when water begins to drain. Nitrogen fertilization is calculated using the balance-sheet method called Azobil (Machet et al., 1990) and a measurement of the soil mineral nitrogen reserve.

The other scenarios are more drastic. They are put forward in the framework of the accompanying measures of the Common Agricultural Policy (Council Regulation No. 2078/92/EEC), which introduced support for the 'adoption of environment-friendly farming practices'. This aid would compensate farmers for complying with specific restrictions on farm practices that resulted in a loss of net income. In order to protect water, farmers have to:

- remove a plot of land from production and convert it into grassland (set-aside cross-compliance, hereafter referred to as *Setas* scenario);
- reduce the nitrogen fertilization level by 20% relative to the optimum level and establish catch crops before all spring crops (input-reduction cross-compliance, referred to as *RedinpC1* and *RedinpC2* scenarios);

The set-aside scenario simulated (*Setas*) consists in removing marginal croplands, i.e. lowest productivity fields (grain yields less than 7 metric tons) which are also the most pollutant. This scenario resulted in a reduction in the harvested cropland area of 17% of the study area.

Two scenarios simulate the reduction of inputs coupled with the introduction of catch crops. Sowing catch crops is fairly demanding because of the new constraints on working techniques thus generated. Farmers can postpone sowing of catch crops until they have time, but this may reduce their environmental impact. If catch crops are sown immediately after the harvest, extra labour must operate and the effectiveness rises. Hence, two scenarios of reduction of inputs are simulated according to the catch crops sowing dates (*RedinpC1* for late sowing; *RedinpC2* for early sowing of catch crops).

Since some agronomists have underlined the considerable ability of catch crops to reduce nitrate pollution without any fertilizer reduction (Addiscott et al., 1991; Laurent and Mary, 1992; Thorup-Kristensen and Nielsen, 1998), and because of the loss of net income induced by the input reduction required by the EU, it was useful to test a scenario relaxing this constraint. We therefore considered another requirement in terms of which farmers have to sow catch crops before all spring crops but have not to reduce their fertilization, only to optimize it. Hence, two other scenarios are modelled: *IntfertC1* for late sowing of catch crops and *IntfertC2* for early sowing.

The impact of these six scenarios are assessed relatively to a baseline scenario ($i$=1) simulating conventional farming practices (*Conv*). In the scenario *Conv* the nitrogen doses often exceed the reference dose for the average achieved yield, due to the fact that farmers are risk averse. Hence, the nitrogen supply depends only on the crop type as opposed to the field and the year.

For the purpose of demonstration, the scenarios simulate the full potential (environmental and economic) impacts of the change of practice, independently of farmers' economic behaviour (risk aversion, moral hazards, etc.). In other words, the simulated scenarios are 'first best' situations where new practices are implemented by all farmers, for all fields and according to the strict requirements defined.

### 3.3. Modelling

In the final analysis, the vector $v_{ijk}$ is defined for year $j$=1,. . .6; field $k$=1,. . .36 and scenario $i$=1,. . .7.

The modelled temporal scale is the two-crop succession (18 months); two crop models are run successively, for each value of $v_{ijk}$ in order to achieve the robustness of the simulations. First, values of yield grain protein content and soil mineral nitrogen at harvest are calculated by using the crop model described by Beaudoin et al. (1998) and Makowski et al. (1999). The input variables of this model are soil characteristics, nature of preceding crops and of current crops, and amounts of nitrogen fertilizer applied. Second, values of drained water and nitrogen leached from the autumn to the following spring are calculated by means of the STICS model (Brisson et al., 1998). The input variables of STICS are those of the model of Makowski et al. (1999) and additional input variables, namely following crop, presence or absence of catch crop, daily climate variables (temperatures, rainfall, radiation), soil water content, sowing date and dates of fertilizer application.

Parameter estimation and model evaluation were performed in previous studies for the different crops and catch crops cultivated in the Bruyères catchment area by means of experimental data (Brisson et al., 1998; Beaudoin et al., 1998; Makowski et al., 1999; 2001). Model errors were described by defining model parameters as random variables (Makowski et al., 2001). Model parameters were supposed to be normally distributed. Expected values, variances and covariances of random parameters were estimated by means of experimental databases from the Parisian Basin (Beaudoin et al., 1998; Makowski et al., 2001). The resulting probability distributions describe the uncertainty on parameter values.

The selection of an optimal scenario requires the calculations of $E(\overline{\mathrm{C}}_i)$ and the evaluation of the constraints (7)–(10) for all scenarios. A first approach would be to derive the exact analytical expressions of $E(\overline{\mathrm{C}}_i)$ and of the constraints. This is not possible here because our models are nonlinear and somewhat complex. As a consequence, the true analytical expression of the probability distributions of the error terms $\varepsilon^{y}_{ijk}$, $\varepsilon^{p}_{ijk}$, and $\varepsilon^{l}_{ijk}$ cannot be deduced from the probability distributions of the parameters. Another method would consist in converting the probabilistic constraints (8)–(10) to their deterministic equivalents (Hardaker et al., 1991; Kampas and White, 2003). The idea is to approximate the true distribution of the random variables by a probability distribution that can easily be computed. Kampas and White (2003) proposed three deterministic constraints that could be used to approximate the probabilistic constraints (8)–(10), but this approach is difficult to apply here. Each of the three proposed deterministic constraints has its own limitations. Some are appropriate when the number of individual fields is high and when the variables are unbounded. Here, the number of fields is only equal to 36 and the nitrate emissions are always positive. Other approximations give satisfactory results when all the variables are independent (Kampas and White, 2003). This is not a realistic assumption for the nitrate emissions of the individual fields. Another problem is that computation of the proposed deterministic equivalents requires knowledge of the variances of the random terms. Some of these deterministic equivalents also take into account the covariances of the random terms. But the variances and covariances of the random terms $\varepsilon^{y}_{ijk}$, $\varepsilon^{p}_{ijk}$, and $\varepsilon^{l}_{ijk}$ are unknown.

For all these reasons, the optimal scenario is selected in this case study by means of a Monte Carlo method. We generate 100 values of $\varepsilon^{y}_{ijk}$, $\varepsilon^{p}_{ijk}$, and $\varepsilon^{l}_{ijk}$ from the parameter probability distributions for each field, each year and each scenario. The expected cost values and the constraints are then computed from these samples. The advantages of this method are that it can be implemented with the original probabilistic constraints, and that it only requires the knowledge of the probability distribution of the model parameters. Yet, our method does not assess policy optimality because the probability is determined for fixed policy. Our purpose is only to stress the optimal scenario, i.e. the farm management practices that have the best environmental and economic impacts when they are implemented by all farmers, for all plots of land. This scenario will be selected in heterogeneous conditions and would be the scenario enforced by policy makers.

Values of scenario costs are calculated taking into account yield and protein content values, costs of establishing catch crops, and profit losses due to set-aside farmland. Three assumptions of agricultural policy and market conditions (crop prices and income support) have been made:

– *1997 circumstances*: The crop prices and income supports are those in force during 1997. Note that a small bonus is granted for quality wheat and quality

spring barley. If the protein content is satisfactory, wheat can be used for bread, spring barley can be used in beer, and crop prices are higher;
- *Agenda 2000*: According to the CAP reform agreed on by the EU government in 1999, cereal prices should have decreased by 15% by 2006 and income support increases. A small increase in bonuses for quality cereals is taken into account according to the terms of payment presently in application;
- *Quality bonus appreciation*: Because of the decrease of cereal prices provided for in the CAP reform, the cereal market segmentation could be reinforced. Prospective planning has therefore been foreseen: the Agenda 2000 conditions are associated with a large increase of bonuses for quality cereals.

## 4. Results

The results of cost-effectiveness analysis of the farm management scenarios are shown in terms of the two temporal scales discriminated above. First, we aim to minimize costs of the scenario to achieve an average threshold: the nitrate concentration is limited to the European maximum in the long term. Second, the cost is minimized to achieve a yearly threshold: the nitrate concentration is limited to the European maximum every year.

### *4.1. Probabilistic cost-effectiveness in the long term*

Expected costs vary widely between the scenarios (Table 4). Yet for most of the scenarios, the expected cost represents less than 10% of the crop gross margin. The scenario with set-asides (*Setas*) is the only exception. The expected cost of this scenario represents 48% of the gross margin.

Table 4 allows us to evaluate costs due to 'integrated fertilization', 'input reduction' and 'catch crop introduction' practices. Integrated fertilization induces a loss of earnings because, in the first place, yields decrease. With peas no loss is induced because this crop is never fertilized. With sugar beet the loss is small and savings are sometimes possible (in 1994, for example). This result highlights potential win–win practices, i.e. win for water quality and win for farmers' income. It indicates input inefficiencies in actual farm management practices.

Compared to the scenarios with 'integrated fertilization' practices (*IntfertC1*, *IntfertC2*), the costs of the scenarios with 'input reduction' practices (*RedinpC1*, *RedinpC2*) are higher by 7 to 17€ $ha^{-1}$, depending on agricultural policy and market prices. These scenarios are more costly because a 20% reduction of fertilization induces large yield losses (especially for cereals) and a decrease of the grain protein contents that do not meet the quality requirements of agro-industrial firms. The costs obtained for the *Intfert* scenario show that the use of catch crops raises costs by 20€ $ha^{-1}$ if they are sown late and by 23€ $ha^{-1}$ if they are sown just after the harvest in August. The *Setas* scenario appears to be very expensive despite the less productive fields have been removed.

The costs of scenarios are about the same for the economic contexts '1997 circumstances' and 'Agenda 2000'. By contrast, costs are much higher

Table 4
Expected costs of scenarios in the long term (€ $ha^{-1}$ $year^{-1}$)

| | *Intfert* | *IntfertC1* | *IntfertC2* | *RedinpC1* | *RedinpC2* | *Setas* |
|---|---|---|---|---|---|---|
| $E(\overline{C}_i)$ 1997 circumstances | 6.3 | 26.0 | 29.6 | 33.5 | 37.0 | 149.3 |
| $E(\overline{C}_i)$ Agenda 2000 | 6.5 | 26.2 | 29.8 | 33.6 | 37.2 | 153.0 |
| For wheat | 4.4 | 34.8 | 40.3 | 39.3 | 44.8 | 362.2 |
| For sugar beet | 2.6 | 2.6 | 2.6 | 19.9 | 19.9 | 1498.1 |
| For peas[a] | 0 | 0 | 0 | 0 | 0 | 396.9 |
| For winter barley | 19.8 | 65.3 | 73.5 | 89.1 | 104.2 | 309.5 |
| $E(\overline{C}_i)$ quality bonus appreciation | 15.1 | 34.9 | 38.5 | 52.4 | 56.0 | 155.8 |

The costs calculated for each assumption of agricultural policy and market conditions are given for the whole catchment. For the 'Agenda 2000', they are also given for each crop separately.

[a] Peas are not fertilized, so they are not concerned by the integrated fertilization, nor the reduced fertilization. Moreover, this crop is never followed by a spring crop, so no catch crop is sown.

for the economic context ‘Quality bonus’. The difference is particularly large for the *RedinpC1* and *RedinpC2* scenarios. These two scenarios are characterized by a reduction of the amounts of applied fertilizer. This reduction decreases the grain quality of wheat crops and therefore the prices at which wheat is sold.

Table 5 shows that conventional agricultural practices (*Conv*) generate a high average value of nitrate leaching. This is due to over-fertilization and to a high proportion of spring crops within the study area.

Table 5 also shows that changes in cropping practices can reduce water pollution. Integrated fertilization (*Intfert*) decreases the nitrate concentration by about 6 $NO_3$ mg $l^{-1}$. The introduction of catch crops is more efficient. It decreases nitrate concentration by 15 to 25 mg $l^{-1}$ depending on the date of sowing (*IntfertC1*, *IntfertC2*). A reduction of fertilizer doses (*RedinpC1*, *RedinpC2*) decreases the nitrate concentration only by 4 mg $l^{-1}$. The introduction of fallows (*Setas*) decreases the nitrate concentration by 18 mg $l^{-1}$.

The two scenarios with early catch crops (*IntfertC2*, *RedinpC2*) satisfy the constraint (7). The *RedinpC1* scenario is almost satisfactory. Because of uncertainty in crop model predictions, only the *IntfertC2* and *RedinpC2* scenarios satisfy constraint (8) with a high probability level.

Which scenario is optimal in the long run? The results displayed in Tables 4 and 5 show that the *IntfertC2* scenario is the most cost-effective. It is for this scenario that the environmental constraints are satisfied at the lowest cost, irrespective of the simulated economic context. Our results show that the sowing dates of the catch crops have a strong influence on the nitrate concentration. Thus, it seems important to define these sowing dates precisely in the administrative measures concerning the reduction of nitrate pollution.

Table 6
Annual expected costs of scenarios (€ $ha^{-1}$) for Agenda 2000 assumption

| | *Intfert* | *IntfertC1* | *IntfertC2* | *RedinpC1* | *RedinpC2* | *Setas* |
|---|---|---|---|---|---|---|
| $E(C_{i1991})$ | 6.0 | 25.7 | 29.3 | 33.5 | 37.1 | 150.6 |
| $E(C_{i1992})$ | 6.2 | 26.0 | 29.6 | 32.6 | 36.2 | 151.6 |
| $E(C_{i1993})$ | 6.4 | 26.1 | 29.7 | 33.9 | 37.4 | 153.1 |
| $E(C_{i1994})$ | 6.7 | 26.4 | 30.0 | 33.4 | 37.0 | 149.9 |
| $E(C_{i1995})$ | 6.4 | 26.2 | 29.8 | 33.5 | 37.1 | 154.3 |
| $E(C_{i1996})$ | 7.0 | 26.8 | 30.4 | 34.4 | 38.0 | 151.7 |

### 4.2. Probabilistic cost-effectiveness under climate variability

Table 6 shows the expected costs of scenarios for the 6 years considered in the case study and for the ‘Agenda 2000’ assumption. Although climate variability is very high, the variability of costs between years is relatively low. Similar results are obtained for the other market and policy assumptions. This is due to the fact that the costs displayed in Table 6 represent differences of incomes relative to the baseline scenario (*Conv*). The climate has the same influence both on the baseline scenario and on the other scenarios. For instance, a dry year decreases the yield values both for the baseline scenario and for the other scenarios. Consequently, the year factor has only a small influence on the cost values.

By contrast, nitrate concentration values vary widely between years (Table 7), especially for conventional practices (*Conv*): the difference between the maximum and the minimum nitrate concentration is equal to 69.5 mg $NO_3$ $l^{-1}$. The variability is lower for all scenarios, especially for those that include catch crops. The variability is particularly low for the *RedinpC2* scenario. For this scenario, the difference between the maximum and minimum value is only equal to 29.1 mg $NO_3$ $l^{-1}$.

No scenario can reduce the nitrate concentrate below 50 mg $NO_3$ $l^{-1}$ every year. Constraints (9) and

Table 5
Expected concentration and abatement (mg $NO_3$ $l^{-1}$); probability of achieving EU target in the long term

| | *Conv* | *Intfert* | *IntfertC1* | *IntfertC2* | *RedinpC1* | *RedinpC2* | *Setas* |
|---|---|---|---|---|---|---|---|
| $E(\overline{E}_i)$ | 77.0 | 70.3 | 55.2 | 44.8 | 50.8 | 41.0 | 58.7 |
| $E(\overline{E}_i-\overline{E}_1)$ | 0.0 | 6.7 | 21.8 | 32.2 | 26.2 | 36.0 | 18.3 |
| $P[\overline{E}_i \leq 50 \text{ mg } l^{-1}]$ | 0.00 | 0.00 | 0.00 | 1.00 | 0.23 | 1.00 | 0.00 |

Table 7
Annual expected concentration (mg $NO_3$ $l^{-1}$)

| | *Conv* | *Intfert* | *IntfertC1* | *IntfertC2* | *RedinpC1* | *RedinpC2* | *Setas* |
|---|---|---|---|---|---|---|---|
| $E(E_{i1991})$ | 56.3 | 49.0 | 36.1 | 31.1 | 33.7 | 29.2 | 33.8 |
| $E(E_{i1992})$ | 84.4 | 76.2 | 66.5 | 54.6 | 61.0 | 49.4 | 66.5 |
| $E(E_{i1993})$ | 92.9 | 84.0 | 72.9 | 58.3 | 67.1 | 52.9 | 88.6 |
| $E(E_{i1994})$ | 109.6 | 103.3 | 56.8 | 37.7 | 50.9 | 33.6 | 80.8 |
| $E(E_{i1995})$ | 40.1 | 38.1 | 32.6 | 26.9 | 31.4 | 26.0 | 21.0 |
| $E(E_{i1996})$ | 78.6 | 70.6 | 66.5 | 60.6 | 60.9 | 55.1 | 61.9 |
| Range | 69.5 | 65.2 | 40.3 | 33.7 | 35.7 | 29.1 | 67.6 |

The '1991' index means the drainage occurring during the 1991–1992 winter period.

(10) are never fully satisfied (Tables 7 and 8). The best result is obtained for the *RedinpC2* scenario. With this scenario, the EU threshold is satisfied for four of the 6 years considered in this study. Note, moreover, that in the four scenarios that include catch crops, maximum concentration levels can be reduced in those years in which climatic and crop conditions are most unfavourable, as in the years 1993–1994 and 1994–1995. During those years rainfall is high (see Table 1) and a large part of the catchment area is allocated to spring crops (71% of the area in 1994–1995). Nitrate concentration reaches high values during those years with the *Conv* scenario, but water drainage and consequently nitrate leaching are strongly reduced when catch crops are sown.

Table 9 shows that if we isolate high-quality soil (loams), 50 mg $l^{-1}$ will never be exceeded with the implementation of one of the two scenarios involving a catch crop sown early. With *IntfertC2*, the maximum concentration reached during the period 1991–1997 would be 41 mg $l^{-1}$. With *RedinpC2* it would be 38 mg $l^{-1}$. For other types of soil these scenarios would allow a more substantial reduction in pollution but would not allow the environmental objective to be met every year.

## 5. Discussion

In the area studied in this paper, the European standard can be achieved only in the long run. None of the scenarios tested in the case study satisfies the nitrate concentration constraint every year. This result shows that it is not realistic to obtain a rapid reduction of nitrate concentrations by implementing the farm management practices suggested by the EU. Integrated practices of fertilization, reduction of the amounts of fertilizer applied, and introduction of catch crops and of fallows are not sufficient to satisfy the nitrate concentration constraint every year. Thus, our results underline the limits of an environmental policy, in such a context. To satisfy the environmental constraint every year, the production system would have to change, for example intensive rotations would have to be greatly reduced. Such changes in farm resource allocation cannot be conceived of without modifying the market incentives and income support of the CAP. In other words, environmental policy and agricultural policy have to be integrated.

In the long run and irrespective of the simulated economic context, the optimal scenario in the studied case is the one that combines integrated fertilization

Table 8
Annual probability of achieving EU target

| | *Conv* | *Intfert* | *IntfertC1* | *IntfertC2* | *RedinpC1* | *RedinpC2* | *Setas* |
|---|---|---|---|---|---|---|---|
| $P[E_{i1991} \leq 50\text{mg l}^{-1}]$ | 0.00 | 0.70 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| $P[E_{i1992} \leq 50\text{mg l}^{-1}]$ | 0.00 | 0.00 | 0.00 | 0.12 | 0.00 | 0.60 | 0.00 |
| $P[E_{i1993} \leq 50\text{mg l}^{-1}]$ | 0.00 | 0.00 | 0.00 | 0.01 | 0.00 | 0.20 | 0.00 |
| $P[E_{i1994} \leq 50\text{mg l}^{-1}]$ | 0.00 | 0.00 | 0.01 | 1.00 | 0.39 | 1.00 | 0.00 |
| $P[E_{i1995} \leq 50\text{mg l}^{-1}]$ | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| $P[E_{i1996} \leq 50\text{mg l}^{-1}]$ | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.03 | 0.00 |

The '1991' index means the drainage occurring during the 1991–1992 winter period.

Table 9
Maximum of annual expected concentration (mg $NO_3$ $l^{-1}$) for the different types of soil

| | *Conv* | *Intfert* | *IntfertC1* | *IntfertC2* | *RedinpC1* | *RedinpC2* | *Setas* |
|---|---|---|---|---|---|---|---|
| Loam | 72.6 | 65.5 | 52.3 | 40.7 | 49.5 | 38.3 | 72.6 |
| Sand | 197.7 | 157.5 | 130.2 | 114.1 | 119.5 | 104.6 | 93.5 |
| Marl and stones | 89.3 | 86.3 | 72.0 | 63.4 | 63.9 | 56.2 | 59.1 |
| Limestone | 141.2 | 135.5 | 90.0 | 57.5 | 80.1 | 49.3 | 130.4 |

with planting of catch crops immediately after the harvest. In order to reduce the nitrate level below the 50 mg $l^{-1}$ critical level, this scenario is less expensive than water purification. Water purification costs (depreciation and operating expenses) have been evaluated at 0.27 €/$m^3$ on average in France (Lacroix and Balduchi, 1995); for small rural districts, this cost has been estimated at between 0.3 and 0.5 €/$m^3$. In the case study, the cost of the optimal scenario (taking into account domestic water consumption in the study area) is in the range 0.06–0.08 €/$m^3$, depending on the economic context. Other economic studies report similar results: in the long run, agricultural measures are cheaper than water treatment (De Haen, 1990; Guillemin and Roux, 1992; Mollard, 1997).

Our results highlight the considerable effectiveness of catch crops which really do act as a buffer against climatic, cropping and soil conditions. Because they catch most of the nitrogen remaining in the soil at harvesting, they are able to reduce variability of nitrate concentrations and thus substantially reduce the risk of exceeding the environmental constraint.

These results were obtained for an area that is representative of the Paris basin. Thus, our results are valid for most of the catchment areas located in this basin. They may even have a more general scope in so far as they are consistent with the results of other studies. Machet et al. (1997) and Justes et al. (1999) also highlight the effectiveness of catch crops. In particular, they emphasize the fact that these crops mitigate impacts of farmers' prediction error: climate variability makes yield predictions and therefore nitrogen requirement predictions uncertain. Vatn et al. (1997) show that catch crops have substantial effects on losses and that they are particularly well suited to cereal-growing areas. In fact, they are more cost-effective than any other measure.

Many studies confirm that in such agricultural contexts the other solutions normally envisaged fail. Thus, results that converge with ours are obtained on the setting-aside of land. This practice is very costly and should be applied only if their other environmental benefits (improve the biodiversity, reduce the soil erosion, etc.) could be valorized. The scenarios of land retirement in the USA simulated by Ribaudo et al. (1994) generate higher social costs than water quality benefits. Other results show that the setting-aside of land can have effects that are difficult to control. For example, it may result in shifts in regional crop production and crop mix (Ribaudo et al., 1994) and so in pollution transfers. It may also have undesirable effects in the long term (Meissner et al., 1998).

Other research, apart from ours, also shows the low level of effectiveness of fertilizer reduction. Weaver et al. (1996) show that for field cropping in a Pennsylvania setting, a 10% decrease in fertilizer application rates has no statistically significant effect on the pollution level. Pan and Hodge (1994) show that in order not to exceed 50 mg $l^{-1}$ in an area of intensive agricultural production in eastern England, nitrogen application must be reduced by 81%. But is such reduction realistic? From this point of view, Ribaudo et al. (2001) show that in the Mississippi Basin, reducing nitrogen fertilizer use by up to 50% is more costly than an alternative strategy usually considered as very expensive, i.e. wetlands restoration in order to filter nutrients coming off cropland. These different results highlight the fact that for field crops, fertilizer inputs are often close to optimum levels, and their reduction can rarely be substantial without incurring high costs.

Although restrictions on fertilizer application rates have a limited effect on pollution, this measure is unavoidable for any pollution control policy. It seems to be indispensable to offset farm operators' inefficiencies since they do exist, as we have shown. It furthermore helps to draw farmers' attention to the non-separability of production, pollution and abatement. But in most cases, in order to comply with EU environmental constraints, it has to be combined with other production practice requirements. From this point of view, the fact of combining it with catch crops seems fully relevant. Yet our results show that it is more cost-effective to demand early sowing of catch crops than to demand 20% reductions of nitrogen inputs, as in the EU specifications. The following recommendations can therefore be made to policy-makers: control of the implementation of these specifications must pay more attention to the date of introduction of catch crops than to the reduction of inputs. Note, moreover, that it is easier to control the date of sowing than to control reduction of inputs. This could help to reduce the administrative costs of this agri-environmental measure.

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