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INTEREST OF SITE-SPECIFIC POLLUTION CONTROL POLICIES

THE CASE OF NITRATE POLLUTION FROM AGRICULTURE

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The case of nitrate pollution from agriculture

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Abstract:

Owing to increasing environmental concerns the current trend is to bend technical production systems in order to adapt them to the specific characteristics of the milieu and diversify them. Inherent to such dynamics is the issue of how to design the accompanying environmental policies. Theoretically, spatially targeted environmental policies are considered optimal, since economic agents tune their efforts according to the sensitivity of the milieu where they operate. But, according to empirical analyses, this advantage is undermined by the high cost of implementation, monitoring and enforcement. This paper outlines the conditions required for site-specific policies to be effective at least cost.

Our starting point is the nitrate pollution of water from agriculture, which varies according to climate, soil type and agricultural production system. Farm management practices enabling to reduce pollution depend on this variability. An interdisciplinary study of the efficiency of differentiating the way this pollution is regulated was carried out on two sites in France. It focussed on assessing the importance of spatial variability in physical parameters and in private and social costs.

Keywords: nonpoint pollution, site-specific technology, site-specific environmental policy, abatement costs, transaction costs

JEL classification codes: C15, H71, Q16, Q25

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In all industrialised countries, the constant improvement in farming productivity and volumes produced over more than 50 years has been achieved through continuous increases in production by surface unit and intensive use of inputs in order to contain the impact of natural random factors. So was the case with the widespread use of chemical fertilizers to maximise yield on all soil types, even low-fertility soils; the development of irrigation to offset the lack of rain or lack of available soil water capacity; the selection of varieties suited to regional conditions, etc… Intensification and mechanisation went on together with greater acreage per farm and with land consolidation, hence relegating the role of agro-ecosystems to that of providing production with neutral and passive support.

Alongside the process of making techniques uniform, an equally uniform agricultural policy has evolved. This was based mainly on price support but offered little differentiation with respect to farm type and was proportional to the quantities produced. Only subsidies awarded to offset natural handicaps and subsidies for rural development have been able to counteract the uniform treatment applied by such policies as this funding takes into account the diversity of farming areas, notably mountain areas.

In the eighties, this long lasting evolution resulted in negative environmental impacts on agro-ecosystems: water and soil pollution, air pollution, erosion problems, decrease in biodiversity, etc... A curve was taken at the start of the nineties, outlining the bases for less intensive and more sustainable farming and aiming to, at least, correct excessive farm practices with significant negative impacts. This switch to more sustainable agriculture is based on more diversified technical systems, more suited to the physical characteristics in which the farming activity is being developed. Thus, taking into account the environment seems to mark the end of standardised technical systems. Indeed, *any environmental problems* – including those with more global effects, like the contribution of agriculture to the greenhouse effect - *always start locally*, even if this is not always easy to identify (nonpoint source pollution). If we consider that anthropogenic emissions of pollution are necessarily localised, and therefore different from one place to another, then the different vulnerability of milieus must be taken into account and solutions must be suited to the local production conditions defined. But this also raises the question of public policies able to take into account different contexts.

Using the example of the nitrate pollution of water, this paper shows that the environmental impact of farming is highly heterogeneous, mainly due to differences in climate, soil types and cropping systems (section 1). Taking into account such spatial and temporal heterogeneity, the technical solutions to be implemented to efficiently reduce pollution differ from one site to the next, according to local conditions (section 2). This raises the following question: should not incentive policies be spatially-targeted? Furthermore, there are already regulations specific to certain areas: “nitrate vulnerable zones”, “areas with nitrogen structural excesses”, etc… and the new European framework directive on water (Council Directive 2000/60/EEC) outlines the prospect of management techniques adapted to different hydrographic contexts. Yet most economists are not convinced that such policies are appropriate. Some results presented in the economic literature are examined (section 3) and the lessons that can be learnt in view of implementation of such policies are outlined (section 4).

1. Spatial and temporal heterogeneity of nitrate pollution

To underline this heterogeneity, a purposefully simple and restricted case is studied: that of water and nitrogen flows under rootzone of maize crops, at regional farming level over an area of 440 km².

This study area corresponds to the plains of Bièvre and Liers (half way between Lyon and Grenoble, south-east of France), which constitute the upstream part of the Bièvre-Liers-Valloire watershed. Together these plains have substantial groundwater resources in terms of volume and use. Nitrate concentration in the water sampled increased considerably over the eighties and nineties. Today, level has reached the critical value of 40 mg/l of nitrate, while values exceeding the European standard of 50 mg/l have also been recorded. This development should be considered at the same time as farming evolution, which is nearly the only economic activity in this area: meadows have been pushed back to make way for forage crops, industrial crops have increased, etc… Today, the farms mainly focus on cattle raising and cash crops, with maize being the most frequent type: in 2000, it accounted for 21% of the utilised agricultural area of the plain studied.

One of the focuses of our research on this site (Sauboua 2001) was maize continuous monocropping, which has repeatedly been pointed out as a prominent source of pollution: it encourages over-fertilization since the maize yield is not negatively affected by excess nitrogen and the long intercropping interval leaves the soil bare throughout the drainage period. The aim of the analysis was to quantify the variability of the water and nitrogen flows in crop fields by modelling various farm practice scenarios in different pedo-climatic contexts representative of the area studied.

The model used (STICS, which stands in French for multi-disciplinary simulator for standard crops), was developed by INRA and simulates the impact of climate, soil and crop management on vegetal production and on environment (water, carbon and nitrogen cycles). It provides a reliable prediction (Brisson *et al.* 2002; Brisson *et al.* 2003; Ruget *et al.* 2002) of nitrate concentration in water drained below the root zone. Under hypothesis of nitrates conservation below this depth (low microbiological activity, aerobic conditions that are unfavourable to de-nitrification), the quality of the groundwater resource is directly affected by the quality of water percolating out of the root zone.

Simulations were performed for ten climatic years and five soil classes. The period studied (1988-1998) gives a good illustration of the climate variability with the average yearly temperatures (from 10.5 to 12°C) and water balance (yearly values ranging from –365 mm for the greatest water deficit, to +580 mm). Although the study area presents a certain amount of geomorphological and pedological consistency, five soil families can be distinguished (old alluvial terraces, recent alluvial plains, colluvial deposits, silt and soil on till).

Depending on these simulation conditions, the expected nitrate concentration of the water drained under non-irrigated maize is 180 mg/l when it is managed according to the usual farm practices of this area[3]. As well as a high level of concentration, the simulations exhibit considerable variability according to soil class and climatic year: the coefficient of variation for the concentration of drained water is around 45 %, with a wide range of variation (275 mg/l of nitrate).

Influence of physical variability

Although the soil types studied here are not very different[4], the results show considerable heterogeneous behaviour between soil classes (see figure 1). The alluvial plains corresponding to the types with the shallowest soil constitute the most vulnerable soil class: the nitrate content in percolated water systematically exceeds 200 mg/l on average and shows the highest degree of variability. On the other hand, the moraine soils stand out for their relatively low pollution

[3] In particular, the mineral fertilization practice considered here consists in split application of 160 units of mineral nitrogen per hectare.

[4] Owing to the homogeneity of the pedogenic context, all soil types are brunisol.

level and show less variability. This observation ties in with the results of other studies (Gorres and Gold 1996; Dubus *et al*., 2003), which underline the differences in behaviour between quite comparable soils.

Furthermore, the concentration in drained water varies greatly according to climatic year (see figure 2). If we take into account all five classes of soil, under non-irrigated maize it varies from 50 mg/l in 1994 (very rainy winter) to 270 mg/l in 1996 (dry winter). Indeed, the climatic conditions determine the amount of drainage and thus the higher or lower dilution of nitrates in the drained water.

An analysis of variance on both soil and climate factors shows that their relative effect is very different according to the process considered: the variance observed in humus mineralization is almost entirely due to the soil while the climatic year, on the other hand, is responsible for most of the drainage variance. However, the other processes modelled using the soil/plant/atmosphere system are influenced to a lesser degree by one or other of these factors. The predominant influence of the pluviometry characteristics on the water balance and of the soil properties on the nitrogen balance has also been shown in other studies (Muttiah and Wurbs, 2002; Thorsen *et al.* 2001).

These results illustrate the importance of taking into account the spatial and temporal variability of the physical characteristics with respect to crop system behaviour. Spatial variability has been long underlined (Addiscott, 1993) and widely studied, notably thanks to the development of Geographic Information Systems. On the other hand, climatic variability has been the subject of few studies from agronomic point of view and its influence is hardly ever taken into account in decision-making, except for flood forecasting.

Influence of farm practices

Even if pedo-climatic conditions play a significant role, the management of cropping systems can extend or reduce the effect of these physical conditions. To emphasize the anthropogenic impact, different maize management scenarios were simulated: irrigated/non-irrigated, presence/absence of manure, usual fertilization practice and reduced fertilization[5].

Of course, irrigation significantly increases yield and consequently greatly lowers variability (see figure 3). However, it hardly influences the level of nitrate concentration in drainage water, or its range of variation. Applying manure increases the level of concentration, undoubtedly owing to the fact that this type of fertilizer is badly taken into account in the long term (increase of organic matter in soil). Reducing fertilization has only a very slight negative effect on yield, since the farming practices of the area studied lead to a definite over-fertilization. However, this reduction helps to substantially lower the level and variability of the concentration.

This case study shows the range of variation in nitrate pollution under the effect of pedo-climatic conditions and cropping system management. More specifically, nitrate pollution presents spatial disparities, which may lead to define high-risk areas (*e.g.* shallow soil), and temporal disparities that should discourage from reasoning on a year-to-year basis. Managing this pollution therefore consists in modifying the anthropogenic impact by taking into account both the location and the time unit of such management.

2. Differentiating solutions to be implemented

[5] Usual fertilization practices for non-irrigated maize consists in 160 units of mineral nitrogen per cultivated hectare or, when manure is added: 140 units + 40 tonnes of cattle manure every other year; for irrigated maize, 210 and 180 mineral units respectively. Reducing fertilization consists in adjusting the application of fertilizers according to the average yield obtained, *i.e.* for non-irrigated maize: 70 units of mineral nitrogen or 0 when manure is applied; for irrigated maize: 120 units of mineral nitrogen or 30 when manure is applied.

To reduce nitrate pollution, more or less radical changes to farming production systems have been planned by agronomists and/or economists, as well as by public authorities. The solutions recommended range from simply reducing the nitrogen input to managing the land use (Lacroix, 1995). Our own interdisciplinary investigations have not tested and assessed the entire range of solutions, only those solutions which are the simplest and the most frequently referred to or implemented: fertilization reduction, management of the intercropping interval and set-aside. These have been implemented on two sites: the Bièvre-Liers plain referred to above and the small watershed of Bruyères-Montbérault in the north-east of France.

The first of these sites underlined that the cost of implementing these solutions can be highly variable (Bel *et al.* 2001). Hence, a program for reducing fertilization and managing the intercropping interval[6] applied to all farms on the Bièvre-Liers plain would generate costs per kilogramme of abated nitrogen ranging from 1 to 7 from one farm to the next. The average cost is €1.50 per kilogramme; less than 10% of the farms have a cost below €1 per kilogramme; 1/3 of them exceed €1.70. This inter-farm variability depends on the context in which the farmers produce and, notably, the combination of different cropping systems they operate. When cattle is raised and spring crops grown, especially maize, costs are lower. On the other hand, costs rise in relation to the amount of winter crops and meadows.

This observation underlines the importance of choosing the appropriate solution among those available and of being attentive to the conditions of its implementation in order to minimise the cost of reducing pollution.

This was the objective of our research on the second site (Lacroix *et al.* 2003). More specifically, we aimed to assess the farm practices encouraged by the European Union in order to reduce nitrate pollution. The method consisted in optimising the economic impact of various farm practice scenarios subject to environmental constrains. This impact was optimised through integrated modelling (STICS linked with an economic model), taking into account yield uncertainty and climatic variability.

Six farm practice scenarios were simulated and compared with the usual practices of the farmers in the region studied (*Conv* for *Conventional* scenario):

- the *Intfert* (for *Integrated fertilization*) scenario, which comes under the “code of good agricultural practice”, aims to optimise yield and the level of fertilization;
- the *Redinp* (*Reduction of inputs*) scenarios, in which the nitrogen fertilization is reduced by 20% and a cover crop is sown before all the spring crops. In practice, this cover crop can be sown immediately after harvesting (*RedinpC2*) or later according to the farmer’s availability (*RedinpC1*);
- the *IntfertC1* and *IntfertC2* (*Integrated fertilization*) scenarios offer a variant of the two afore-mentioned scenarios in which the nitrogen input is simply optimised rather than reduced;
- the *Setas* scenario consists in removing from production the less productive plots, which also generate the most pollution. Hence, 17% of the utilised agricultural area of the watershed is withdrawn from production.

[6] This program specifically consists in

- adjusting nitrogen fertilization (both mineral and organic) according to the average crop yield, *i.e.* the average reached over the last five years,
- systematically using a cover crop to catch the nitrates before the spring crops,
- making sure harvesting residue is better managed, notably by crushing up and burying the maize stalks.

The optimum scenario, *i.e.* the scenario which achieves the European standard specifying a 50 mg/l nitrate concentration at least cost, was determined for two different time horizons: in the medium term and in the short term.

In the medium term, the environmental constraint is to achieve average compliance with the standard over the period considered (1991-97). This is an adequate time scale for the aquifers with a long water residence time, just like groundwaters. In this case, several scenarios would allow a significant reduction in pollution (see table 1). The optimal scenario consists in combining integrated fertilization with cover crops (*IntfertC2).*

In the short term, the environmental constrain is stricter: the standard must be matched for each year in the period. This reasoning concerns aquifers with a high level of renewal, like shallow waters, and for which pollution must be reduced quickly. For this time scale, and for the entire watershed, no scenario is effective (see table 2). However, if only the good quality soils (silt) are taken into account, then the *IntfertC2* is the optimal scenario. Furthermore, it also substantially reduces pollution in other soil types.

More generally speaking, these results show that:

- set-aside is a very costly strategy which only makes sense if the other environmental benefits it offers (increase in biodiversity, less soil erosion) can be valorised;
- reducing fertilization in an intensive crop context only offers a small reduction in pollution. This result ties in with those of other studies carried out in similar contexts (Pan and Hodge 1994; Ribaudo *et al.* 2001; Weaver *et al.* 1996). For field crops, fertilizer inputs are often close to optimum levels, and their reduction can rarely be substantial without incurring high costs. Conversely, in areas where cattle is raised, fertilizers reduction can be more effective owing to the possibility of substituting mineral fertilizers with manure and to the fact that there are technical inefficiencies;
- using cover crops proves to be very effective: these substantially reduce inter-annual variability, notably by decreasing the concentration in years when the climate conditions and the farming context are most unfavourable. Because of this, they do act as a buffer against climatic, cropping and soil conditions. However, this solution proves to be insufficient when it comes to improving waters with fast renewal cycles, except when the soil conditions are favourable.

In pedological conditions with higher risks, more drastic scenarios should be envisaged in order to obtain rapid reductions in pollution. Studies (Szoege *et al.* 1996; Trabada-Crende and Vinten 1998) show that, in the highly polluted areas of the United Kingdom, substantial modifications in land use (less intensive crops and more meadows, forests, etc…) are required in order to reduce nitrate concentrations below 50 mg/l.

The heterogeneity of physical parameters has a strong influence on costs and on the effectiveness of measures to reduce pollution. To be cost-effective, techniques must clearly be adapted to local conditions, especially to the hydrological, pedological, climatic and farming context (relative share of different cropping systems), as well as to the landscape (amount of uncultivated soil, amount of forest, etc…). This result points out the challenge of adding a spatial dimension to environmental policies that could benefit from this heterogeneity. Yet, studies carried out on spatially-targeted policies do not all point to economic benefits. This is why a critical examination of these studies is performed in the next section.

3. What we can learn from literature about spatially-targeted environmental policies

Applying a spatial focus to a policy consists in differentiating space according to the objectives to be reached and the resources to be implemented in order to abate more where it will be most effective and least costly. In other words, it means encouraging economic agents to modulate their efforts according to the sensitivity of the milieu where they operate. According to a number of studies, spatially-targeted policy does not reduce the abatement cost of pollution significantly as compared with a uniform policy. Furthermore, it generates high transaction costs. The survey of literature will therefore distinguish these two types of costs and aims to outline the conditions required for spatial policies to be cost-effective.

Results on abatement costs

In theory, whether the objective is the efficient level of pollution (Pigou, 1920) or the cost minimization of the abatement (Baumol and Oates, 1971), uniform taxes imposed on pollution emissions can fulfill the objective. The tax rate is considered uniform, since it is based on models that do not take into account the complex nature of relations within ecosystems. However, authors who have attempted to introduce spatial heterogeneity of pollution into their theoretical models (Tietenberg 1974; Xepapadeas 1992) have shown that the optimum scheme is one which applies a spatially discriminated tax rate.

Empirical studies confirm that spatially differentiated policies offer economic advantages in relation to uniform policies. The results presented in table 3 indeed show that targeted policies offer lower pollution abatement costs, but in highly varying proportions depending on the study considered. An analysis of the results helps to explain the extent of such cost reductions, which depends on:

- the *degree of heterogeneity* taken into account. Indeed, the reduction is all the greater the more highly differentiated the physical characteristics. Hence, Mapp *et al.* (1994), who study five different regions of the United States, clearly show that the spatial solution is far superior. Conversely, Helfand and House (1995), when they study two silty soils with little difference between them, obtain only small savings;
- the *level of detail* of this heterogeneity. Braden *et al.* (1989) underline that the more the spatial approach takes into account the complex nature of the environmental reality, the greater the savings on pollution abatement costs;
- the *population targeted* by public policies. The small difference in costs illustrated by Fleming and Adams (1997) can be attributed to the fact that one of the four areas studied contributed massively to reducing pollution, in both policies envisaged, and that this area represents over half the surface of the area studied.

In fact, to valorise the advantages of targeted policies, the spatial scale must clearly differentiate abatement costs and only geographic areas with the lowest costs should contribute to abating pollution. This point of view ties in with the results obtained through experiments in which tradable emission permits were applied. Although, in theory, regulation instruments based on prices or quantities are equivalent (Weitzman, 1974), applications show that, in reality, permits are more efficient as their greater flexibility allows better use of the heterogeneous nature of the abatement costs (Dwyer, 1992).

To design these policies properly, detailed spatial information is necessary, like the amount of damage or cost of reducing pollution. Given how difficult it is to collect such information, Fleming and Adams (1997), Helfand and House (1995) consider that applying a spatial approach to policies is an advantageous but impractical solution. Other authors consider that this information collection and detailed policy design could engender costs that are too high.

Results on transaction costs

Spatial policies are supposed to generate high transaction costs stemming from their implementation (information, design and contractual agreement) as well as their monitoring.

McCann and Easter (1999) recall that the level of these transaction costs depends on the number and diversity of agents involved, the technology available, the policy under consideration (notably according to the instrument chosen, the buy-in or resistance of agents with respect to this regulation, the duration of regulation, etc…) and the amount of abatement or the size of the transaction. Vatn (1998) underlines that transaction costs also depend on what the emissions are made up of (one or several chemical compounds), and their environmental impact (identical or geographically different).

It is obvious that the degree of heterogeneity of the agents taken into account (through their technological diversity, the diversity of their environment, the diversity of the techniques used to reduce pollution, etc…) has a specific impact on the cost of the information required to design the policy. However, the monitoring costs should not be particularly affected by giving a spatial focus to policies, since they depend in principle more on the instrument used, the number of agents concerned and the target abatement in pollution.

Basing their reasoning on a given type of instrument, *i.e.* a contractual agreement to use best farming practices, Carpentier *et al.* (1998) highlight that through targeting the farms to be subjected to this contractual agreement, the target population is restricted and the transaction costs significantly reduced, especially thanks to the 75% drop in monitoring costs. Through a sensitivity analysis they even show that choosing between a uniform policy and a targeted policy is much more dependent on a comparison of the pollution abatement cost amount than on the transaction cost amount.

This brief overview points out that in order to control the transaction costs of site-specific policies, information costs (*i.e.* the cost of data research, collection and analysis) must first be controlled. The most important question is therefore to know whether increased transaction costs will be covered by the augmented precision of policy. Another way to approach this question, as explored by (Tietenberg 1974), consists in seeking policy modifications which lessen the information requirements considerably while maintaining as many desirable properties as possible.

4. Discussion and perspectives

The advantages of giving policies a spatial focus can be valorised, on condition that differentiating regulation areas is based on two types of heterogeneity: i) heterogeneity of the milieu by reducing pollution first in areas where it will be most effective, ii) heterogeneity of agents aiming at those with the lowest costs so that they bear the greatest reduction. This means identifying the most vulnerable milieus and the areas where agents can most economically modify their production technique. The prime question here is therefore: what is the most appropriate spatial scale to valorise these heterogeneities without generating exaggeratedly high information costs?

Let us first of all note that, whatever the scale adopted, there is a high amount of imprecision with respect to knowledge of the physical context. Indeed, even at plot level, the soil is never homogeneous or uniform[7] (Vauclin, 1983). Developing studies at local or regional scale therefore raises the issue of model uncertainty and encourages stochastic methods which provide results in the form of random variables. For example, taking into account the uncertainty of soil parameters (Sauboua 2001) leads to an increase in the range of variation in nitrate pollution and the analysis of variance shows that concentrations between different soil classes are not significantly different. This underlines the complexity of assessing the appropriate spatial scale for modelling and decision-making. Taking into account the

[7] which means that these properties are distributed in a multimodal fashion and that their distribution is not of the Dirac type but is spread out.

information available and, above all, its lack of precision, the best trade-off between inter-site variability and intra-site variability should be found.

To take spatial heterogeneity into account, the use of spatially focussed information systems (GIS) offers considerable and advantageous methodological potential. The creation of such tools has enabled many existing data[8] to be stored and summarised and highly detailed spatial analyses to be developed. Hence Opaluch and Segerson (1991) propose to use a GIS linking with a micro-parameter distribution model as this would allow the behaviour described at micro level to be aggregated at a meso-economic level. This methodological combination can help to simplify and select the information necessary: the micro-parametric model indicating the key variables of the process analysed (e.g. soil types), and the GIS to scan the scope of possibilities (information available, resolution grid, etc…). As long as there is a good interface between these two tools, the simplification achieved should not diminish the results obtained. Indeed, Tietenberg (1974) demonstrates that policies can be designed by significantly reducing the degree of detailed information without sacrificing all the properties of efficiency. However, in order to do so, policy differentiation must be based on the kind of spatial detail that is important for describing pollution.

In the case of nitrate pollution from agriculture, it is possible to specify the approach needed to design a site-specific policy. The bases of such a spatial approach already exist: the so-called “nitrate vulnerable zones” where the aquifer nitrate concentration is close to or exceeds the European standard, are areas in which pollution should be reduced first and where the effort should concentrate. However, this binary area definition is not sufficient to be used as the basis for an efficient water quality policy. The framework directive on water, which differentiates areas according to the objectives to be reached, resources to be implemented, etc… offers an ideal framework for such a policy since it will structure all territories of Europe.

After having defined the target sites, in terms of the effort to be made in each one, the population of farmers who will have to bear this effort should then be targeted. Of course, the amount of effort to be made will depend on the farming context, soil conditions and characteristics of water resource to be preserved. To minimise abatement costs, the effort must first and foremost come from agents bearing the least costs. Furthermore, to control the transaction costs, only a small population should be targeted. Our research has enabled us to distinguish three target populations by increasing order of effort and of costs:

- livestock farmers, who should be incited to reduce their nitrogen input. For this category of farmers abatement costs are weak or nil, and they may even make savings through removing technical inefficiency;
- crop farmers, who should be encouraged to reduce the duration of the intercropping interval by sowing nitrate catch crops;
- farmers operating in very unfavourable conditions, who should be encouraged to modify their production systems.

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There are therefore ways definitely available to implement spatially-targeted policies to reduce nitrate pollution and, beyond these, policies to reduce the negative environmental impacts of farming. There is nevertheless one recurring question: that of farmers’ agreeing to policies that would create new forms of differential rents. These new rents would be added on top of traditional problems relating to the unequal fertility of soil ...

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[8] It has also provided the incentive for large-scale data collection campaigns, such as the campaign that led to the drawing up of a European-wide soil map.

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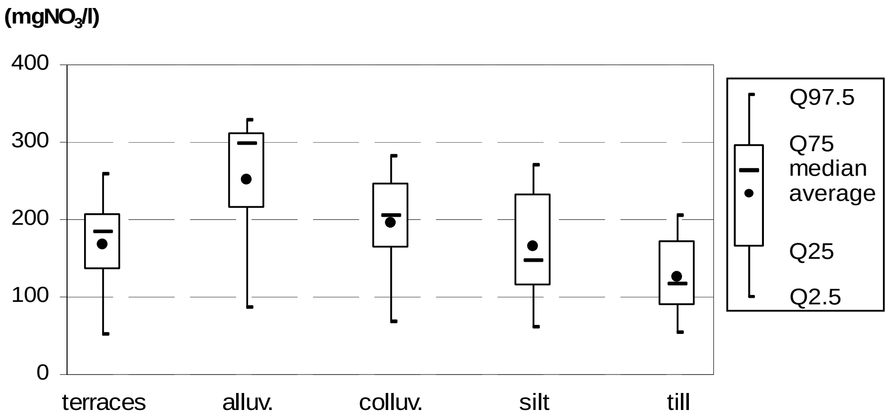
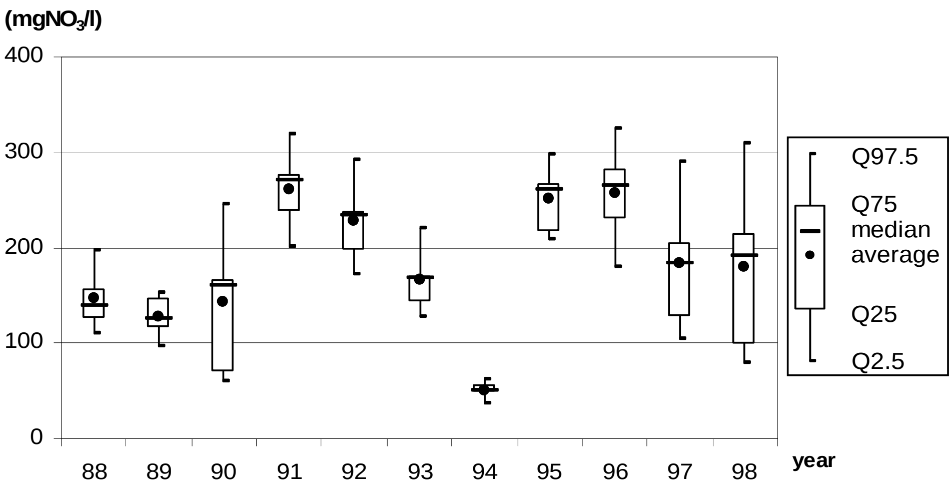
Figure 1. Nitrate concentration in water drained under non-irrigated maize according to soil class.



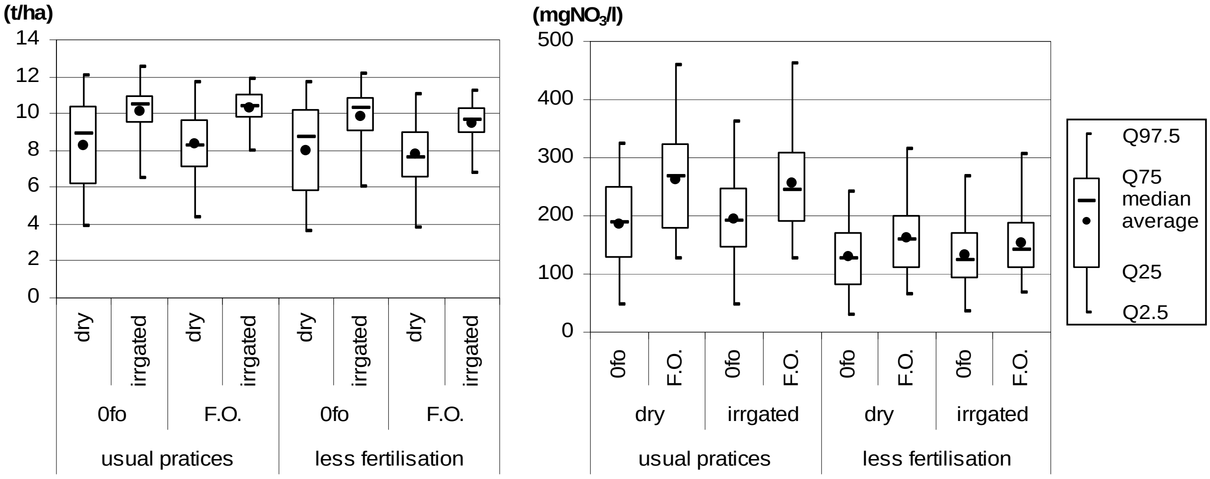
Note: these are the results obtained over 10 climatic years

Figure 2. Nitrate concentration in water drained under non-irrigated maize according to climatic conditions.



Note: these are the results obtained for 5 soil classes

Figure 3. Yield and nitrate concentration in water drained under non-irrigated maize according to different farming practice scenarios.



Note: these are the results obtained for 5 soil classes and over 10 climatic years

Table 1 . Expected results of farm practice scenarios in the medium term

	Conv	*Intfert*	*IntfertC1*	*IntfertC2*	*RedinpC1*	*RedinpC2*	*Setas*
$E\left(\overline{C}_i\right)$		6.5	26.2	29.8	33.6	37.2	153.0
$E(\overline{E}_i)$	77.0	70.3	55.2	44.8	50.8	41.0	58.7
$P\left[\overline{E}_i \leq 50mg.l^{-1}\right]$	0.00	0.00	0.00	1.00	0.23	1.00	0.00

$E\left(\overline{C}_i\right)$: Expected cost for each scenario calculated for the entire basin (€/ha)

$E(\overline{E}_i)$: Expected concentration (mg $NO_{3.}$ l^{-1})

$P\left[\overline{E}_i \leq 50mg.l^{-1}\right]$: Probability of achieving the European standard

Table 2. Maximum annual expected concentration (mg NO3. l-1) for the entire watershed and for different types of soil.

	Conv	*Intfert*	*IntfertC1*	*IntfertC2*	*RedinpC1*	*RedinpC2*	*Setas*
Entire watershed	109.6	103.3	72.9	58.3	67.1	55.1	66.5
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

Table 3. Results of various studies on the implementation of spatially-targeted policies

	Costs taken into account	Objective	Simulated policies	Heterogeneity taken into account	Results on costs
Braden *et al.* (1989)	abatement	Reduction of sediment pollution (erosion)	Modification of management practices (crop rotation, tillage, structural measures.)	Four sediment transport models representing differents level of abstraction	Significant difference in favour of practices spatially differentiated
Carpentier *et al.* (1998)	abatement + transaction	Reduce nitrogen runoff by 40%	BMPs to achieve an uniform performance standard v/v a targeted performance standard	237 Pennsylvanian dairy farms in specific soil and topography conditions	Costs 3 times as high for the uniform policy
Fleming and Adams (1997)	abatement	50 mg NO_3 per litre	Tax on nitrogen fertilizers (uniform, by area)	4 areas in Malheur County (one of these areas has a total of half of the total utilised agricultural area)	Little effect of spatial variance in physical parameters
Helfand and House (1995)	abatement	Reduce nitrate runoff by 20%	Uniform and differentiated input taxes (nitrogen fertilizers and irrigation water)	Two silty soils in California's Salinas Valley	Small difference in favour of differentiated taxes
Kampas and White (2004)	1) abatement 2) abatement + transaction	50 mg NO_3 per litre	Restriction on nitrogen fertilizers (uniform or by land class[9])	Three land classes in England	1) Small difference in favour of differentiated restrictions 2) no difference
Mapp *et al.* (1994)	abatement	Restriction on quantity nitrogen fertilizers applied (reduce by 33%)	Limitation on the total quantity, limitation on per-acre (on specific soil, on specific cropping system)	Five Central High Pslain regions in the USA	Significant differences in favour of spatial policies (specific soil, specific cropping system)
Moxey and White (1994)	abatement	Reduce of 10 to 40% of the nitrate concentration	Quota on nitrogen fertilizers (uniform or by land class[9])	Nine land classes in the Tyne basin (England)	Targeted quota advantage

[9] The land class classification is based on soil, topography and climate characteristics.