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Controlling Groundwater Pollution from Agricultural Nonpoint Sources: An Overview of Policy Instruments

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Abstract

The aim of this paper is to provide a review of policy instruments aimed at controlling pollution from agricultural diffuse sources, and compare their pros and cons. The review also includes a description of instruments introduced though recent reforms of the European Common Agricultural Policy (CAP), reforms aimed, *inter alia*, at integrating environmental protection into policies traditionally designed to achieve other objectives.

The major results of this review may be summarised as follows. Firstly, a major barrier to the implementation of effective policy measures is the lack of information about the nature, extent, and social costs of groundwater pollution from agricultural diffuse sources.

Secondly, policies aimed at controlling pollution from agricultural sources have usually relied, and still largely rely upon what is often referred to as "voluntarism", but which can probably be better described as a "soft-persuasion-though-subsidisation" approach. Besides being in contrast with the polluter pays ethics dominating other environmental policies, this approach has not brought about a significant and widespread reversal of pollution trends.

Finally, there is a need for clearer policy framework specifying the principle for a division of labour between CAP and environmental policy provisions, and between payments and regulation related to positive and negative externalities of agricultural production.

CONTROLLING GROUNDWATER POLLUTION FROM AGRICULTURAL NONPOINT SOURCES: AN OVERVIEW OF POLICY INSTRUMENTS

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1. Introduction

Agricultural incidental impacts upon groundwater quality can be either traced back to the use of potentially harmful inputs – namely, fertilisers and plant-protection products – or to other farming practices (irrigation techniques and groundwater abstractions). Aquifer enrichment takes place through potential pollutants accumulating on farmland (e.g. nitrogen surpluses) or coming from outside the farm-gates (saltwater in coastal areas) (Giacomelli *et al*., 2000).

Although the agricultural activities and the natural processes through which pollutants are generated and/or intruded into aquifers may be quite varied, they often share a number of general features. For instance, most of the agriculture-related groundwater pollution problems can be described as being *nonpoint source* (NPS) problems, in that they typically involve many geographically dispersed agents which cause intermittent low-pollution discharges which, in general, cannot be easily intercepted and neutralised through "end-of-pipe" structural devices. These features tend to make it difficult, or even impossible, to apply the battery of environmental policy measures traditionally employed to manage pollutant discharges from large and readily identifiable industrial and municipal point-sources.

According to a recent EC Commission communication on the state of Europe's environment, whilst there have been substantial improvements in surface water quality due to reductions in point source discharges such as emissions of phosphorous (–30/60% since the mid-1980's) and organic matter discharges (–50/80%), pollutant emissions from agricultural diffuse sources have shown little change since 1980, and EU maximum groundwater concentrations of nitrate and certain pesticides are frequently exceeded

(European Commission, 1999). In 1996, the Commission released the proposal entitled *An Action Programme for Integrated Groundwater Protection and Management*, where it is stressed that proper management of groundwater should be a key component of Member States' environmental policies, and, within the overall objective of groundwater conservation, "relieving the pressure from diffuse sources should have the highest priority" (European Commission, 1996).

There are various explanations for the still modest control of NPS pollution in general, and in particular, actions to regulate groundwater pollution from agricultural sources.

One explanation surely lies in the difficulties still faced by policy-makers in updating traditional pollution control strategies and regulatory approaches in order to address NPS problems. For instance, water pollution control has mostly relied upon *ex post* structural correctives (privately or collectively managed water treatment facilities), or *ex ante* regulatory measures which take observable individual emissions (mandatory effluent standards or, less frequently, environmental charges) as a reference point. In the case of agricultural NPS pollution, due to the high cost of monitoring individual pollutant discharges, transaction costs[i] associated with regulatory policies are particularly high. "These higher costs may be one of the reason why point sources … have been emphasised in water quality legislation" (McCann & Easter, 1999, p. 402).

Secondly, economic activities that are responsible for NPS pollution problems, agriculture in particular, have substantially been, and are still, although to a lesser extent, exonerated from mandatory regulation, or have not been confronted by effective economic incentives aimed at internalising the social costs (benefits) of pollution (abatement). On the contrary, rather than addressing market failures and promoting a more sustainable use of natural resources, agricultural policies have often added further distortions, and by so doing, have often worsened the misuse of resources.

The purpose of this paper is to provide a taxonomy and literature review of proposed policy instruments aimed at controlling pollution from agricultural diffuse sources, and compare their pros and cons. The review also includes a description of instruments introduced though recent reforms of the European Common Agricultural Policy (CAP), reforms aimed, *inter alia*, at integrating environmental protection into policies traditionally designed to achieve other objectives.[ii]

Groundwater pollution from agricultural nonpoint sources: key features and implications for policy design

2.1. NPS pollution: key features

The underlying characteristics of NPS pollution have been documented in studies dating from the late 1970s. These features can be summarised as follows.

Firstly, it is difficult to rely on structural devices for intercepting and neutralising polluting substances. For instance, while discharges of waste water from industrial plants or municipal point sources are generally easy to treat – for example, by installing filters in the pipes through which effluents are released in the environment – NPS effluents are difficult to intercept and neutralise because of the geographical dispersion of sources, and because pollutants may follow tortuous paths before reaching water bodies.

The second important feature of NPS pollution is the part played by the physical characteristics of the site where farmers operate, as well as the area through which pollutants move, in determining both the generation of potential pollutants (henceforth *on-site emissions*) and their ultimate environmental effects (*off-site discharges*). For instance, the same farming practice may have different impacts upon water quality depending on the characteristics of the farmland, climatic conditions, and the location of the farm in relation to potentially affected water bodies. This aspect is not exclusive to NPS pollution. Even for point sources, the firm's "type" (e.g. the relative efficiency of machinery and equipment) and location may affect on-site emission rates and their ultimate environmental impacts. However, there are two features which tend to characterise NPS problems (Dosi & Tomasi, 1994). The first is the sheer number and variety of sources (heterogeneity of farmland characteristics, hydrological and climatic conditions). The second is the role played by exogenous and partly unforeseeable events (such as weather conditions) towards the generation of potential pollutants and their *delivery ratio* (i.e. the ratio between off-site discharges and on-site emissions).

The third and probably most definitive feature of NPS pollution is the difficulty of monitoring individual pollutant discharges. While pollutants from point sources enter the environment at a specific, single location (such as a single pipe), NPS effluents (which often have a fairly low density per unit area) do not enter water bodies at a defined point, and are usually dispersed by natural processes. Inferring individual responsibilities from ambient pollutant concentrations is also difficult. While pollutants from point sources are usually delivered to water bodies more or less proportionally to on-site emissions, NPS pollutants may travel long distances and undergo a qualitative change before delivery.

The underlying features of NPS pollution that have been described have two main implications for policy design.

Firstly, the difficulty of relying upon end-of-pipe structural correctives makes a preventive approach (abatement of on-site emissions) preferable. It is sometimes the only viable option for controlling groundwater pollution from diffuse agricultural sources.

Secondly, because of the difficulty/impossibility of monitoring individual discharges, the effectiveness of regulatory measures aimed at preventing the generation of pollutant loads depends essentially on policy-makers' willingness and political ability to enforce alternative ways of establishing the causal link between farmers' activities and observable groundwater quality problems.

2.2. A taxonomy of NPS pollution control policy instruments

2.2.1. General classifications of environmental policies

Before focussing on the classification of specific measures developed to address NPS pollution problems, let us first briefly look at the more general taxonomies of environmental policy instruments.

A common classification is one that highlights the underlying difference between economic instruments, voluntary approaches, and mandatory regulations.

A standard definition of *economic instruments* can be found in OECD (1991;1997), where they are described as "instruments that affect costs and benefits of alternative actions open to economic agents, with effect of influencing behaviour in a way that is favourable to the environment" (OECD, 1991, p.10). These instruments typically involve either a financial transfer between polluters and the community (e.g. charges/taxes or subsidies) or the creation of new markets (e.g. tradable emission/pollution permits).

So-called *voluntary approaches* (VAs) are somewhat more elusive and difficult to define. A quite general and comprehensive definition of VAs is provided by Lévêque (1997), who describes them as commitments of polluting firms or sectors to improve their environmental performance. According to Brau & Carraro (1999), these commitments can be placed into three categories: (i) *unilateral commitments*, which consist of environmentally friendly adjustments established by firms themselves (e.g. a spontaneous switch to organic farming, either for "ideological reasons", or taking advantage of consumers' willingness-to-pay for "green" products); (ii) *public voluntary schemes*, in which participating firms agree to standards developed by public bodies (e.g. farmers' adhesion to the agri-environment schemes, introduced through Regulation 2078/92, described in section 3); (iii) *negotiated agreements*, i.e. specific contracts between public authorities, or other intermediate subjects, and polluting firms, e.g. agreements between water authorities, or water supply companies, and farmers operating within or near drinking water catchment areas.

Finally, policy provisions which do not make appeal to economic rationality or social responsibility, but involve a compulsory restriction of the polluters' choice domain, can be labelled *command-and-control* policies.

Besides the distinction between policies aimed at promoting self-regulation and mandatory regulations, a complementary criterion for classifying environmental policies is whether or not the *polluter pays principle* (PPP) applies. Broadly speaking, policy instruments are generally believed to be consistent with PPP if agents who use the environment either deliberately or incidentally as a sink for pollutants face a cost for the damage imposed on the rest of society.

However, labelling policy provisions according to their consistency with PPP becomes more difficult when compensation is foreseen for economic agents who voluntarily commit themselves to go beyond (overcomply) the minimum environmental standards set up by mandatory regulations. Whilst these payments may not, at first glance, appear to be consistent with the PPP ethics, they are claimed to be so in various official documents in which overcompliance is *ex lege* assimilated to the provision of "environmental services".

In this respect, it is worth noting that the European Commission has quite explicitly identified the "legal" borderline between negative environmental externalities – whose internalisation does not make farmers eligible for compensation – and the provision of environmental services which, on the contrary, should be remunerated by society:

> The underlying rationale of the Commission's proposals for integrating environmental concerns into agriculture rests on two principles:
>
> - firstly, farming, as any economic sector, should attain a basic standard of environmental care without specific payment. This should be contained within the scope of good farming practice (which includes many matters other than environment) and comprises observance of regulatory standards and an exercise of care which a reasonable farmer would employ. This basic standard is also referred to as the reference level;
> - secondly, wherever society asks farmers to provide an environmental service beyond the reference level, and the farmer incurs cost or income loss, society must expect to pay for the service. This standard is also known as the target level. (EU Commission, 1998, p.115)

2.2.2. A taxonomy of NPS pollution regulatory strategies: "indirect" and "direct" policy approaches

There is a large body of literature dealing with policy instruments aimed at controlling water pollution from agricultural diffuse sources through preventive measures. The proposed instruments can be classified according to the criteria illustrated in section 2.2.1. One classification is the way in which pollution control operates, i.e. through introducing compulsion to the farmers' choice domain or through affecting the pros and cons of alternative courses of action legally open to farmers. Another classification is the social distribution of the costs of pollution abatement (i.e. whether farmers are compensated for environmentally friendly adjustments).

However, when considering policies specifically addressing NPS pollution, it is useful to adopt an additional classification criterion based upon the way in which the *monitoring problems* that arise from the characteristics of NPS pollution are addressed. For instance, whilst most of the authors base their recommendations on the difficulty of monitoring individual impacts upon water quality, the proposed regulatory approaches vary across the economic literature.

Following the taxonomy proposed by Dosi & Moretto (1993), regulatory approaches can be classified according to the reference basis adopted for setting policy measures, namely *estimated individual pollutant-discharges* ("indirect regulatory approach") or *observable total discharges* ("direct regulatory approach").

As far as the *indirect approach* is concerned, from the seminal papers of Griffin and Bromley (1982) and Shortle & Dunn (1986) onwards, many authors have considered that estimated rather than observable individual impacts upon water quality should provide the point of reference for designing regulatory tools (economic instruments or mandatory regulations). Such estimates could be obtained by means of available, albeit imperfect, models of pollutant generation (and transport) which provide predictions of on-site emissions (off-site discharges) attributable to a single farm or to a specific set of farming practices (Giacomelli *et al*., 2000). For example, methods for calculating nitrogen surpluses (known as the farm-gate balances) have been developed in some Member States in order to highlight areas at risk of nitrogen pollution (European Commission, 1999).

In contrast, there are authors who have recommended *observed* concentrations of pollutants at particular water bodies (e.g. nitrate concentration in a confined aquifer) as an alternative to estimated individual impacts upon water quality, The rationale behind the *direct regulatory approach* is that by setting an incentive mechanism based on an observable variable (total off-site discharges) the regulator would induce certain unobservable actions (abatement of individual on-site emissions). Policy instruments consistent with such a regulatory approach typically take on the form of tax/subsidy schemes that, broadly speaking, depend on deviations between measured and desired ambient pollutant concentrations.

NPS pollution control: the indirect regulatory approach

3.1. Estimated emission charges and standards

3.1.1. Emission charges

Environmental charges may be considered as being a way of putting prices on the use of the assimilative capacity of the environment. In practice, they work either as emission charges or as product charges.

The former are charges on effluents, and the tax burden is calculated according to the quantity or quality of pollutant emissions; the latter are levied on products (raw materials, intermediate or final products) whose quantity consumed or produced is taken as a proxy of the ultimate environmental impacts of a specific economic activity. In general, emission charges are considered to be more efficient than product charges because they leave target agents the freedom to select more cost-effective strategies to reduce effluents.

Obviously, the environmental effectiveness of an effluent-charge system crucially depends on the regulator's technical and administrative ability to monitor target agents' emissions and to evaluate their ultimate environmental impacts (pollutant concentrations at the receptor water body). Because of the number and geographical dispersion of sources, the often intermittent nature of pollutant emissions, and the spatial variability of transfer coefficients between on-site emissions and pollutant concentrations at the receptor, it follows that a charge system based upon observable individual discharges is not, in general, a viable policy option for dealing with NPS pollution problems. Similar considerations apply to reward systems foreseeing polluters' subsidisation according to observable reductions of effluent-discharges.

However, as advocates of the indirect regulatory approach emphasise, the difficulty of monitoring individual emissions could be partly overcame if *estimated*, rather than observable individual pollutant discharges, form the basis of the system.

The decision to adopt individual estimated discharges as a reference basis for implementing economic instruments aimed at affecting polluters' behaviours has two main policy implications.

First, the model used to estimate individual environmental impacts has to be granted "regulatory legitimacy", i.e. it has to be defined as the legal basis for computing the tax burden (Dosi & Moretto, 1993).

Secondly, the adoption of an estimated emission-charge system requires that the regulator provide target agents with two types of schedules, one that relates the tax burden to the estimated effluents and one that relates estimated emissions to specific farming practices. As Shortle & Dunn (1986) stress, since the two schedules link the farmer's choice of farming practice or practices (e.g. application rates of nitrogen fertilisers) to the tax, a charge system based on estimated emissions is, in essence, equivalent to a *product charge* system (e.g. nitrogen fertiliser levies).

3.1.2. Emission standards

Emission standards do not represent a viable policy option for dealing with NPS pollution problems for the same reasons which make a charge system based upon observable individual discharges an impracticable regulatory strategy.

However, the difficulty/impossibility of implementing an effluent-based regulation could be overcome if estimated, rather than observable emissions, form the basis of the regulatory scheme. Again, this requires granting legal legitimacy to a predictive NPS pollution model, and implies that the evaluation of compliance (or illegal behaviour) with the standard will be based upon monitoring farming practices which may or may not involve discharges exceeding the legally imposed emission threshold.

Since a charge system taking estimated emissions as a reference basis is *de facto* equivalent of a product charge system, it follows that a regulatory scheme based upon emission standards is, in essence, equivalent to regulatory schemes based upon *technological standards* (e.g. restrictions on input use or mandatory codes of good agronomic practices).

3.2. *Input and output-oriented policy measures*.

In light of the above arguments, when individual pollutant discharges are not technically monitorable at a reasonable cost, *input/output oriented policy measures* (product charges, subsidisation of environmentally friendly production methods, or technological standards), i.e. policies aimed at discouraging, promoting or mandating specific farming practice or practices, can be considered a reasonable second-best regulatory approach.

Input/output oriented policy measures aimed at addressing pollution from agricultural sources exhibit a great deal of variety. They include input and output levies, mandatory restrictions on input use, codes of good agricultural practice, reforms of agricultural policies, contingent subsidies ("cross-compliance measures"), and compensation for abandonment of potentially polluting activities ("set-aside").

3.2.1. Product charges, mandatory restrictions on input use and application zones

In principle, *product charges* such as levies on specific potentially polluting inputs will induce farmers to adopt "precision technologies": i.e. in order to lower the tax burden, reducing input by using appropriate application methods to increase efficiency and plant uptake.[iii] The difficulty of collecting data on input use for individual farmers may lead to a charge system based on observed choice of technology (or choice of crop). However, to be environmentally effective and economically efficient, such a system requires fixed-proportion production and pollution "technologies" (Shah, Zilberman, & Chakravorty, 1993).

When output (y) and pollutant discharges (z) are produced though variable input (a) using one or several distinct technologies (i): i.e. $y = f(a, i)$ and $z = g(a, i)$, an indirect optimal control of pollution can be established by output taxes or through several rates of input taxes that vary according to technology i. If a farmer adopts a technology which has a higher input-use efficiency (e.g. lower nitrogen surpluses), s/he will be charged through a lower tax rate (on output and/or on inputs).

However, establishing such tax rates is difficult because of non-linearity and the need to collect data on output and input use according to farmers' technology. It is obviously simpler to design uniform output or input levies in the form of sales taxes, but such uniform tax rates are sub-optimal. When there is a significant technological heterogeneity, taxing output may be especially counterproductive for those cases where farmers have adopted precision technologies: they will have higher levels of outputs and will have to pay higher taxes.

In addition to taking technological heterogeneity into account, when environmental variability is of greater significance, product charges should also be spatially differentiated. In other words, as with technological heterogeneity, heterogeneity in environmental conditions makes such economic incentives lose much of their theoretical appeal.

Since product charges such as levies on fertilisers and pesticides can hardly be distinguished within the same market, mandatory regulation, which may be spatially differentiated, may be more appealing (Zeitouni, 1991; Goetz & Zilberman, 1995). Optimal spatially differentiated mandatory regulations may be deduced by taking into consideration the hydrological properties of groundwater resources, their directions and speed of flow, and aquifer accessibility. Although these considerations require profound knowledge of the local conditions, the data needed to feed "simple" models may be available, and they can provide guidance for identifying areas locally sensitive to pollution, and the relevant *application zones* (Goralic, Remson, & Cottele, 1979; Millon, 1987; Zeitouni, 1991).

When information about aquifer properties is not available to the regulator, the optimal level of applied polluting inputs (such as fertilisers or pesticides) could be deduced by applying the "safety-first" approach to risk management (Roy, 1952). In relation to this, Lichtenberg & Zilberman (1987) have suggested that the establishment of water pollution control policies should minimise the cost of attaining those environmental quality objectives which have a certain degree of statistical reliability. This requires that the probability of exceeding the quality target does not exceed a pre-specified level. From this optimisation, a shadow price for the risk can be calculated. This shadow price can be interpreted as the marginal cost for increased safety.

Braden *et al.* (1989) expanded this approach further in developing a regional land management and input choice model to reduce the cost of reaching a water quality target with a certain degree of reliability. Their analysis emphasised the importance of modifying farming practices in environmentally sensitive areas, either by direct control or by appropriate economic incentives.

Although it is somewhat different, specifying quality standards in terms of risk instead of in concentrations enables differentiability in standards according to the sensitivity of the area to which they are to be applied. The reason is that a certain pre-specified risk over a sensitive location may entail more restrictive regulation, while the same risk applied to less sensitive locations may indicate a less restrictive concentration of pollutants.

It should be noted however, that setting a uniform level of risk for all places may not minimise costs. Lichtenberg, Zilberman, & Bogen (1989) compared water quality standards for reducing the risk of DBCP in groundwater in California. One of the things they found was that regional risk targets could be met with reduced costs by setting lower standards for rural wells than for urban wells, since rural wells serve a smaller population.

3.2.2. Codes of good agricultural practice and vulnerable zones: The EC Nitrate Directive

Broadly speaking, the term "good agricultural practice" refers to farmland management and production methods able to prevent or to reduce environmental damage. In EC legislation, the term is more commonly applied to the regulation of nitrate pollution from diffuse sources, and in this context, it can be seen as being an application to agriculture of the concept of best environmental practice that is applied in industry.

In 1991, the EC Council adopted the Nitrate Directive (91/676/EEC). Its aim was to reduce water pollution by nitrates from agricultural sources and to prevent further pollution. According to the Directive, Member States (a) must establish codes of good agronomic practice to be implemented by farmers on a voluntary basis, and (b) must identify *vulnerable zones* within their territory and implement action programmes which should contain mandatory measures for agricultural practices. The Directive defines a vulnerable zone as an area where nitrate concentrations exceed, or are likely to exceed in the future, the maximum admissible concentration of 50 mg/l.

Member States should have: (a) implemented the Directive in their national legislations, established a code or codes of good agricultural practice, and designated vulnerable zones by December 1993, (b) introduced action programmes imposing compulsory restrictions of farming activity by December 1995.

In 1997, the first planned Commission report on the implementation of the Nitrate Directive was produced. The Commission noted that six years after its adoption, “the status of implementation in most Member States is unsatisfactory [and] the failure to implement the Directive fully, in addition to its legal aspects, constitutes a failure to deal with serious environmental and human health problems” (European Commission, 1997).

For instance, only four Member States met their implementation obligations by the set deadline (Denmark, Spain, France and Luxembourg). At the time the report was prepared, most Member States had yet to designate vulnerable zones (Belgium, Greece, Spain, Portugal and the United Kingdom). Action programmes, which should have started on December 1995, were notified to the Commission only by Germany, Luxembourg, Austria and Sweden, on June 1997.

3.2.3. Reforming agricultural policies: the EC 1992 agri-environment programme

Although pollution abatement generally requires the implementation of *ad hoc* environmental policies, in many instances polluters’ behaviours could be positively affected through *reforming existing sector policies* to remove distorting incentives, or to integrate environmental protection into policies traditionally designed to achieve other public objectives. This is especially true for the European agricultural sector and for the Common Agricultural Policy (CAP).

As noticed by Brouwer (2000), it is difficult to assess to what extent CAP has affected the course of agricultural development and, in particular, structural changes such as intensification, specialisation, and concentration which are commonly believed to be responsible for observed negative water quality trends. For instance, even in a complete *laissez-faire* scenario, European farmers would have “overused” the environment, because of the basic failure of market mechanisms to drive a socially efficient use of natural resources. However, one can legitimately claim that farming support policies, and in particular support though subsidisation of commodity prices, rather than promoting a more efficient use of the environment, have often added further distortions (Dosi & Ferro, 1990).

For instance, at the time when CAP’s objectives were drawn up, agricultural expansion (and expansion of production in general) was automatically accepted as being a desirable social goal, while environmental issues were considered to be extremely marginal. The heart of CAP was the system of guaranteed high prices for unlimited production which, by distorting output-input price relationships, has encouraged the intensification of agricultural activities and surpluses of farm products. Quotas on some products were introduced during the 1980s, but the purpose was to maintain guaranteed high prices, not to deliver, even indirectly, environmental benefits (European Environment Agency, 1995).

The legal requirement to integrate environmental protection into other EC policies was established in 1987 by the Single European Act (SEA) and was given a more comprehensive legal basis in the Maastricht Treaty. However, even before the SEA, the Commission acknowledged in various policy documents the need to update CAP in order to include environmental considerations. In particular, the 1985 *Green Paper Perspectives for the CAP,* stated explicitly that agriculture should be seen as being an economic sector which, like other sectors that are potentially damaging to the environment, should be subjected to restraints and controls in order to avoid environmental degradation, and that in general, the polluter pays principle should be applied (European Commission, 1985).

The need to inject substance into the general commitments made in the *Green Paper* and other policy papers[iv] partly influenced the 1992 *McSharry reform* package, the first comprehensive and substantial update of CAP since the Treaty of Rome. This package included three measures to accompany the principal CAP reform measures, namely: (a) the agri-environment programme (Regulation 2078/92); (b) the early retirement scheme (Regulation 2079/92), and (c) the forestry aid scheme (Regulation 2080/92).

To properly interpret the "environmental provisions" included in the McSharry reform, it is worth recalling the surrounding political, budgetary and economic context. As Baldock & Lowe (1996) emphasise, "it would be wrong … to see in this and subsequent policy initiatives the triumph of environmental interests …. Agricultural policy makers have responded to environmental concerns, not necessarily through any deep convictions, but because of the perceived coincidence between the aims of environmental improvement and the need to reduce agricultural output, thereby contributing to the alleviation of surplus and budgetary problems. [Moreover, especially in northern Europe] farming leaders, in a context of chronic oversupply of staple products and falling farm incomes, have begun to look to the provision by farmers of environmental 'products', in order to underpin or renew their claims for public support" (Baldock & Lowe, 1996, p.12-13).

As far as the potential contribution to water pollution abatement is concerned, the most important 1992 CAP reform accompanying measure is the *agri-enviroment programme* established through *Regulation 2078/92* [v] , which foresees compensations for farmers who undertake to reduce ("substantially") input use (namely "fertilisers or plant protection products"), to change to other more extensive crop patterns and more environmentally friendly production methods, or set aside farmland for at least 20 years with a view to protecting hydrological systems. While most of the Community farming support policies are not subject to additional funding by Member States, the agri-environment schemes are only partly financed though the European Agricultural Guidance and Guarantee Fund (EAGGF).

As recently reasserted in a Commission's report on the state of application of Regulation 2078,

> [the agri-environment programme] is not a regulatory one and only intervenes in the range of activities over which a farmer has discretion to act. Thus action to prevent illegal

pollution or to ensure that farmers observe minimum environmental standards in applying pesticides, should be the subject of regulation and codes of good agronomic practice. But not the aim of agri-environment measures. (European Commission, 1998, p.18).

Under Regulation 2078/92, the total expenditure by Member States for 1998 is estimated at ECU 1.73 billion, which represents about 4% of EAGGF which, in turn, accounts for about 50% of the entire Community budget. [vi]

About 20% of the total European Union's farmland (EU15) has been affected by Regulation 2078, with significant differences, however, within and between Member States. For instance, in southern Europe (Greece, Spain, Italy, and Portugal), the percentage of hectares covered is below, sometimes well below, the average (0.6%, 2.9%, 13.6% and 16,8%, respectively). In France the percentage is slightly above the average (22.9%) (European Commission, 1998).

As far as the effectiveness of Regulation 2078 is concerned, according to the previously mentioned Commission's report, there is evidence of:

- "highly positive results … for reduced input measures, especially organic farming, nature protection measures and maintenance of landscapes; some difficulties arose with extensification, set-aside for 20 years … resulting in low take up".
- "arable conversion to extensive grass shows improvement in landscape quality in one region, while not enough data exists on reduction of N-leaching".
- "positive results from erosion prevention measures … and N-leaching reduction measures, such as green-cover crops".
- "extensification of livestock measure has not been successful in several regions, one reason may be that the measures are not paid sufficiently".
- "application on highly profitable land is not satisfactory in the absence of sufficiently high premia. Greater use of targeting is generally suggested to ensure appropriateness of payments". (European Commission, 1998, pp.7-8).

In general terms, according to the Commission, the results of the first agri-environment programme have been quite positive, in that "at 4% of CAP Guarantee spending, [the substantial environmental benefits] represent good value for money" (European Commission, 1998, p.8).

However, besides the programme's "internal rate of return", the key issue is whether or not only reliance upon farmers' voluntary undertaking of subsidised environmental friendly adjustments can be considered as being substantial progress toward integrating environmental objectives into CAP. In this respect, it is legitimate to state that Regulation 2078 has not been a very effective engine for driving widespread and substantial groundwater quality improvements.

As forecast by some commentators immediately after the approval of the McSharry package, the agri-environment schemes have proved to be not attractive, and, consequently, they have not significantly affected the behaviours of those farmers for whom the cost of abandoning environmental unfriendly farming practices is relatively high (following the terminology employed in the Commission's report, farmers operating on "highly profitable land").

For instance, operating on highly profitable land does not necessarily mean that there is a higher pressure upon groundwater quality. However, when there is an overlap between farmland productivity and environmental sensitivity, reliance upon voluntarism and untargeted subsidisation is unlikely to be an environmentally effective (and efficient) policy provision. The shortcomings of a not properly targeted subsidisation of environmentally friendly farming adjustment are testified to by the very modest impacts of Regulation 2078 upon intensive agricultural systems: in a large number of European regions, there have been little changes in groundwater pollutant concentrations.

3.2.3. Reforming agricultural policies: Cross-compliance measures

Polluters can be induced to abandon certain practices or adopt certain conservation measures if this is set as a condition for eligibility for other public programs that they find attractive. When these programs foresee subsidisation of output prices – the traditional and still prevailing CAP support scheme – cross-compliance measures can be interpreted as an implicit form of *output charges*, in that failure to acquire eligibility implies a reduction of (a "levy" on) guaranteed prices.

To our knowledge, cross-compliance measures tied to environmental objectives were first introduced in the USA, as part of the Conservation Title of the 1985 reauthorization of the Food Security Act.

In the EC, they were only later formally considered as a policy option, and they were introduced though the recently agreed *Agenda 2000*, as a Member States' policy option.[vii] [viii] The European Commission's position on the proper use of cross-compliance measures is quite clearly stated in various working documents: "cross-compliance is most appropriate in ensuring adherence to the *reference level*" (European Commission, 1998, p.115)[ix], i.e. attaining a basic standard of environmental care, and not the provision of additional environmental services involving costs or income losses that should be paid by society (see section 2.2.1).

Generally speaking, the link between farming support (either in the form of price support or direct income support) and farmers' environmental performance can be implemented in different ways, ways that tend to exhibit a different degree of environmental effectiveness. Following the taxonomy proposed by Batie and Sappington (1986), two general approaches can be identified: (a) the *red ticket approach*, where eligibility for certain benefits (e.g. guaranteed prices) is made contingent upon the farmer attaining a given environmental standard or set of standards; (b) the *green ticket approach*, where farmers become eligible for higher levels of support if they comply with or exceed a given environmental standard.

It follows that the basic difference between the red and the green ticket approach is whether or not the benefits from existing farming support policy schemes are made contingent upon reduction of environmental damages, or whether pollution-abatement *per se* entitles farmers to get additional benefits with respect to the farming support "baseline". For instance, with a green ticket policy "a basic direct support is paid regardless of compliance with environmental standards and the additional support for complying or exceeding a given set of standards can be seen as a voluntary environmental scheme" (Christensen & Rygnestad, 1999, p.5).

Christensen and Rygnestad (1999) provide examples of red and green schemes with reference to Danish legislation, which has implemented the Agenda 2000 reform. Reductions in "hectare payments" and "headage payments" are foreseen for farmers who do not complete field plans or fertiliser plans, and for farmers who do not complete fertiliser accounts and over-fertilise, respectively (examples of "red ticket" schemes). Farmers operating in designated areas must comply with certain farming practices, including a reduction of fertiliser use, in order to receive subsidies only provided for environmentally friendly agricultural practices undertaken in environmentally sensitive areas.

In between the red and the green ticket schemes, is what is described by Baldock (1993) as the *orange ticket approach*, where eligibility for support payments is dependent upon farmer's willingness to enrol in an otherwise voluntary scheme which attracts *ad hoc* payments. An example of an orange ticket policy is the US Conservation Reserve Program. The Program, introduced as part of the Conservation Title in the 1985 Food Security Act, was designed to achieve multiple objectives, namely conservation of soil resources, reduction of surplus stocks of agricultural products, enhancement of wildlife habitat, and maintenance of farm income (Dosi, 1994). Farmers were allowed to be included in the program if at least one third of cultivated fields were classified as highly erodible land, and strong penalties were established for violation of CRP contracts. These include loss of access to price support programs, government crop insurance, loans, and, obviously, CRP payments.

Regardless of the "colour" of the cross-compliance provisions, their environmental effectiveness (in terms of pollution abatement) obviously depends, first of all, on the attractiveness of the host program (i.e. the program providing the benefits which would be lost if a farmer fails to acquire eligibility) as well as on the cost of acquiring eligibility requirements. Cross-compliance measures are obviously pointless if farmers perceive that the cost of complying with pollution abatement requirements is higher than the foreseen reduction of benefits stemming from the host program.

Moreover, the effectiveness of cross-compliance measures in terms of pollution abatement depends on the correlation between the economic characteristics of those farms which enrol in order to acquire eligibility, and the intrinsic environmental vulnerability of their sites of operation. *Ceteris paribus*, the higher the cost of being eligible faced by farmers operating in sensitive areas, the lower will be the environmental effectiveness of a cross-compliance measure.

In this respect, it is worth noting that a major potential problem with cross-compliance measures stems from the political difficulty of establishing mutual consistency between the original objectives pursued through the host program and NPS pollution control requirements. For example, if the legislator intended to support low-income farmers, it is probable that when the host program was designed, the beneficiaries were identified according to their economic status. However, to be effective (and efficient), cross-compliance measures addressing water pollution problems should be targeted according to those farm aspects that are environmentally relevant. It follows that cross-compliance provisions tied to environmental objectives may be difficult to reconcile with the host program's original objectives, and their political viability may be undermined by opposition from targeted agents: this opposition will be all the stronger the more they feel themselves deprived of the "right" to benefit from a program which other farmers with the same socio-economic status (e.g. acreage, or regional location) continue to benefit from (Dosi, 1994).

In May 1999, cross-compliance measures were introduced into CAP's instrument portfolio through *Regulation 1259/1999*.[x] According to Article 3:

> Member States shall take the environmental measures they consider to be appropriate in view of the situation of the agricultural land used or the production concerned and which reflect the potential environmental effects. These measures *may* include:
>
> - support in return for agri-environmental commitments,
> - general mandatory environmental requirements,
> - *specific environmental requirements constituting a condition for direct payments* [xi]

Looking at these measures in more detail, Article 4 specifies that:

> [Member States] may decide to reduce the amounts of payments which would … be granted to farmers in respect of a given calendar year where:
>
> - the labour force used on their holdings … falls short of limits to be determined by the Member States, and/or

- the overall prosperity of their holdings during that calendar year, expressed in the form of standard gross margin corresponding to the average situation of either a given region or a smaller geographic entity, rises above limits to be decided by Member States, and or
- the total amounts of payments granted under support schemes in respect of a calendar year exceed limits to be decided by Member States.

Finally, Article 5 establishes the principle that "Member States shall apply the measures referred to in Articles 3 and 4 in such a way as to ensure *equal treatment between farmers* and to avoid market and competition distortions ".[xii]

The principle of equal treatment, although politically appealing, does not however appear *prima facie* consistent with the need to take environmental heterogeneity into account when designing policy measures aimed at addressing NPS pollution problems. For instance, as already stressed, when environmentally-oriented cross-compliance measures are designed, farm and farmland characteristics and location differentials (which affect on-site emissions and pollutant delivery-rates) should be taken into account rather than farmers' socio-economic status. The achievement of other policy objectives, such as income support, should be pursued through other instruments, such as lump-sum transfers.

3.2.4. Land retirement (set-aside)

Land retirement, otherwise known as set-aside, is one of the options available for reducing agricultural harmful impacts upon groundwaters.

As Ribaudo and Osrorn (1994) emphasise, one of the main justifications for a *properly targeted* land retirement program is that the characteristics of NPS pollution and shortcomings in our ability to link on-field practices to environmental conditions make a practice-oriented approach very costly from an administrative standpoint. For instance, "while land retirement may be seen as unduly restrictive in that [social costs stemming from polluting activities] could be internalised and the land still remain in production, much lower administrative costs may justify its use when the merits of keeping the land in production are marginal" (Ribaudo & Orsorn, p.85).

In both the United States (the Acreage Reduction Program, ARP) and in the European Community (EC Regulation 1094/1988)[xiii], set-aside was initially introduced as a supply control policy instrument. However, with the 1985 U.S. Conservation Reserve Program (CRP) (and later on, with EC Regulation 2078/19920) the objectives of set-aside were broadened, in that land retirement was seen also as an environmental policy instrument.

As far as the US experience is concerned, in the initial years of implementation, the CRP enrolled somewhat less acreage, and at somewhat higher cost, than originally planned (Rodgers *et al*., 1990). One predictable source of ineffectiveness and inefficiency was the attempt to achieve too many objectives through a single instrument (set-aside): soil conservation, supply control, and budget discipline. This and the fact that CRP competed with the more supply-oriented ARP, have led to inefficiency in both programs. In an attempt to partly overcome these problems, Taff and Runge (1987) suggested a refinement in the eligibility criteria for both land set-aside programs: farmland highly productive but not environmentally sensitive should have been targeted through ARP, whilst farmland less productive but environmentally sensitive should have been targeted through CRP. Non-productive and non-environmentally sensitive farmland should have been made ineligible for both programs. Other authors recommended adding additional targeting criteria, including a "NPS index" (Pearce, 1987), in order to take into account not only the gross erosion potential of farmland, but also geographical position in relation to potentially affected water bodies.

An additional criticism of the 1985 CRP was the lack of formal links between conservation provisions and the provisions of the federal Clean Water Quality Act (in particular, section 319, dealing with NPS pollution problems). Kuck *et al*. (1990), in particular, suggested that the Farm Bill's conservation provisions should be targeted to watersheds that the States, as required by the Clean Water Act, had identified as not achieving federal water quality standards because of residual NPS pollution from agricultural sources.

The 1990 Reauthorization of the US Farm Bill partly accounted for these suggestions and criticisms by expanding the definition of eligible land to include areas subject not only to severe soil erosion, but also groundwater pollution.

The US experience (and something similar could be said for the EC experience), clearly shows that the effectiveness of the set-aside program, and, in particular programs tied to NPS water pollution abatement, crucially depends on the regulator's ability to differentiate between eligible and non-eligible farmers based on farmland characteristics and location relative to sensitive water bodies.

Moreover, as with the cross-compliance provisions, the environmental effectiveness of set-aside programs depends on the degree of correlation between farmland productivity and farmland environmental sensitivity, as well as on the foreseen compensation for land retirement. In this respect, as Ribaudo and Osrorn (1994) have emphasised, to be cost effective, set-aside programs should mainly focus on "marginal cropland and/or cropland that discharges into particularly valuable water resources" (p.85).

NPS pollution control: the direct regulatory approach

4.1. Ambient tax/subsidy policy schemes

Many authors have recommended the implementation of incentive mechanisms based on *observable ambient pollution levels* (groundwater pollutant concentrations) as an alternative to policies taking as a reference basis specific farming practices ("input/output oriented policy instruments). The main rationale underlying this regulatory approach is that, similarly to individual effluent discharges, monitoring farming practices may be administratively difficult, or prohibitively expensive.

There is a key difference between policies which take specific farming practices as a reference basis (indirect regulatory approach) and policies based upon observable ambient pollution levels (direct regulatory approach). While "with policies that hinge only on firm-specific decisions …, once the policy has been set, each firm does not have to consider its own pollution types or the types/actions of other firms since its own profits are independent of those types/action …, with policies based on ambient pollution …, each firm's profits will depend on ambient pollution, which is in turn a function not only of its own type/actions but also of the types/actions of other firms." (Tomasi *et al.*, 1994, p.10).

The prototype of NPS pollution regulatory schemes based on observable ambient pollution levels is the tax/subsidy scheme proposed by Segerson (1988), where every farmer who is presumed to have contributed towards water quality impairment (or improvement) should be charged (or rewarded) according to the deviations between the measured and the desired ambient pollutant concentrations.

According to Segerson (1990), a main advantage of this approach is that it allows the desired water quality goal to be achieved in a cost-effective manner. Those farmers for whom changes in management practices would have little effect on water quality will not seek to alter their farming practices, whereas those farmers whose behaviour substantially affects water quality would be induced to take steps to reduce pollution. Moreover, those polluters for whom changes would be effective would have greatly flexibility to reduce pollution using techniques that are the least costly ones for their specific site characteristics. Finally, although it requires monitoring of water quality, a tax on ambient concentrations does not require either individual pollutant-discharges or farming practices to be monitored.

As Bystrom and Bromley (1996) stress, the implementation of Segerson's environmental charge system is equivalent to forming associations within particular watersheds and making the group of farmers collectively responsible for water quality. For instance, if ambient pollution fees are levied, they are assessed against the collective as a group. This then forces the members of the group to monitor each other's behaviour, and to assess miscreants accordingly.

Despite the theoretical advantages, the direct regulatory approach suffers from several potential drawbacks.

Firstly, under this regulatory approach, taxes (or rewards) would be paid (or received) by every farmer, irrespective of his/her individual impact upon water quality. In other words, individual tax payments (rewards) will depend not only on her/his behaviour, but also on the behaviour of other polluters. This may raise legal or equity issues which could undermine the political viability of this incentive scheme. In particular, "ambient taxes can … be difficult to accept for agents who have already lowered their emissions in the past and now have to pay charges for common emissions" (Millock *et al*., 1997, p.5)

Secondly, this regulatory approach underlines the assumption that farmers possess adequate information about the nature and extent of their on-site emissions; moreover, farmers are assumed to be able to make correct evaluations about the ultimate impacts of their on-site emissions upon water quality. Both assumptions are somewhat questionable and, in any case, their feasibility should be assessed for each specific situation.

As far as the first assumption is concerned, farmers may not possess private (better) information regarding their on-site emissions, i.e. about the amount of pollutants originating on farmland which potentially may affect particular water bodies. This is especially true when potential pollutants begin as residuals of productive processes (e.g. nitrogen in excess of a crop's uptake). Consequently, they may be unable to properly identify the most effective (and efficient) management practices to reduce the generation of potential water pollutants.

As for the second assumption, farmers may be unable to predict with sufficient precision the cause and effect relationship between their management practices (their presumed on-site emissions) and the concentrations of pollutants observed in particular water bodies. In this respect, the direct regulatory approach appears to be potentially more appealing for relatively small watersheds, where few potential polluters operate, and for water bodies that do not rapidly flush out pollutants. It is however, much less suitable for large watersheds, where many farmers operate (sometimes together with other economic sectors), undertaking different activities (crop patterns, livestock and crop production, etc.) which may involve the generation of pollutants with complex transport paths.

4.2. Investment in monitoring equipment.

As has already been emphasised, the major drawback of a regulatory scheme based on collective is that individual penalties (or rewards, if observed ambient concentrations do not exceed the desired water quality standards) will not only depend on individual behaviour, which cannot be monitored, but also on the behaviour of other farmers.

In an attempt to partly overcome this problem, Xepapadeas (1994) explored an alternative policy approach. He provides what to our knowledge is the first model that attempts to endogenise monitoring of individual emissions. In the literature on NPS pollution regulation, this is generally assumed to be either technically impossible, or prohibitively expensive.

Xepapadeas (1994), however, considers the case where monitoring individual emissions is technically possible, so that information about individual effluents could, at least in principle, be improved by investing in "monitoring equipment". Using a theoretical model, he explores the potential advantages (in terms of regulatory efficiency) of a policy scheme comprising of (a) an emission tax (based upon the observable part of individual emissions); (b) an ambient-tax *à la* Segerson; and (c) an investment policy (undertaken by the regulator) on monitoring equipment. Xepapadeas shows that under certain circumstances, the proposed "policy package" generate regulatory benefits. In particular, the increased observability of individual emissions lowers the ambient tax component, which is the component that is most likely to generate strong political opposition in the regulatory package.

The idea of endogenising monitoring of individual emissions (rather than assuming than monitoring NPS discharge is either impossible or nearly always so) has been explored further by Millock *et al.* (1997). The main difference with Xepapadeas' model is that investment in monitoring equipment is not carried out by the regulator, but polluters themselves are induced, though appropriate incentives, to invest in monitoring in order to signal their true environmental performance.

Millock *et al.* (1997) show that the proposed "monitoring incentive scheme" aimed at inducing agents to exhibit their true characteristics, can be implemented even if the regulator is unable to monitor polluting input use on each farm, as long as the polluters can be identified. According to the proponents, the main advantage of the incentive scheme – which tends to transform part of the NPS pollution problem into a point source one – is that it would significantly reduce the regulator's information requirements.

5. Final remarks.

Groundwater pollution has been identified as one of the major environmental threats faced by European countries.

Even though other sectors may be the ultimate cause of pollution problems, the role of agriculture is not in doubt. Nonetheless, agricultural pollution is still far from being effectively addressed. This is partly attributable to the intrinsic difficulty of managing groundwater pollution from diffuse sources. But it is also attributable to the special status which has been granted to farmers who, generally speaking, have been exempted from credible mandatory regulations, and have not been confronted by economic incentives able to effectively influence their behavioural options.

Since agricultural impacts upon groundwater quality generally occur outside the borders of farms, and affect other individuals, policies which appeal to self-interest are useless. Similarly, appeal to social responsibility has rarely been an effective engine for substantial changes in polluters' behaviours. It follows that if, as asserted in various policy documents, NPS groundwater pollution abatement is one of the key European environmental objectives, major changes in policy styles and regulatory approaches towards the agricultural sector are required.

On the grounds on the available literature and the results of EC policies implemented during the last decade, the recommendable changes may be summarised as follows:

(a) Public awareness and NPS pollution control legitimation

A major barrier to the implementation of effective policy measures is the lack of information about the nature, extent, and social costs of groundwater pollution from agriculturally diffuse sources. Public consciousness of agricultural impacts upon groundwater quality has to be raised to the same level as for pollution problems such as surface waters impairment due to effluent discharges from large and readily identifiable point sources. As long as those suffering from pollution are unaware of the short/long term costs of groundwater contamination, there will inevitably be a political bias in favour of polluters who oppose effective policy measures.

Institutional or legal barriers to the implementation of these measures often stem from what is probably the most definitive feature of groundwater pollution from agricultural diffuse sources, that is, the difficulty of identifying individual responsibilities. Since it is technically difficult or prohibitively expensive to acquire full information about individual discharges, alternative ways for identifying farmers' responsibilities, and for sanctioning, or discouraging environmentally harmful behaviours through appropriate incentives, have to be politically legitimated.

Policy options do exist for addressing the problem. One alternative would consist in granting regulatory legitimacy to available NPS bio-physical predictive models. This would enable estimation of individual discharges so as to acquire a reference basis for mandatory regulations or economic instruments. An alternative approach envisaged by the economic literature would be to identify, with the aid of watershed-based models, the group of farmers who are presumed to be contributing to groundwater contamination, and then, through a "bubble" policy targeted to observable total pollutant concentrations, making the group collectively responsible for groundwater quality.

Although in principle appealing, both of the regulatory approaches may prove to be difficult to implement. In particular, as far as the former approach is concerned, in order to be effective and efficient, economic incentives aimed at penalising or rewarding specific farming practices should be spatially differentiated so that the heterogeneity of environmental conditions can be taken into account. However, a properly targeted geographical differentiation of economic incentives may be administratively difficult or legally impossible, which explains why instruments such as input levies tend to lose much of their appeal when used in relation to NPS pollution problems, while spatial differentiation of mandatory regulations (e.g. compulsory implementation of codes of good agronomic practice in vulnerable zones) may prove to be easier to implement and more appealing.

As far as the "bubble" policy approach is concerned, its major potential drawback is that individual penalties (or rewards, if observed pollutant concentrations do not exceed a pre-identified threshold) would be paid (or received) by every farmer operating within a watershed, irrespective of his/her individual impact upon groundwater quality. This obviously raises legal or equity issues which could undermine the political viability of this incentive scheme. However, this drawback could be attenuated by combining a "bubble" policy with economic incentives based upon the available information on individual emissions, information which could be acquired through predictive models or through investments in monitoring devices.

(b) The polluter pays principle and the borderline between environmental services and the prevention of environmental damages

Policies aimed at controlling pollution from agricultural sources have usually relied, and still largely rely upon what is often referred to as "voluntarism", but which can probably be better described as a "soft-persuasion-though-subsidisation" approach. Besides being in contrast with the polluter pays ethics dominating other environmental policies, this approach has not brought about a significant and widespread reversal of pollution trends.

This ineffectiveness is at least partly attributable to the somewhat ambiguous distinction between farmers' environmental services and environmental damages, a distinction which should provide the legal basis for deciding whether or not farmers are eligible for compensation for farming adjustments.

In various EEC policy documents, environmental services (*target levels*, according to the Commission's terminology) are defined as the outcome of any environmentally friendly adjustment of farming which goes beyond the basic standards of environmental care (*reference levels*). Is this politically defined borderline between farmers' positive and negative environmental externalities "equitable" and consistent with the polluter pays principle? The answer obviously depends on which agricultural practices are interpreted as being part of the collection of farmers' rights.

Traditionally, the European Common Agricultural Policy has relied upon two implicit assumptions: (i) since agricultural production *per se* provides social benefits which exceed consumers' willingness to pay for agricultural products, farmers are entitled to be rewarded through subsidised output prices, and (ii) since farmers' endowments include the right to use their land as they want, any environmentally friendly adjustment of farming requires additional subsidisation.

More recently, budgetary constraints, international trade disputes, and the increasingly popular idea that the environment belongs to society as a whole, have induced European policy-makers to slightly revise the traditional agricultural policy armoury by partly abandoning the system of guaranteed high prices for unlimited production, and by looking at alternatives for the rationale behind farmer subsidisation.

The previously-mentioned Commission distinction between “target” and “reference levels” may be seen as being one of the outcomes of this process, in that it reflects a partial reassignment of property rights between farmers and the rest of the society. According to this reassignment, farmers do not hold the right to use their land as they want, but hold the right to be rewarded for any adjustment of farming which goes beyond basic standards of environmental care.

As with any other politically constructed property rights systems, the system envisaged by the Commission is obviously questionable. What matters, however, is that to be credible and operative, this system requires a rigorous and unambiguous definition of the “reference level” in order to assess farmers’ compliance with legal regulations, and to have a benchmark for identifying farmers’ environmental services to be compensated by society.

However, in the EU in general, and particularly in those Member States which have not properly identified and credibly imposed basic standards of environmental care (e.g. failure to implement the Nitrate Directive), the in some ways intrinsically ambiguous distinction between farmers’ negative and positive environmental externalities has reinforced the attitude among farmers that they should wait for compensation for any environmentally friendly adjustment of farming. Such a consolidated attitude is likely to make the implementation of cross-compliance measures, which, in principle, could potentially partly bridge the gap between farmers’ subsidisation and farmers’ environmental performances, politically difficult.

(c) Integration between agricultural and environmental policies

The need to include environmental protection into CAP has been acknowledged by European authorities, and has to some extent influenced recent reforms. The “agri-environmental component” of CAP, introduced through the 1992 MacSharry Reform, is destined to expand under the recently approved Agenda 2000 policy package.

However, there is a clear need for better and more effective integration and coordination between agricultural policy, water resources management and environmental policy provisions.

Integrating groundwater protection objectives into CAP will, in practice, involve an ability to match agricultural policy more closely to environmental conditions by taking into account location differentials, and by tailoring policy provisions to the impact upon groundwater of alternative farming practices, rather than to the socio-economic status of farmers.

Rather than coming up with new Europe-wide specific measures, what is needed is a clearer European framework specifying the principle for a "division of labour" between CAP and environmental policy provisions, and between payments and regulation related to positive and negative externalities of agricultural production. Moreover, as Brouwer (2000) makes clear, in order to arrive at a concrete formulation of the groundwater conservation conditions that have to be fulfilled, a clear and unambiguous definition of the term "good agriculture practice" is essential. Codes of good agronomic practice, properly defined by taking into account the heterogeneity of environmental conditions, could, *inter alia*, become benchmarks for deciding whether a farmer is or is not eligible for public support, e.g. in the context of cross-compliance recently included in the CAP instrument portfolio.

Notes

[i] Following McCann and Easter (1999, p.404), transaction costs include: research, information gathering and analysis; enactment of enabling legislation including lobbying costs; design and implementation of the policy; support and administration of on-going program; monitoring/detection; and prosecution/inducement costs.

[ii] For an overview of the potential impacts of agricultural policies on farmers' "direct" or "incidental" water use, see Dosi (2000), Part III (Agricultural Policy and Water Use).

[iii] A special input that should be considered somewhat separately is water. Although water in itself is not a groundwater polluting input like nitrogen fertilisers and pesticides, its "overuse" may directly or indirectly contribute to aquifer depletion. As Giacomelli *et al.* (2000) point out, the potential contribution of irrigation to groundwater contamination is twofold. Firstly, aquifers are vulnerable to over-abstraction which may increase concentration of pollutants already introduced to the aquifer, or seawater intrusion in coastal aquifers. Secondly, deep-percolating water from irrigation contributes to aquifer enrichment by pollutants accumulated on farmland.

[iv] The *Future of Rural Society* (European Commission, 1988a) and *Environment and Agriculture* (European Commission, 1988b).

[v] The roots can be very broadly identified in earlier measures such as voluntary set-aside, experimental extensification and most significantly, in Article 19 of Council Regulation 797/85, which authorised Member Countries to introduce special national schemes in sensitive areas to subsidise environmentally friendly farming adjustments.

[vi] Total expenditure (1993-1998) for the implementation of Regulation 2078/92 is expected to reach nearly ECU 5.5 billion (European Commission, 1998).

[vii] To our knowledge, the first official EC document where cross-compliance was considered as a policy option to address agricultural related water pollution problems is the already mentioned Commission's proposal *An Action Programme for Integrated Groundwater Protection and Management*, where it is stated that: "all possibilities, including use of economic instruments in order to reduce use of manure and chemical fertilisers to the amount required for crop production and compatible with protection of the environment and fresh water quality should be explored …. The development of codes of good agricultural practice … should be at the centre of action taken. …. As compliance with the codes in itself may not be sufficient to achieve the objectives in certain regions, measures of a further-going nature to ensure environmentally compatible production could be developed. Possibilities for using the principle of *cross-compliance* should be explored in this context" (European Commission, 1996; Action Line 3.2).

[viii] A cross-compliance measure was introduced at the Community level through Regulation 1765/1992, forming part of the McSharry reform package (see section 3.2.4, footnote 5). However, this cross-compliance measure was not targeted to environmental goals, but to the reduction of production of surplus crops.

[ix] Italics added by the authors.

[x] The Regulation applies to payments granted directly to farmers under support schemes financed in full or in part by the "Guarantee" section of the EAGGF, except those provided for under EC Regulation 1257/1999.

[xi] Italics added by the authors.

[xii] Italics added by the authors.

[xiii] The first EC set-aside program provided for subsidies to any farmer committed himself to retiring the whole or part of his land from crop production for at least five years. The main objective was to reduce production of surplus crops; despite certain claims on this subject, the program did not have (and was not designed to achieve) environmental goals. Through Regulation 1765/1992, forming part of the McSharry reform package, another set-aside program was introduced with the aim of reducing production of surplus crops; the main difference with respect to the 1988 program lying in the fact that apart from certain categories of farmer, retirement of a certain amount of cropland was compulsory, or, more precisely, non-compliant farmers were made ineligible for price support programs. In this respect, the 1992 land retirement provisions can be interpreted as the first cross-compliance measure adopted at the Community level, although the measure was not targeted to environmental goals. A voluntary environmentally oriented set-aside program was included in the "agri-environmental program" established through Regulation 2078/1992.

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