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Environmental Pressures and Regulation in European Agriculture: A Survey of Current Issues and Policies

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for Project: GENEDEC, No 502184
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"A quantitative and qualitative assessment of the socio-economic and environmental impacts of decoupling of direct payments on agricultural production, markets and land use in the EU."

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Preface

This survey is financed by European Commission, under the activity "Scientific Support to Policies" of the 6th *Research Framework Programme*, for the project GENEDEC: "A quantitative and qualitative assessment of the socio-economic and environmental impacts of decoupling of direct payments on agricultural production, markets and land use in the EU", No. 502184.[1] The purpose of the survey is to provide a review of current issues and policy schemes designed to deal with agricultural pollution, with special focus on the EU agriculture.

As it is well known agriculture is closely associated with the environment mainly via the production of food and fibre and the habitation of the countryside. Furthermore, agriculture is a decisive factor for maintaining the viability and diversity of rural communities, landscape and habits, for facilitating the provision of tourism, recreational facilities and environmental protection (OECD, 1993). However, despite the potential beneficial environmental services European agriculture has been regarded as contributing to a number of environmental problems.

Economic theory teaches us that, due to well known externalities, in the absence of policy interventions unregulated markets fails to induce farmers to operate in the socially optimal way, and regulatory interventions are required for the achievement of Pareto optimality. Given the non-point-source (NPS) character of agricultural pollution standard instruments of environmental policy such as Pigouvian taxes and tradeable permits can not be easily employed, since individual discharges are unobservable and typically stochastic. In this context NPS pollution control has focused on other elements that may be observable such as polluter's choices (input-based schemes) and the consequences of polluters' actions (ambient-based schemes), as well as, policy schemes based on a new style of interaction between the regulator and polluters, the so-called Voluntary Approaches.

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[1] Opinions expressed in this study do not necessarily reflect the views of the European Commission

This survey presents major pollution problems associated with the EU agriculture and current policies adopted especially in the context of Pillar II, as well as the policy instruments developed in the Environmental Economics literature for dealing with these problems. In particular:

Chapter 2 describes the relation between the European agriculture and environment, with a focus on the relation between Pillar II, environmental concerns and environmental regulation. The adverse impacts of agriculture on the environment are described with emphasis given to the problem of nitrate leaching (the removal of nitrate from the soil by the action of water). The mechanism behind nitrate leaching and the factors that directly and indirectly contribute in leaching are presented.

Chapter 3 focuses on the nature of agricultural problems as Non-Point-Source pollution problems. After defining the difference between Point-Source and Non-Point-Source pollution problems, the features that classify agricultural nitrate leaching as a NPS pollution problem are described in detail. Finally, given the special features of agricultural problems, the set of feasible NPS policy schemes is presented.

Chapter 4 develops the NPS pollution model both under a static and dynamic context, since the generation of pollution has flow characteristics but its impact is related with accumulated pollution stocks, as well as under asymmetric information. A series of alternative agricultural pollution models are presented to provide insights into the complex nature of the problem and the approaches for regulating agricultural pollution. Market failures are identified in order to justify the introduction of environmental policies.

Chapter 5 provides a thorough description of available policy schemes for environmental regulation in the area of NPS pollution, depending on the available information set. Emission-based, input-based and ambient-based schemes, as well as mix-based schemes are described in the context of certainty and uncertainty, as well as in a static and dynamic framework. Within each category the alternative forms of the available instruments are provided and the requirements for their implementation as well as drawbacks are discussed.

Chapter 6 is solely dedicated to voluntary approaches (VAs), as an alternative tool to regulate NPS pollution problems. The major features of voluntary approaches such as, typology, differentiation criteria, motivation, as well as the characteristics of economic agents who appear the most likely to initiate or participate in VAs are thoroughly described. The advantages and drawbacks of VAs, as well as the existing implementation difficulties are also presented. The effectiveness of VAs is assessed and some rules about their effective use are also presented. Finally, the approaches of modeling voluntary agreements from the agriculture's point of view, are presented both in static and dynamic context. Finally, Chapter 7 concludes.

2

Executive Summary

This report was submitted to the coordinator on the 30th of August 2005 and circulated among partners for final comments and modifications to be made by December 2005, as suggested by the coordinator, in view of the additional 3 months granted by the Commission at the time that the contract was signed.

When the project was conceived, the European Commission had already admitted that, among the factors responsible for a series of adverse environmental services of agricultural activities, the various CAP measures could be included (Baldock D. et al, 2002). Such a formal recognition of the problem stressed the need for policy makers to account for both environmental issues and the recent developments in environmental regulatory policy in the design of the common agricultural policy.

In this context lies the contribution of Partner 6 within Work Package 3 under the Project GENEDEC. **Work Package 3** (WP3) is entitled: *New development and linkage between farm models and partial market model and environment* and has set the following objectives (see, Technical Annex I, pg 17):

Objective 1: Develop a set of models that would describe the interactions between EU and world markets (the "small country" assumption is irrelevant for the EU).

Objective 2: Improve the assessment provided by farm-type models (in WP2) by sharpening land opportunity cost thanks to land market modelling.

Objective 3: Examine the effects of decoupling on structural change in farming (e.g. farms number and size, full or part-time farming, entry-exit).

Objective 4: Determine the relationship between EU agricultural production and its physical environment.

Objective 5: Study Non point source pollution problems, which are typical environmetal problems associated with agricultural activities.

The work to be undertaken under the WP3 can be described as follows.

- To fulfil the first objective a set of models that would describe the interaction between EU markets and the world will be developed. Then EU prices will be adjusted to world prices using a simplified FAPRI-like model, i.e. AgMeMod.

  Land market will be modelled under condition of non-tradable premium rights using shadow prices from WP2.

  Series of historical transition probabilities will be developed and a non-stationary Markov Chain model, which could be coupled to models reviewed in WP2, will be implemented. It may be necessary to supplement the Markov Chain model with some farmer-decision models to examine issues such as succession and labour allocation.

  The coupling of economic and biophysical models will be developed to get an improved technical approach, to spatialize land use and production localisation. Theoretical approach related to the implementation of environmental policies will be tackled through the transformation of a non point source pollution problem into a point source one.

Work Package 3 will provide two **deliverables**:

D4: List of prices and land opportunity costs, projections concerning number of farms, proportion of full and part-time farming on a member state basis, technical indicators, extensification indicator like the average productivity of land, agro-technical and cross reference data bases and maps.

D5: Report or article on theoretical aspects concerning the implementation of environmental policies.

Regarding the **milestones and expected results** the technical Annex I (pg 17) involves:
Prices adjusted to world market and land opportunity costs are meant to implement exogenous prices in farm-type models.

Objectives 3 and 4 will provide necessary materials to feed WP4 and WP5.

The contractual obligation of Partner 6 under the Work Package 3 consists of objective 5 (see Technical Annex I, pg 20) that results in Deliverable 5 (D5): *Report or article on theoretical aspects concerning the implementation of environmental policies*, consisting of two parts:

D5.1: Conducting of an extensive survey of current issues and policies regarding environmental pressures and regulation in European agriculture.

D5.2: Development of a conceptual framework under which information gathered by an environmental regulating agency can be used to estimate individual emissions in an agricultural non-point source pollution (NPS) problem.

The present document constitutes the first part of Deliverable 2.

Deliverable 5 will provide material for Work Package 4: *Quantitative assessment of socio-economic impacts of the Commission proposal of de-coupling* and Work Package 5: *Ex-ante evaluation of the economic effects of decoupling on structural change at farm and regional level.* The more specific contribution of Deliverable 5 to the other workpackages of the project is to provide a solid theoretical foundation for assessing policy impacts using either the framework of differentiative production function or the linear programming framework under appropriate modifications.

3

The Environmental Impacts of European Agriculture

Even though European agricultural is not a major economic sector, since it contributes about 1.8% of GDP of European Union (EU) it is highly important from an environmental aspect, in addition to its well known and well recognized general importance in political and economic structure of the EU (Baldock et al., 2002). Covering on the average 51% of EU territory,[1] farming activities have been responsible to a large extent for the development and stewardship of the landscape. They are also a major determinant of biodiversity in the EU. In particular European agriculture contributes to:

- The maintenance of many cultural pastoral and arable landscapes, as well as semi-natural habitants.[2]

- The decline of greenhouse emissions through soil carbon storage and biomass energy crops.[3]

- The conservation of valued cultural landscapes[4] and farmlands with high natural value in mountainous regions, though low-intensity farming systems.[5]

- The promotion of soil conservation through irrigated agriculture[6] since less land is abandoned and the amount of vegetation on the slopes increase.

- Gains to biodiversity and landscape from certain traditional or "leaky" irrigation systems in some localized areas or from large-

[1] In many Member States agricultural activities cover more that 50% other their territory (Baldock et al., 2002).

[2] Major sectors: beef, sheep and goats, diary and arable.

[3] Major sectors: grassland, energy crops.

[4] The cultural landscapes evolved as low input-low output, usually labour intensive, but sustainable systems of farming often high in biodiversity (EC, 2004).

[5] Examples of such systems include extensive grazing of livestock, small-scale mixed farming traditionally managed, long-established orchards and olive plantations, non-irrigated systems of arable farming in dry lands, hand-operated extensive irrigation systems or even drip irrigation, and systems of low or nil inputs of fertilizers and agro-chemicals.

[6] Based either on drip irrigation or moderated use of water (irrigation by hand).

scale water transfer.[7]

Unfortunately according to reports of the European Commission the role of European agriculture is not always environmentally friendly. In particular farming activities impose pressures on the environment and have been found responsible for:

- **Loss of biodiversity** due to the marginalization or abandonment of farming activities, as well as the switch to simplified, intensive cultivation. Drainage of wetlands, irrigation of arid lands and the ploughing up of unproved grasslands can have a major impact on biodiversity and wildlife. The conversion of large areas of land from dryland production to irrigated or higher-value cropping, as well as the seasonality of demand for irrigation water[8] may disrupt aquatic and wetland ecosystems and may threaten the survival of rare species. For example in a UK RAMSAR site an average of 35 species was recorded in spring-fed areas in the 1950s but by 1992 only 5 species were found.[9]

- **Loss of landscape[10] diversity and quality as well as decline in important habitants and species** due to: increasing scale of production and homogenization on landscapes; intensively managed and irrigated farmland, such as horticulture and arable production; lowland dairying and other livestock housed indoors; decline in labour input for undertaking sensitive land management. For example in the case of peatland fens and bogs, dessication can lead to peat fires which can wipe out large areas of habitants, irreversibly. Large-scale and long-distance water transfer[11] can lead to potentially irreversible negative impacts as landscapes and habitats are submerged under water or damaged by construction activities, and make difficult or even prevent the migration of terrestrial species as artificial barriers are created.[12]

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[7] For example creating artificial aquatic habitants providing new feeding and/or breeding opportunities for wildlife.

[8] That coincides with the period when water flow in rivers tends to be lowest.

[9] Major Sectors: Pastoral systems, areas of former mixed farming now wholly arable, traditional olives and vines in southern Member States.

[10] As *landscape* can be regarded as a system comprising a specific geology, land use, natural and built features, flora and fauna, water sources and climate. To this could be added habitation patterns and socio-economic factors.

[11] Associated with irrigation projects to exploit new sources of water in order to reduce overexploitation of existing sources.

[12] Major Sectors: Arable, dairy, beef, sheep and goats, horticulture, olives, wine, sugar, intensive "southern" crops.

- **Threats to high natural value farming systems and traditional forms of agriculture in marginal areas**. Marginal areas fall into two main categories: regions where extensive systems dominate and those where small scale agriculture is predominant. Although these kinds of agriculture are ecologically sustainable, most of them are not economically viable either due to economic trends - they are relatively labour intensive and produce low value crops - or due to farm enlargement and intesification. Most traditional practices are led either to marginalization and decline or to abandonment.[13]

- **Soil quality pollution** includes reduced organic content and fertility, compaction, heavy metal and agrochemical contamination and acidification. Agrochemical can alter biochemical processes in soils[14] by increasing denitrification or slowing down the mineralization of organic matter. The most important impact is soil erosion - for example water erosion in Portugal and Spain is recorded to 22 tons/hectare per annum - due to selection of erosion prone crops, use of heavy machinery, intensive irrigation[15] and intensive monoculture practices, cultivation on vulnerable soils and land on slopes, excessively large field sizes[16], abandonment of formerly hand-irrigated, traditional terrace agriculture and land left unprotected during rainy periods. This can lead to the subsequent desertification of some arid areas with light and erosion prone soils, particularly on steep slopes. Moreover, improper irrigation practices, the lowering of groundwater table and bad soil management can lead to salinization of land, obstructing roots aeration and generating compounds that are highly toxic to plants, leading to reduced productivity.[17]

- **Air pollution** by ammonia, methane, greenhouse gas emissions and use of toxic substances. Pollution by ammonia is an important source of acidification with impacts on soils, forests, water, biodiversity and buildings. In several European countries agriculture

[13] Major Sectors: Beef, sheep and goats, dairy arable and some "southern" crops e.g. wine, olives

[14] Change the composition of soil: pH, electrical conductivity and capacity for cationic exchange.

[15] Erosion by irrigation water can include: the impact of drops of water on the soil surface (sprinkler systems), laminar erosion from flooding (gravity systems), and erosion in furrows and ditches.

[16] Maize farmers have removed many field boundary features to enlarge fields and thereby reduce the cost of crop husbandry.

[17] Major Sectors: Cereals, maize, oilseeds horticulture, sugar, sheep and goats

accounts for 95% of ammonia emissions - about 80% arises from livestock wastes, and the most of the reminder from nitrogen fertilizers and fertilized crops. In 1990-1997 agriculture contributed about 11% of total EU greenhouse gas emissions.[18]

- **Water pollution** includes eutrophication from farm nutrients and wastes, pesticide contamination and soil sediment, in areas of intensive and irrigated agriculture vulnerable ecosystems or intensive livestock husbandry, affecting both surface and groundwater and also marine environment. Overexploitation of aquifers mostly in areas where irrigated arable crops are grown on thin, drought-prone soils, can have detrimental effects upon the physical and chemical characteristics of these resources, putting into danger the sustainability of their use. Groundwater abstractions can lower water tables and thereby reduce flows into wetlands and rivers, over 600,000 hectares of natural area have been affected by dessication due to groundwater shortages. Surface water abstractions can reduce the volume and increase the variability of flow rates, increasing flooding risks and threatening wildlife due to drying out, water temperature rises or reduced dilution of potentially harmful contaminants. Diffuse pollution of freshwater, contamination of drinking water resources and groundwater salinization, either due to overabstractions or excessive use of fertilizers.[19]

3.1 Pillar II and Environmental Concerns in the EU

The EU common agricultural policy is based on three main instruments (Garaulet and Lawyer, 1999):

1. Markets policy, implemented through the common organizations of the market (COMs).
2. Rural, social and structural policy or else know as the second pillar of the CAP.
3. Harmonization of national legislation for issues not covered by the COMs, transferred to public policy under the Treaty of Amster-

[18] Major Sectors: Cattle (dairy and beef), pigs (Ammonia), Cattle, pigs (greenhouse gas emissions)

[19] Major Sectors: pigs, dairy, beef, horticulture, arable, olives, sugar , arable, dairy, maize, olives, horticulture, sugar, wine, , horticulture, arable, olives, vines, sugar.

dam.

The markets policy of CAP has been accused for a series of environmental damages, including nitrate pollution. Even though the actual environmental impacts can not be easily identified due to the highly variable conditions within Europe and a number of other factors, there exist evidence that the link of the farm support with output levels, increased production at levels that would have not occurred without the support (Baldock et al., 2002). Such supports resulted into intensification and specialization, expansion of cropped area, an overall rise in livestock numbers. A combination of the above factors imposed pressures on the ambient environment.

In particular, the arable regime of CAP encouraged the intensification and expansion of arable agriculture through the plow up of grassland and other valuable habitats, while the olive regime resulted in the marginalization and abandonment of some low input systems in the EU territory (Baldock et al., 2002). For instance, the differentiation of compensation payments per hectare for price cuts among dry and irrigated arable crops of the same type, created incentives to initiate irrigation and led to expansion of the irrigated agricultural area with associated adverse effects on water quality. Furthermore, direct payments encouraged farmers with eligible land to keep cultivating continuously all this land, while led to an expansion of forage maize production which is highly intensive and can cause significant problems regarding soil erosion, water pollution, impacts on wildlife from pesticides and destruction of valuable extensive pasture.

According to the European Commission the introduction of a maximum level of livestock density in the beef and veal CAP regime appears to have supported mid- or even higher intensity systems since farmers maintain the option of meeting the limits by increasing their forage area rather than by reducing stock density on their existing land (Baldock et al., 2002). This could have adverse consequences for biodiversity when this process involves marginal land. Finally, there is evidence that the wine regime has failed to provide adequate support to traditional dryland wine growing, which is being outcompeted by irrigated arable farming, while the restructuring and modernization programmes are pushing technological developments which are not tailored to the environmental requirements concerning land and may increase pollution and result in unsustainable resource use.

Regarding the environmental impacts of rural development CAP measures, they have not yet been assessed, since Pillar II measures were only launched in 2000 by the Agenda 2000 reform. Nevertheless, their potential effects can be inferred through the impacts of their predecessor policies.

Rural development policy is essential for the balanced development of the Union, since rural areas account for more than 80% of its territory (EC, 2004). The aid provided under the Pillar II measures aims to complement the reforms of the markets (first pillar of the CAP) so that the competitiveness and the viability of EU agriculture is enhanced in a sustainable way. Hence the rural development measures can be classified according to the objective they serve:

1. Strengthening the viability of agriculture and forestry section.

This objective can be achieved via measures that assist: (i) modernization of farms, (ii) processing and marketing of quality agricultural products, (iii) setting-up of young farmers, (iv) early retirement, (v) the conservation and improvement of ecological stability of the forestry sector (Garaulet and Lawyer, 1999).

- Aid for young farmers consists either of a single premium or an interest subsidy on loans taken to cover establishment costs and targets heads of holdings under the age of 40 years, who are setting up in farming for the first time (EC, 2004).
- Assistance for early retirement can be granted to farmers and farm workers (family helpers or paid farm workers) over the age of 55 years but not yet of retirement age, who decide to stop all commercial farming activity definitively and their land is reassigned to other farmers or to non-agricultural uses (i.e. forestry, ecological reserves).[20]
- Forestry measures seems to have highly variable environmental impacts, even though there are claims that their implementa-

[20] Farmers who retire early must have practised farming for at least 10 years before stopping, while farm workers must have devoted at least half of their working time to farm work during the five years before stopping (EC, 2004). Moreover, they can receive the annual payment up to the retirement age (age of 75), but not for a total period of more than 15 years per farmer and 10 years per farm worker (Garaulet and Lawyer, 1999).

tion does not satisfy environmental needs with negative consequences on biodiversity (Baldock et al., 2002). Under this measure support is granted to private forest owners or municipalities for: preserving woodlands (i.e. maintaining fire breaks), afforestation of farm land, and investments on non-farm land in order to upgrade harvesting, processing and marketing of forestry products, open up new outlets for forestry products.

- Aid for investment in holdings is provided if the investment pursues certain objectives such as: reducing production costs, promoting best possible product quality, improving or diversifying productive activities, conservation and improvement of natural environment, health and hygiene conditions or animal welfare standards.[21] Even though some rural development measures have been linked to environmental damage in the past (i.e. aid for land reparcelling and water resources management), the particular measure includes extra environmental safeguards and constraints to ensure that the investment aid does not support environmentally harmful developments (Baldock et al., 2002).

- Support for improvements in processing and marketing of products is provided to economically viable firms that comply with minimum standards.[22] The ultimate goal is to increase the competitiveness and added value of agricultural products by improving their presentation, processing procedures and marketing channels, reorienting production to new outlets, applying new technologies, monitoring quality and health conditions, encouraging innovation and protecting the environment.

2. Improving competitiveness of rural areas.

In this context support for vocational training is intended to improve the occupational skill and competence of persons involved in

[21] The total aid is limited to a maximum of 40% of the investment value and 50% for less-favoured regions. The percentages increase to 45% and 55% respectively in the case of young farmers (Garaulet and Lawyer, 1999).

[22] The aid is limited to 50% of the total investment eligible in Objective 1 regions and up to 40% elsewhere. Where Objectives 1 includes the regions which are lagging behind, having either per capita gross domestic product below 75% of the EU average or being less populated (Garaulet and Lawyer, 1999).

agricultural and forestry activities, to help them adapt to changing market conditions and opportunities (EC, 2004). This type of support offers particular environmental opportunities if it is used to raise awareness of environmental impacts and management techniques that are compatible with environmental protection and maintain landscape, hygiene and animal welfare (Baldock et al., 2002). Furthermore, support may be granted to activities not covered by the above measures, but which contribute to the renovation of villages and protection of heritage, promotion of farm-related tourism and craft activities, as well as contribute to converting and improving farming activities such as: improved water management, land reparcelling and land improvement.

3. Preserving the environment and Europe's unique rural heritage.

This objective can be achieved via (i) agri-environmental measures and (ii) compensatory payments in support of farming in less-favoured areas and areas subject to specific environmental constraints. Such measures integrate environmental aspects into the CAP and represent a decisive progress towards recognition of the multifunctional role of agriculture and promotion of environmentally appropriate farming methods (EC, 2004).

It is notable that the agri-environmental measures are the only obligatory CAP measure for all Member States. Annual supports per hectare are granted to farmers, whose agricultural production methods are designed to protect the environment and maintain countryside (agri-environment) for a minimum five year period.[23] Such supports is calculated on the basis of forgone income and additional costs, and aim to promote environmental planning, extensification, conservation of farmed environments of high natural value and the upkeep of landscape. Moreover, compensatory payments are paid per hectare of agricultural land granted in less favorable regions (i.e. mountain areas,), areas affected by specific handicaps and areas subject to environmental constraints.[24] Compensated farmers must ap-

[23] A longer period may be set for certain types of underrtaking (Garaulet and Lawyer, 1999).

[24] Less favoured areas and areas with environmental disadvantages are defined as *mountainous areas* - with min. 700m altitude or min. 20% inclination or min. 500m altitude and 50% inclination. Other less favoured areas with agricultural disadvantages

ply, for at least five years, usual good farming practices compatible with the requirements of environmental protection, maintenance of countryside and sustainable farming. It is notable that no aid is granted where residues of prohibited are found on a holding.

Agri-environmental schemes are expected to bring benefits as they limit pressures from input use, constraint pollution and overgrazing. They have led to modest but worthwhile improvements in the management of livestock; the upkeep and maintenance of field boundaries and small habitats; the application of manure and inorganic fertilizers; the utilization of pesticides; and reduction in the volume of irrigation water. However, there are doubts concerning their effectiveness since the schemes are applied for a relatively short period (five year) and not all farmer rejoin them putting into danger the achieved environmental benefits (Baldock et al., 2002). Moreover, there are doubts whether their current payment formula provides adequate incentive to continue production under the mirroring current market conditions, implying that a more comprehensive formula for calculating payments may be necessary. Finally, several Member States were slow to implement these schemes because government administrations and farmers need time to develop their understanding of new measures and capacity to implement (Baldock et al., 2002).

Summarizing, although there is evidence that Pillar I measures have contributed to environmental pollution problems (water, air, soil etc.) in some areas in the EU territory, there is still little evidence regarding the environmental impact of Pillar II measures which have been actually designed to internalize many major environmental considerations such as for example nitrate pollution which is examined in the following section.

3.1.1 Environmental Regulation and Pillar II

Rural development has become the second pillar of CAP and it is based on two principles: (i) decentralization of responsibilities and, (ii) flexibility. It allows a specific territorial approach to address the needs of rural society and citizens' expectations regarding quality,

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are defined by: number of agricultural holdings: max. 30 and max. 55 inhabitants/km^2 or high employment rate in agriculture (>15%), and by *small areas* - with max. 30 agricultural holdings per region, hilly regions, wetlands and flood plains, border regions (EC, 2004).

food safety and the viability of rural societies. The objective is to introduce a sustainable and integrated rural development policy, to ensure better coherence between rural development and the price and market policy of CAP, and to promote all aspects of rural development by encouraging the participation of local actors. Pillar II complements the reforms of the markets by other actions that promotes a competitive, multi-functional farming in the context of a comprehensive strategy for rural development.

The new rural development policy relating to farming has as main objectives:

- to improve agricultural holdings,
- to guarantee the safety and quality of foodstuffs,
- to ensure fair and stable incomes for farmers,
- to ensure that environmental issues are taken into account,
- to develop complementary and alternative activities that generate employment, with objective the slowing down of the depopulation of the countryside and the strengthening of the economic and social fabric of rural areas,
- to improve living and working conditions and promote equal opportunities.

One of the main innovations was to regroup the series of rural development measures into a single and coherent package, serving three different main objectives:

1. **Strengthening the agriculture and forestry section**. The main measures relate to the modernization of farms and the processing and marketing of quality agricultural products. The viability of farms will be strengthened by measures to assist young farmers to set up farming and by improved conditions for early retirement. The forestry sector was for first time recognized as an essential aspect in rural development, serving an ecological, economic and social function, and a new measure was provided to support the environmental functions of this sector. Forest-fire protection plans are reviewed.

2. **Improving competitiveness of rural areas**. The main objectives are to maintain the quality of life in rural communities and encourage diversification and the creation of new activities.

The measures are designed to create new sources of income and jobs for farmers and their families.

3. **Preserving the environment and Europe's unique rural heritage**. The agri-environmental measures represent decisive progress towards recognition of the multifunctional role of agriculture and promotion of environmentally appropriate farming methods (organic farming). The integration of environmental aspects into the CAP is to be strengthened by expanding the compensatory payments, traditionally in support of farming in less-favoured areas, to areas where farming is restricted because of specific environmental constraints.

Actions to promote the environment are the only compulsory element of the new generation of rural development programmes. This represents a decisive step towards recognizing the role agriculture plays in preserving and improving Europe's natural heritage.

The rural development measures fall into two groups:

- **Accompanying measures of the 1992 reform**: early retirement, agri-environment and afforestation, as well as the less-favoured areas scheme;
- **Measures to modernize and diversify agricultural holdings**: farm investment, setting-up of young farmers, training, investment aid for processing and marketing facilities, additional assistance for forestry, promotion and conversion of agriculture.

Accompanying measures of the 1992 reform

- **Early retirement aid**. Assistance for early retirement is designed both to renew agricultural workforce and make holdings more viable through rationalization, consolidation and improvements in skill levels. Support may be granted to farmers over 55 years of age but not yet of retirement age, who decide to stop all commercial farming activity definitively after having practised farming for at least 10 years before stopping. Support is also available to farm workers (family helpers or paid farm workers) of the same age, belonging to a social security scheme, who have devoted at least half of their working time to farm work during the five years before stopping. This support aims to ensure that older farmers have enough income and can be replaced (provided the holding is profitable) or their land reassigned to non-agricultural uses.[25] Farmers who retire early can receive up to €15 000 per year up to the age of 75. Farm workers can receive up to €3 500 per year up to normal retirement age.[26] If the farmer receives a pension, such as a retirement pension from a Member State, this aid is granted as a supplementary payment (pension 'top-up'). This payment may not be paid for a total period of more than 15 years per farmer and 10 years per farm worker. If the retiring farmer is replaced, the farmer taking over the holding must take over all or part of the land released, possess adequate competence and continue to improve the viability of the holding for at least five years.

- **Agri-environmental measures**. It is the only obligatory measure for all Member States. Annual reports per hectare are granted to farmers giving agri-environmental commitments for a minimum five year period. A longer period may be set for certain types of undertaking. Support can be granted to farmers who, for at least five years, use agricultural production methods designed to protect the environment and maintain the countryside (agri-environment) in order to promote farming methods compatible with the protection of the environment, environmental planning in farming practice, extensification, conservation of farmed environments of high natural value and the upkeep of landscape. The aid is calculated on the basis of forgone income, additional costs and the financial incentive needed to encourage agri-environmental undertakings. Aid may not exceed € 600 for annual crops and € 900 for specialized perennial crops. Aid for all other land uses may not exceed € 450 per hectare per year.

- **Less-favoured areas and areas subject to environmental constraints**. Compensatory payments are paid per hectare of agricultural land granted in less favorable regions, i.e. mountain areas, affected by specific handicaps and areas subject to environmental constraints.[27] The aim is to ensure continued and sustainable

[25] For example forestry, the creation of ecological reserves, etc.

[26] For farmers the maximum is €150 000, while for farm workers it is €35 000.

[27] Less favoured areas and areas with specific environmental disadvantages are defined as: *mountainous areas* - with min. 700m altitude or min. 20% inclination or min. 500m altitude and 50% inclination, other *less favoured areas* - with agricultural disadvantages are defined by number of agricultural holdings max. 30 and max. 55 inhabitants/km^2 or high employment rate in agriculture (>15%), and *small areas* - with max. 30 agricultural holdings per region, hilly regions, wetlands and flood plains, border regions.

agricultural land use, preservation of the countryside, and fulfilment of environmental requirements. Farmers undertake to pursue their farming activity for at least five years, applying usual good farming practice compatible with the requirements of environmental protection, maintenance of countryside and sustainable farming. No aid is paid where residues of prohibited substances or substances are found on a holding. Payments must be sufficient to contribute effectively to a compensation for handicaps without leading to overcompensation. The amount of payments range between € 25 and 200 per hectare, depending on relevant regional development objectives, the seriousness of the permanent natural handicaps affecting farming and the type of production, environmental problems and type of holding. Farmers in areas subject to environmental constraints may also receive support of up to € 200/hectare to cover the additional costs and income losses resulting from implementation of the EU environmental rules.

Measures to modernize and diversify agricultural holdings

- **Investment in holdings**. The basic aim of the measure is to increase farms competitiveness. The total value of EU's aid is limited to a maximum of 40% of the investment value and 50% for less-favoured regions. These percentages increase to 45% and 55% respectively when the investment for diversification is made by young farmers (see below). Support is granted to improve agricultural incomes and living, working and production conditions. Such investments must pursue certain objectives: reducing production costs, improving or diversifying productive activities[28], promoting best possible product quality, conservation and improvement of natural environment, health and hygiene conditions or animal welfare standards, encouragement of diversification of on-farm activities. Only economically viable farms which comply with minimum standards[29], and where the farmer possesses adequate competence, are eligible.

- **Improvement of processing and marketing of products**. Firms which are economically viable and comply with minimum standards may receive such support, limited to 50% of the total investment eligible in Objective 1 regions and up to 40% else-

[28] Except those for which there are no market outlets.
[29] Regarding the environment, hygiene and animal welfare.

where. The goal is to increase the competitiveness and added value of agricultural products by improving their presentation, rationalism processing procedures and marketing channels, reorienting production to new outlets, applying new technologies, monitoring quality and health conditions, encouraging innovation and protecting the environment. Attention is also paid to THC efficiency improvement of distribution channels. No support is available for investments at the retail level or investments in the processing or marketing of products from third countries. Preference is given to low environmental impact initiatives and to projects for product innovation and quality improvement.

- **Aid for the conservation and improvement of ecological stability of woodlands**. Support may be granted to private forest owners or municipalities for the management and sustainable development of forestry, preservation of resources and extension of woodland areas, in order to maintain the economic, ecological and social functions of woodland in rural areas. Aid per hectare depends on the costs involved and may contribute to:

- Improve non-farm land. Measures include afforestation, investment to enhance the value of forests and improve harvesting, processing and marketing of forestry products, opening up of new outlets for forestry products, promote joint action by forest owners and assist the recovery of forestry production after a natural disasters or fire.
- Afforestation of farm land. Aid may be granted to cover the costs of planting and maintenance and to compensate farmers for income forgone, depending on farmer's characteristics. Aid may amount between € 725 and 185 per hectare per year and it is paid for a maximum of 20 years to farmers and producer group.
- Preserving woodlands, where their protective and ecological role is in the general interest and where the cost of preventive measures exceeds the income from silviculture, and maintaining fire breaks. Support can vary between € 40 and 120 per hectare per year.

- **Setting-up of young farmers**. Aid for young farmers under the age of 40 years, who are setting up in farming for the first time. Their holdings must be viable and comply with minimum standards. Aid consists either of a single premium (up to € 25 000), or an interest subsidy on loans taken to cover establishment costs.

- **Vocational training**. Such support is intended to improve the occupational skill and competence of persons involved in agricultural and forestry activities, to help them redeploy production, apply production practices compatible with environmental protection, maintenance of landscape, hygiene and animal welfare, and manage their holdings better. It assists farmers to adapt to changing market conditions and opportunities.

- **Facilitating the development and structural adjustment of rural areas**. Support may be granted to activities not covered by the above measures, but which contribute to converting and improving farming activities, such as improved water management, land reparcelling and land improvement, development of key services in rural areas, renovation of villages and protection of heritage, promotion of farm-related tourism and craft activities, etc.

Member states must, however, ensure that farmers are able to demonstrate that they do not exercise effort solely for the purpose of obtaining the benefit of support payments. They are responsible for ensuring the effective monitoring of implementation of rural development programming.

To improve integration of environmental objectives in CMOs the new reform enables Member States to make direct aid payments conditional on compliance with environmental provisions.[30] All these measures have been applied horizontally and implemented in a decentralized way.

> **Horizontal regulation**: Member States must define environmental measures to be applied by farmers and also define penalties (reduction of direct aid payments) where there are environmental infringements. Direct payments per holding can be determined in relation to the number of jobs on the holdings or the overall wealth of the holding. The resulting funds are available for additional support for agri-environmental measures, LFA, early retirement, rural development and reforestation.

This "horizontal approach" means that there is a common set of rules that can be considered as essential for a sustainable development of agriculture and rural development and they are applied in the same way in all autonomous communities, to ensure that all

[30] Cross-compliance: observance of environmental criteria (Glossary, pg 35).

farmers are equally treated. The horizontal measures are designed to promote sustainable development by improving both the conditions of production and agricultural infrastructure, and provide measures for environmental protection.

Rural development: potential CAP effects

Second pillar measures are both environmentally sustainable by design and use aid to improve the competitiveness of EU agriculture in a sustainable way. However, the rural development measures were only launched in 2000 and thus it is too early to assess the environmental impacts of the second pillar of the Agenda 2000 reform. The likely impacts can be achieved by comparing the impacts of predecessor policies that have formed the basic components of these measures.

- Aid for investment in holdings under the second pillar includes extra environmental safeguards and constraints to ensure that the investment aid does not support expansion in the main commodity sectors, or developments which could be harmful to the environment.
- Forestry measures seems to have had highly variable environmental impacts. Some NGOs claim that their implementation does not integrate environmental needs with negative consequence on biodiversity.
- Some rural development measures have been linked to environmental damage in the past, such as aid for rural development, land reparcelling and water resources management.
- Support for training offers particular environmental opportunities if it is used to raise awareness of environmental impacts and environmental management techniques.
- Member States have to specify verifiable standards of usual Good Farming Practices (GFP), which will form a baseline to ensure that farmers observe mandatory environmental requirements. The later are designed to provide agri-environmental premia only for farmers' commitments going beyond the reference level of GFP.
- Agri-environmental schemes provide both incentive payments and a more supportive policy context for farmers pursuing forms of production which are well matched with environmental requirements but potentially less able to compete with alternative practices. There are doubts whether the current payment formula provides adequate

incentive to continue production under the current market conditions, implying that a more comprehensive formula for calculating payments may be necessary.

- Agri-environmental schemes can bring benefits as they limit environmental pressures from input use, and constraint pollution and overgrazing. They have led to modest but worthwhile improvements in the management of livestock, the upkeep and maintenance of field boundaries and small habitats, the successful application of manure and inorganic fertilizers, the utilization of pesticides and the use of irrigation water.
- There are doubts concerning the effectiveness of agri-environmental schemes, applied for a relatively short period (five year), since not all farmer join these schemes at the end of five years putting thus into danger the achieved environmental benefits.
- Several Member States were slow to implement these because government administrations and farmers needed time to develop their understanding of new measures and their capacity to implement them.

3.2 Surface and groundwater pollution from fertilizers and pesticides: the Case of Nitrate Leaching

Potentially one of the most notable and widely discussed, in economic literature, agricultural pollution problems is nitrate removal from the soil by the action of water (Owen et al, 1998). This phenomenon is known as nitrate leaching (NO_3), includes both leaching below the crop's roots due to the downward movement of water (percolation) and leaching due to the flow of water over the surface of the land (runoff).

Most agricultural cultivations are known to need at least 16 essential nutrient elements to grow, either in large amounts (macronutrients) or in only small quantities (micronutrients) (Owen et al, 1998). For optimum plant growth a specific concentration of each element is required, otherwise either the concentration in the plant root zone is too low or too high and plant growth is restricted (Owen et al, 1998). Among the essential macronutrients is nitrogen, which is the most commonly detected agricultural chemical pollutant in the form of water-soluble nitrates (Johnson et al., 1991).[31] The main sources of nitrogen in the soil for plant uptake are nitrogen fixation (Helfand G.E. and B.W. House, 1995), spreading of animal manure and application of mineral fertilizers (site G), while among the sources of soil nitrate losses are root absorption and leaching (Owen et al., 1998).

The mechanism behind leaching is that the negative electrical charge of the clay particles of soil attracts the positively charged ions (cations) of nutrient elements, while does not retain negatively charged ions very well (Owen et al., 1998). Therefore, if too much nitrogen fertilizers are applied to a crop only the nitrogen ions bond on clay particles are available for plant uptake, preventing leaching, while the rest are not taken by the crop. This residual nitrogen may be lost in the environment and may flow into water aquifers below or lakes and reservoirs in the surface where it accumulates causing contamination (Classen and Horan, 2001; Helfand and House, 1995). Notable is a US study according to which the amount of fertilizer nitrogen taken by crops is rarely greater than 70% and it is typically closer to 50%, while the 90% of applied nitrogen may be lost to the environment when crop yields are near optimum levels (Classen and Horan, 2001).

3.3 Why is it important to regulate nitrate leaching?

Even though farmers do not value chemical leaching directly, it has a positive shadow value to them since it is a by-product positively correlated to production and farmers receive direct benefits from production (Chambers and Quiggin, 1996). In particular, farmers have private incentives to release nitrate emissions (Kampas and White, 2004) since their productions costs are negatively correlated to their emissions,[32] reflecting the fact that technologies that cut emissions are more expensive to employ (Cremer and Gahvari, 2002). However, even though the unintended generation of nitrate emissions offers

[31] Phosphorous is also a major nutrient of environmental concern. Losses generally occur when phosphorous attached to soil particles is removed with sediment during soil erosion. Losses have been estimated to increase approximately linearly with fertilizer application rates (Classen and Horan, 2001).

[32] In general a higher level of emission reduces the private (average) production costs of polluting goods.

private benefits to individual farmers, their decisions may have an adverse effect on the production of all the farmers (Legras, 2004)[33] and creates external social costs on the rest of society (Chambers and Quiggin, 1996) in terms of both natural environment and human health consequences. Therefore the majority of the society wants to control nitrate leaching generated by agriculture. One of the characteristics of leaching which is important for the design of policies to mitigate it, is that its generation is a flow variable while its impact is a stock variable (Legras, 2004).

Among the environmental costs associated with nitrogen flows is water pollution, that poses a threat to freshwater and marine ecosystems (Huhtala and Laukkanen, 2004) and reduces water's value to humans and nature (Owen et al., 1998).[34] The potential for surface water pollution from ground water contaminated with leached nitrate is an important environmental concern (Johnson et al., 1991) since approximately 30% of surface water steam flow is from groundwater sources (Fleming and Adams, 1997). Surface water pollution threatens aquatic life and may induce ecological disasters (Millock and Zilberman, 2004), and has already been linked with loss of aquatic life (Abler and Shortle, 1995) such as benthic invertebrates. Eutrophication of slow flowing rivers, lakes, reservoirs and marine areas appears through the proliferation of algal bloom, which degrades bottom fauna, fish stock and wetlands (Huhtala and Laukkanen, 2004) (Isik , 2004). Some of the effects of such a bloom is the destruction of lake's aesthetic and in particular the chemicals released by blue-green algae are poisonous to fish, cattle and humans (Owen et al., 1998). In this point the question naturally arises regarding the human health consequences of pollution from agricultural fertilizers and pesticides: "*If we drink contaminated water with toxic organisms, will we eventually come down with some serious illness*?". Unfortunately, the answer is not clear since at this point medical knowledge is inadequate (Owen et al., 1998) and few cases

[33] Characteristic is the problem of salinity as described by Karagiannis and Xepapadeas (2001). In such a case individual withdrawals from an groundwater aquifer increase the level of salinity by causing sea water intrusion and hence affect negatively the production fuction of the rest farmers.

[34] Nitrogen flows are also associated with soil and air pollution. In particular soil is at a high risk of eutrophication in cases where excessive nitrogen depletes oxygen in the soil, affecting the functioning of micro-organisms and soil's fertility. Eutrophied soils are also a source of N_2O - a powerful greenhouse gas.

of death or severe illness are directly linked to agricultural contamination (Johnson et al., 1991). The human health consequences of nitrate exposure include methemoglobinemia (or else known as blue-baby syndrome) in infants, gastric cancer in adults and other human risks (Fleming and Adams, 1997; Abler and Shortle, 1995).

3.4 Factors Contributing to Nitrate Leaching

Even though it seems that farmers could eliminate the majority of nitrate leaching by more careful management of nitrogen and water applications, they face a more complex problem (Johnson et al., 1991) since there are factors affecting nitrate residuals which are out of their influence. According to (Horan et al., 1998) and (Shortle et al., 1998):

> Nitrate emissions depend on management decisions, environmental variables and site characteristics (soil type and topography).

Thus even though individual farmers can influence the distribution of their nitrate emissions (Shortle et al., 1998) through their management decisions, they face uncertainty regarding environmental variables and imperfect knowledge and heterogeneity regarding the physical environment of their cultivations.

The factors that contribute to nitrate pollution, as outlined in the environmental economics literature, include:

1. **Imperfect knowledge about soil moisture levels** in the crop root zone at the time of irrigation decision which may result in excessive application of water compared to field capacity resulting in leaching (Johnson et al., 1991).

2. **Imperfect knowledge about soil fertility levels** which may result in excess nitrogen applications (Johnson et al., 1991). Even though nitrogen requirements of the crop depend on the soil quality as represented by its maximum potential yields (Isik , 2004) farmers typically apply more fertilizers than is needed (Owen et al., 1998).

3. **Uncertainty about future weather-related events** which may result in leaching (Johnson et al., 1991) since irrigation and fertilizer decisions are made each day based on expected state of

the nature, before the realization of the uncertainty (Chambers and Quiggin, 1996). Annual chemical losses tend to be largely influenced by a small number of significant weather-related events such as an unexpected heavy rain immediately after irrigation, changes in temperature and wind (Mapp et al., 1994). Even if farmers know with certainty the current fertility and moisture states of their fields, climate variables can change irrigation efficiency, resulting in too much or too little water entering the soil profile (Johnson et al., 1991).

4. **Location and the physical attributes of agricultural land** which affect the production of nitrate pollution (Wu and Babcock, 2001). In particular leaching is more intense and of longer duration in warm and humid regions (Owen et al., 1998). Moreover nitrogen leaching differs between fields because the physical attribute of the field**,** such as soil type, depth of water table, slope (Helfand and House, 1995).

5. **The risk characteristics of ex post output profile** which are related to the degree of nitrate pollution (Chambers and Quiggin, 1996). Production uncertainty is a significant source of risk in all agricultural activities and is dealt through the use of fertilizers and pesticides (Karagiannis and Xepapadeas, 2001). Different production inputs affect in different ways the output's riskiness (Chambers and Quiggin, 1996). Fertilizers are considered to be risk-increasing inputs since they increase both mean yields and yield variability, while pesticides are risk-reducing inputs that increase mean yields and reduce the variance of yields (Karagiannis and Xepapadeas, 2001). Particularly chemical fertilizers can reduce but also increase output riskiness since in the event of severe moisture shortfall they can decrease production (Chambers and Quiggin, 1996).

6. **The risk of underapplying inputs** in parts of the field which may lead to applications of key inputs (water and nitrogen) in excess of the plants' physiological needs (Helfand and House, 1995). This can be attributed to the soil heterogeneity of a field,

7. **Soil texture** which determines soil water permeability and storage, and soil fertility (Owen et al., 1998). Therefore soil heterogeneity can cause nitrate leaching, since a field may have substantial variability in water holding capacity (Johnson et al., 1991). Typical example is a farmer that overirrigates the field to ensure that the most drought-prone part of the field is never stressed (Johnson et

al., 1991). Light-textured soils, such as mocho soils and loamy soils, are more porous and are characterized as nitrate pollution intensive since they permit nitrogen and water to leach more readily below the crop root zone (Helfand and House, 1995). They have significant impact on groundwater pollution, as attempts to keep soil moisture at near-optimum levels for physical output may result in excessive water applications and the leaching of water and nitrates (Johnson et al., 1991). On the other hand high-quality soils, such as clay loam soil and silty soils, are less porous and thus less vulnerable to nitrogen leaching (Wu and Babcock, 2001). They leave little open pore space and prevent water pollution by degrading and immobilizing agricultural chemicals into the soil (Owen et al., 1998) and thus require less irrigation water and nitrogen fertilizer (Helfand and House, 1995).

8. **Farm production choices** defined as: timing, amount and method of fertilizer application, the crop, the particular crop cultivar, application of other inputs and field management practices affecting crop growth (and nitrogen consumption by the plants) and the movement of water across fields, and the use of pollution abatement techniques, influence in general agricultural nitrate losses to the environment (Horan et al., 2002). Some of these choices are presented below as distinct factors.

9. **Management practices** through which water and fertilizer are applied can have significant effects on leaching (Helfand and House, 1995).[35] In general farmer's management decisions include: the use of new or larger quantities of inputs, changes in mechanisms employed, the variations in the numbers, distribution and methods of rearing livestock, and alterations in cropping patterns[36], landscape features and water use. There is a variety of driving forces that affect European agricultural management and have resulted in intensification, marginalization, concentration and specialization of farming, leading into further unbalance the agricultural-environment relationship and indirectly in nitrate leaching. Among these driving forces we can distinguish:

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[35] Changes in management practices and cropping patterns are less likely in the short-run than changes in input use levels.

[36] They include choices of crop, livestock type and breed, cropping patterns and diversity, etc.

- **Changes in market conditions** such as shifting patterns of market demand and supply, increased competition in European markets, changes in relative prices for inputs/factors, market returns, consumer preferences etc.[37] Particularly the low relative cost of nitrogen is considered responsible for elevated nitrate levels in water (Fleming and Adams, 1997; Johnson et al., 1991).
- **Broader economic and social changes in rural areas** such as alterations in the costs of labor, land and other factors of production, availability of credit, population mobility, training, communications, infrastructure and lifestyle choices[38], as well as structural changes.[39]
- **Technology development** in machinery, new varieties, as well as in food processing and distribution, and in input industries. Improved irrigation processes have increased the profitability of some intensively managed damaging crops and have led to the conversion of pasture to irrigated cropland.[40]
- **Public policy measures** in different policy realms covering land ownership and tax, food safety and hygiene, social security, interest and exchange rates, employment, etc. It would be wrong to claim that policy measures are the only or main factor influencing environmental processes, it would be right to claim that they are the easiest one to influence. The Common Agricultural Policy (CAP) is regarded as the primary driving force. Many critiques consider the relatively high support levels offered to intensive agriculture as the principal driver of intensification, while many low input farming systems[41] receive a relatively low share of support. Moreover policy context involves

[37] Notable is the influence of major supermarkets and retailers on farm decisions - regarding enlargement and specialization, the pattern of input and land use, as well as basic husbandry - since they can influence price and food quality attributes, compliance with standards, packaging and presentation, etc.

[38] Which have encouraged people to move away isolated hill communities and towards "growth poles".

[39] Structural adjustments include changes in field and farm size, land drainage, the introduction of irrigation to dry areas, or specialization. Specialization has led to a decline of mixed farming systems. The separation of arable and livestock production results in an increasing dependence on artificial fertilizers to maintain soil fertility.

[40] With the associated biodiversity losses and landscape impacts.

[41] Generally more benign environmentally.

conflicting objectives such as environmental protection and agricultural income support (Huhtala and Laukkanen, 2004) since policies that mitigate farmer risk can affect pollution incentives and for instance increase fertilizers use and thereby increase nitrogen run off (Chambers and Quiggin, 1996). In Particular, full insurance may create inappropriate incentives for pollution control (Chambers and Quiggin, 1996).

- **Institutional changes** that affect farm organization, infrastructure, specific agricultural advice and information.
- **Commercial considerations** to maximize returns and minimize costs, have given rice to a marked intensification of agriculture and to the enlargement of farm size and the removal of landscape features. Concentration of farms in an area may increase efficiency in distribution, the purchase of inputs, and attraction of markets exhibiting high levels of consumer demand.
- **Independent and partly endogenous environmental changes**, such as natural disasters, global warming, flooding, etc.

10. The frequency and duration of irrigation is largely responsible for leaching (Larson et al., 1996). Particularly intensive irrigation practices have been identified as increasing the likelihood of nitrogen and pesticide losses in runoff and percolation (Mapp et al., 1994). Irrigation practice types can be distinguished based on technical and physical criteria:

- **Technical characteristics** concerning whether irrigation supply depends upon pressure or gravity. *Traditional gravity-fed systems* include furrow irrigation where water is transported from surface sources via small channels and whole-field/sheet irrigation where water is used to flood or furrow-feed agricultural land. *Pressure systems* include sprinklers and drip irrigation. Drip systems tend to be more efficient in their use of water but they are far too costly to be within the means of the majority of small irrigators.[42]

[42] Thus these systems tend to be concentrated in regions where farms are relatively large businesses, crops are high-value and/or water price is well established.

- **Time-related characteristics of irrigation** include *permanent irrigation* practiced all-year and every year to produce crops, *support irrigation* practiced every year mainly in short periods during the dry/peak growing season, and *temporary irrigation* practiced only occasionally when there is water shortage.

- **Crop types** can be divided into *extensive crops* that contain lower value or permanent crops, where irrigation is used mainly in arid regions to stimulate enhanced growth and productivity at a fairly low level, *semi-intensive crops* that contain lower value crops where irrigation is used to improve growth rates and productivity either seasonally at times of peak demand or for most of the crop period, *intensive crops* that contain high value crops where irrigation can be critically important to maintain yields and quality and *saturated crops* where water is used to flood fields to facilitate the production of crops.

11. Finally, operating characteristics of the agricultural activity such as farming experience, education and other human capital measures affect both agricultural productivity and production of pollution (Wu and Babcock, 2001).

4

Agricultural Pollution: A Non-Point-Source Pollution Problem

In the environmental economics literature pollution problems are classified as point-source (PS) or non-point-source (NPS) problems. This categorization is based on the available information framework and particularly on the degree of uncertainty or incomplete information about the location of polluting sources, the magnitude of their contribution in the aggregate concentration of the pollutant and their distinctive characteristics (Kaplan et al., 2003). In particular in a PS problem there is perfect information regarding individual emissions, while the opposite occurs in a typical NPS problem. Agricultural pollution, as will be shown in the following sections, is a typical NPS problem and probably the most important. For example, agricultural non-point sources represent over 90 percent of the nitrogen load flowing into the Gulf of Mexico (Cason et al, 2003).Therefore an effective policy, dealing with agricultural pollution, should take into account the NPS pollution features of such problems.

In the following section a more detailed description of PS and NPS pollution problems is provided. We focus in showing the non-point-source nature of agricultural pollution and the many aspects of the problem that should be taken into account in the design of an effective environmental policy. The design of such a policy is very important for the overall EU's agri-environmental policy, since the most important problems associated with EU agriculture are predominantly NPS pollution problems.

4.1 Point-Source Pollution Problems

The pollution problem is a"pure" point source problem if there is perfect information and complete certainty of the location of polluting sources and the individual contributions to aggregate (or ambient) pollution (Kaplan et al., 2003). Nevertheless, this is an extreme information framework that mark the one end of the spectrum of pollution problems (Kaplan et al., 2003), since a pollution problem is also defined as PS problem under an incomplete information framework if the regulatory body can eventually identify the origin and the amount of agricultural pollutants with sufficient accuracy and at a sufficiently low cost.

Point-source pollution is associated with fixed sources usually emitting high levels of pollutants (EU site, yellow) in a well-defined location (Owen et al., 1998). For example discharges of waste water from the pipes of industrial plants into a stream are point sources of nitrogen (EC, 2004).[1] Emission-based instruments can be applied to internalize the external cost of pollution and thus to induce individual agents to manage the export of pollutants at the socially desirable levels (Xepapadeas, 1995). Such regulatory policy instruments are:

- Charges per unit of emissions, known as Pigouvian taxes.
- Systems of marketable emissions permits.
- Direct controls on emissions levels.
- Subsidies as rewards for emissions reduction.
- Deposit or refund systems according to whether certain prespecified conditions of behavior by a potential polluter are satisfied.

Under certainty and perfect monitoring emission taxes or permits are preferable because they directly deal with the source of the discrepancy between private and social costs (Schmutzler and Goulder, 1997) and a first best solution is straightforward: each polluter should pay the marginal external costs of their emissions according to the well known *polluter pays principle.*

4.2 Non-Point-Source Pollution Problems

The pollution problem is a "pure" NPS problem if the regulatory body has no knowledge of the location of polluting sources or the individual contribution in the aggregate pollution (Kaplan et al.,

[1] It is important to underline that even though industrial emissions measured at the factory's pipes are considered as PS pollution their further consequences on soil or water are classified as NPS pollution. Therefore the distinction between PS and NPS pollution is not always clear-cut since (Cochard, 2003).

2003).[2] Nevertheless, the fact that a pollution problem is called NPS problem - or second generation pollution problem (Xepapadeas, 1995) - does not imply that individual emissions cannot be estimated at all but mainly that this is a technically very demanding task and potentially prohibitively costly (Cochard, 2003). In general:

> NPS pollution can be characterized as an information or an uncertainty problem (Kaplan et al., 2003), where informational issues are the core of NPS externalities analysis (Legras, 2004) and the question "*What information is available at what cost?*" plays crucial role in the determination of the best regulatory mechanism (Cabe and Herriges, 1992).

Potentially the most important feature of NPS problems is the associated uncertainty about the decision makers (polluters) and the degree of each agent's responsibility, a fact that eliminates emission-based instruments from the set of available NPS pollution instruments (Shortle et al., 1998). In short the origins of this uncertainty can either be attributed to stochastic influences affecting fate and transport of pollutants, the great number of sources of pollution emissions that can be either static (farms, households) or mobile (vehicles), and/or the regulator's inability to infer individual emissions from ambient pollution levels or inputs used (Xepapadeas, 1995).

Specifically the NPS pollution problems are characterized by:

- "Stochastic pollution processes"

NPS emissions are typically unobservable,[3] stochastic and site-specific (Classen and Horan, 2001; Horan R.D. et al., 2002). Stochastic pollution processes do not only result from variability in natural processes in particular climate (weather, soil or topology) (Kampas and White, 2004) but also by technological uncertainty (Xepapadeas, 1992) and stochastic events such as equipment malfunctions, variations in input quality and process upsets (Malik, 1993).[4] In a NPS

[2] This is an extreme information framework marking the other end of the spectrum of pollution problems (Kaplan et al., 2003).

[3] Unobservable emissions mean that polluters' performance cannot be observed directly (Shortle et al., 1998; Horan et al., 1998; Horan et al., 2002).

[4] Consequently firms do not control completely their emissions (Malik, 1993).

pollution problem only ambient pollution can be observed at prespecified receptor points, but no specified portion of the pollutant concentration can be attributed to a specified discharger (Xepapadeas, 1991) - only quantified approximations can be made (Franckx, 2002; Camacho and Requate, 2004).

- "Multiple dischargers and diffuse pollution"

By definition NPS pollution sources are numerous and spatially distributed (Legras, 2004), while NPS pollution is diffuse in origin, originating from a wide range of actions and geographic locations (Herriges et al., 1994). In this case the fundamental relationship between polluted area and source is not known with certainty (Kaplan et al., 2003). It is difficult and very costly for the regulator to obtain information that enables him to link the damage to the responsible agent among a large population of potential polluters (Millock and Zilberman, 2004) due to the off-site consequences of individual chemical applications (Cabe and Herriges, 1992) that involve moving as well as mixing over large areas (Larson et al., 1996). Particularly, in the agricultural case:

> The effects of nitrate leaching can be felt and measured (if at all) after they have entered the ecosystem, but identifying polluting resources source may be impossible (Chambers and Quiggin, 1996; Kampas and White, 2004) since agricultural pollutants does not enter waterbodies at a defined point (Helfand and House, 1995), they dissipate quickly and the area vulnerable to pollution is extensive (Johnson et al., 1991).

The pollutant can cause harm in multiple zones (Cabe and Herriges, 1992) due to biogeophysical processes (Wu and Babcock, 2001) that transform human activity in a place (application of a chemical) into chemical concentrations in another place (pollution) (Cabe and Herriges, 1992), affected by stochastic environmental variables that influence transport and fate of pollutant once it has left the field to water resources. Even if the effluent were observed, these biogeophysical processes (Wu and Babcock, 2001) characteristics may make impossible to relate the effluent to its source (Helfand and House, 1995).[5]

- "Monitoring and measurement inefficiency"

By definition in NPS pollution problems the individual emissions can not be observed directly or be inferred indirectly by observed inputs or ambient pollution concentration (Xepapadeas, 1997). The problem is associated with the inability to monitor and measure efficiently individual emissions or abatement efforts due to budgetary restrictions related to the cost of monitoring technology and to personnel limitations or legal restrictions such as the inability to enter the polluter's premise (Xepapadeas, 1991). Emissions levels can be approximated through inspection and monitoring (Cremer and Gahvari, 2002) of each polluter, however measurements cannot be obtained without incurring costs (Schmutzler and Goulder, 1997). The cost of information relating to monitoring and measurement and the associated budgetary restrictions often limit the extent of controls (Kaplan et al., 2003). These limitations are more noticeable especially where continuous monitoring is required (Schmutzler and Goulder, 1997) since it is relatively easy to determine whether the polluter has established adequate abatement equipments but it is difficult to verify that this equipment is operated at the desirable level (Xepapadeas, 1991). Nevertheless, even though the current monitoring technology renders prohibitive the accurate measurement of nonpoint emissions at reasonable cost and introduces uncertainty about nonpoint emissions and their fate (Shortle et al., 1998), it is notable that:

> The classification of an individual source of pollution as NP may change over time as monitoring technology advances and the cost of monitoring declines (Millock et al., 2002).

An important difference between the PS and NPS regulatory mechanisms is due to the different cost structures for the acquisition of information regarding important parameters of the problem such as individual emissions (Cabe and Herriges, 1992) often creating an

[5] Nevertheless in the case of pollution of waterbodies the number of involved agents can be reduced to those located above the watertable (Legras, 2004).

ill-posed estimation problem (Kaplan et al., 2003)[6] Moreover, the limitations on the capability of existing technology for emissions monitoring (Millock et al., 2002) may impose some imperfections since there is a probability that the regulator does not infer correctly the existing pollution state and a firm may be erroneously fined (Malik, 1993). For agriculture it is extremely difficult and expensive to determine how much a certain action pollutes ground or surface water (Underwood and Caputo, 1996), as well as to determine the groundwater pollution and monitoring groundwater quality (Cabe and Herriges, 1992). Consequently, a monitoring technology that guarantees perfect observation of emission levels of individual economics agents would be prohibitively costly for any environmental regulator (Franckx, 2002).

- "Informational asymmetries"

NPS pollution problems operate in a setting of incomplete information and dual information asymmetry (Cason et al., 2003).[7] In the real environment neither the regulator nor firms have full information, having instead access to a subset of the information set (Horan et al., 2002). They both experience additional uncertainty about the exact specification of the emission, the nature of the transport mechanism,[8] as well as imperfect knowledge of relevant physical processes and the ambient concentrations (Horan et al., 2002).[9] However the information that is often needed for policy design by environmental authorities is only known by those who are to be regulated (Wu and Babcock, 2001) and this is one reason why it is particularly impossible to achieve a so-called "first-best" solution to NPS pollution reduction problem (Šauer et al., 2003). In particular, the regulator is usually unfamiliar with the full range of microeconomic parameters of NPS pollution problems (Šauer et al., 2003) and has limited information about the strategic environment of private polluters, who know only their own payoff function, their maximal emission level

[6] This problem could be avoided if the regulator could wait for a sufficiently long time so that data can be collected, to balance, in any given period, the number of observations with the large number of polluting sources. However there is a risk that irreversible damage could occur in the mean time (Kaplan et al., 2003).

[7] They are often cast as non-cooperative, asymmetric games.

[8] Due to factors such as weather.

[9] According to Cabe and Herriges (1992) both the regulator and firms form different priors on the distribution of these unknown factors.

(Cochard et al., 2004) and have better information about aspects of their operation - such as production techniques, abatement or polluting input choices (Cremer and Gahvari, 2002). Moreover, benefits and environmental damage costs are not well known and some times are completely unknown (Spraggon, 2002) by the regulator due to either imperfect information about the true costs and benefits of pollution abatement or stochastic factors such as weather (Wu and Babcock, 2001).

> Under these informational asymmetries the NPS pollution problems are subject to moral hazard in teams, characterized by hidden actions, and/ or adverse selection.

Moral hazard is defined as the incentive problem of inducing polluters to provide socially targeted levels of abatement effort given that their actions cannot be effectively monitored (Herriges et al., 1994).[10] In this case polluters choose higher emission levels (lower level of abatement) than the socially desired to increase their profits (Xepapadeas, 1992), since their actions can not be observed and expected costs of shirking are lower under this information barrier (Cavaliere, 2000). On the other hand, adverse selection is associated with the inability to know the specific characteristics or type of each polluter (Xepapadeas, 1999) (e.g. profit functions) (Cochard, 2003). Agriculture is a collective enterprise where the outcome of all dischargers', that is farmers, combined effort is observed by the regulator (e.g. in terms of water quality) (Xepapadeas, 1991), while the exact conditions under which production takes place cannot be observed (e.g. nitrogen use) (Chambers and Quiggin, 1996) and thus an individual's contribution to the team's output (nitrate leaching) is not distinguishable (McAfee et al., 1991). Therefore in a situation characterized by these informational asymmetries it is impossible to charge each agent according to its individual emissions productivity. A difference between socially and individually optimal actions is observed, leading to inefficient equilibria and environmental shirking that implies too little effort and too much pollution (Herriges et

[10] The moral hazard problem occurs whether the relationship between individual's net emissions and ambient concentration levels is deterministic or stochastic (Xepapadeas, 1991).

al., 1994). Consequently, the relative performance of NP pollution controls depends on their effectiveness in reducing environmental shirking (Shortle et al., 1998).

4.2.1 Policy Instruments for Non-Point-Source Pollution Problems

These characteristics of NPS agricultural pollution limit the range of potential policy instruments and also the efficiency of many remaining options (Horan et al., 2002; Classen and Horan, 2001). By definition NPS pollution problems are not susceptible to traditional direct policy controls (Kaplan et al., 2003), since they are based on individual emissions that are unobservable and typically stochastic. Standard instruments, such as Pigouvian taxes, tradeable permits and emission standards, appear to be inadequate to handle efficiently with the NPS pollution problem and deliver the Pareto optimum outcomes, in terms of environmental quality (Xepapadeas, 1999).

Even though the regulation of NPS pollution problems such as nitrate pollution of lakes and groundwater is a major policy challenge (Hansen, 2002; Cochard et al., 2004), it is:

> The combination of inability to employ emission-based policy and agricultural land features that makes NPS pollution unique and more difficult to control than PS pollution (Wu and Babcock, 2001).

Due to the limited relevance of emission-based economic incentives (Horan et al., 1998) NPS control has focused on other elements of NPS pollution problems that may be observable. Consequently the regulatory authority can handle NPS pollution through policy schemes based either on output, inputs, emission proxies, ambient pollution or ex post liabilities for real damages. It is evident that the potential NPS measures can either be associated with polluters' decisions (inputs, management practices and technologies) or with the concequences of their actions (output and ambient pollution) (Cochard, 2003). These measures can be further distinguished in fiscal (price-based) approaches consisting of tax or subsidies, and command-and-control approaches consisting of mandated technologies or performance standards (restrictions on inputs or outputs).

In economic literature relative emphasis has been given on fiscal policy schemes that can be classified into two main classes of policy schemes: *input-based schemes* and *ambient-based schemes.* Input-based measures are purely individual schemes (Cochard, 2003) that involve the indirect control of ambient concentrations through the control of observable inputs, related to the creation of emissions. Particularly, they include taxes for inputs that increase NPS emissions, as well as subsidies for inputs that reduce NPS emissions such as observed pollution control equipment (Shortle et al., 1998). These measures can be further distinguished into non-uniform or uniform and into broad or targeted. On the other hand, ambient-based schemes are dependent on observed ambient pollution in a given receptor point. They involve direct control of ambient concentrations either through collective or random penalties, with budget or nonbudget balancing features. These policy measures can be further distinguished into damage or variance based-schemes, and into linear and nonlinear ambient tax mechanisms. This category also includes NPS instruments that consist of collective ambient and individual Pigouvian charges, known as mixed-based schemes, even though some times these instruments are regarded as a separate NPS policy instrument.

In short the existing policy options for NPS pollution problems, under each potentially available information set, are:

Non-Point Source		
	Available information	*Potential regulation*
Input use	may be observable	Input-based schemes
Individual emissions	Unobservable or only at very high cost	no regulation
Ambient pollution	may be observable	Ambient-based schemes

Source: Cochard F., (2003)

Agricultural NPS problems such as nitrate pollution can be handled via Voluntary Approaches (VAs). This particular instrument is actually a complement to a conventional regulatory system since it combines both voluntary and mandatory tools (i.e. input-related performance standards, ambient taxes), and can be regarded as a very important instrument of EU's current agricultural policies. VAs are based on a new type of interactions between regulators and economic actors, and are usually classified as unilateral commitments, public voluntary schemes and negotiated agreements. Apart from this first classification there is large list of criteria to further differentiate voluntary approaches such as : initiator, degree of detail, legal obligation, sanction types which will be presented in detail latter on.

Finally NPS pollution can be also handled through liability for damages, land-set aside programs which are promoted by the EU in the context of VA type regulation, markets and moral suasion. Markets involve trades between PS and NPS emissions permits, even though there is still question about the appropriate basis for measuring NPS performance (Cochard, 2003). Finally moral suasion involves educational programs supplemented by technical and financial assistance for the reduction of chemical inputs by farmers(Abler and Shortle, 1995).

5

Modelling Agricultural Pollution as Non-Point-Source Pollution Problems

In this section agricultural pollution is analyzed in the context of NPS pollution. By presenting a series of alternative models, insights are provided not only to the complex nature of the problem but also to methods and approaches for regulating agricultural pollution. The NPS pollution model is provided both under a static and dynamic context, since the generation of pollution is a flow variable but its impact is a stock variable (Legras, 2004), as well as under an asymmetric information framework.

5.1 Static Non-Point-Source Pollution Model

Consider a geographical region that supports a wide variety of tradeable agricultural commodities, produced by a large number of $i = 1, ...n$ small farmers. The activities of farmers do not only result in an specific, intended crop[1] but also in unintended emissions of pollutants. To make the problem more specific and more relevant to European agriculture, the ongoing analysis is held in the context of Nitrate Directive (91/676/EEC) and Pillar II measures, associated with agricultural pollution. Therefore it is considered that emissions constitute nitrate leaching, generated by farming activities and could be damaging for a particular resource located in the territory such as the water quality of the groundwater aquifer or lake. Even though production itself is socially desirable, its by-product is undesirable and in the absence of regulatory intervention the competitive market fails to deliver the ex-ante efficient allocation of resources.

In this section the behavior of agents involved (that is farmers)

[1] For simplicity in the exposition a single crop is considered such as wheat, corn etc.. The analysis and the results regarding policy implications will not change considerably in a multi crop set up.

is analyzed under a context of certainty and uncertainty, The individual and collective nitrate leaching processes are examined and we further analyze market failures that justify the introduction of environmental policies.

5.1.1 *The Model under Certainty*

Agents' actions are associated with emission generation as a by-product of agricultural production. Based on Xepapadeas (1997) the net nitrate emissions released in the ambient environment by individual farmers can be specified as the difference between the gross emission function $e_i^G = e_i(\mathbf{x}_i^p)$ and the abatement function $a_i = a_i(\mathbf{x}_i^a)$. Thus individual net emissions are equal to:

$$e_i^N = e_i^G - a_i \tag{5.1}$$

The emission function is convex in production choices $\mathbf{x}_i$ and strictly increasing in productive and emission-generating inputs $(\mathbf{x}_i^p)$ while it is strictly decreasing in abatement inputs $(\mathbf{x}_i^a)$. In particular it holds that $e_i^N > 0$ for polluting inputs with $\frac{\partial^2 e_i^N}{\partial(\mathbf{x}_i^p)^2} > 0$, and $e_i < 0$ for abating inputs with $\frac{\partial^2 e_i^N}{\partial(\mathbf{x}_i^a)^2} > 0$.

The collective by-product of farmers' activity is defined as ambient pollution and affects the water quality of the associated groundwater aquifer or lake. Based on Xepapadeas A. (1997) a deterministic model of ambient pollution can be specified as:

$$e = \sum_{i=1}^{n} \{e_i(\mathbf{x}_i^p) - a_i(\mathbf{x}_i^a)\} \tag{5.2}$$

The ambient pollution e is convex and strictly increasing in individual nitrate leaching e_i.

Farmers: In the absence of any regulation the individual profits of each nonpoint farmer i are defined as:

$$\pi_i(\mathbf{x}_i) = pf(\mathbf{x}_i) - \mathbf{w}_j \mathbf{x}_i$$

The vector $\mathbf{x}_i = (x_{i1}, ..., x_{im})$ represents the production choices of the agricultural activity i among a set of $j = 1, ..., m$ inputs, where x_{ij} denotes the quantity of input j used by agent i. Production choices include both *material inputs* such as land, machines, seeds, nutrient and pesticides application rates and *immaterial inputs* such as practices, knowledge, technology (Cochard, 2003). Moreover, farmers operate in a perfectly competitive market and they are price-takers since their actions do not to have any collective influence on the vector of input prices $\mathbf{w}_j = (w_1, ..., w_m)$ or on the output price p (Horan et al., 1998). Input and output markets are assumed to be free of distortion in the absence of government intervention (Horan et al., 2002). Finally, $y_i = f(\mathbf{x}_i)$ represent the crop yields with non-negative marginal products.

The vector $\mathbf{x}_i$ of input choices can be further divided into a vector of g productive and emission-generating inputs $\mathbf{x}_i^p = (x_{1i}^p, ..., x_{gi}^p)$, such as nitrogen, and a vector of $(m - g)$ abatement inputs $\mathbf{x}_i^a = (x_{(g+1)i}^a, ..., x_{mi}^a)$, such as drip irrigation. In this case the objective function is specified as:

$$\pi_i(\mathbf{x}_i) = pf(\mathbf{x}_i^p) - \mathbf{w}_j^p \mathbf{x}_i^p - \mathbf{w}_j^a \mathbf{x}_i^a$$

where $\mathbf{x}_i = (\mathbf{x}_i^p, \mathbf{x}_i^a)$. It is worth mentioning that Pillar II measures are expected to influence both productive and abatement input choices, as well as production patterns. In particular the provision of *voluntary training* is intended to improve the occupational skill of farmers and help them apply production practices compatible with environmental and natural resources protection, maintenance of landscape, hygiene and animal welfare. More importantly *agri-environmental programs* involve the provision of support to farmers that reduce substantially the use of fertilizers and / or plant protection products, or maintain the reductions already made, introduce or continue with organic farming methods, change to or maintain extensive production (both in terms of crop and livestock) and set-aside farmland.

Under certainty the i-th polluting agent, the farmer in our case, wishes to achieve the maximum private net benefits from production for any choice of inputs $\mathbf{x}_i = (\mathbf{x}_i^p, \mathbf{x}_i^a)$. Therefore farmer i's problem can be defined as:

$$\max_{\mathbf{x}_i} \pi_i(\mathbf{x}_i) \tag{5.3}$$

Equivalently the polluters' problem can be determined in terms of abatement cost as $\min_{\mathbf{x}_i^a} C_i(\mathbf{x}_i^a)$.

The solution of the maximization problem (5.3) defines the optimal vector of input choices $\mathbf{x}_i^o$ that maximizes farmer i's profits in the unregulated competitive equilibrium. The mn first-order-conditions (FOC) with respect to x_{ij} are:

$$FOC_{x_{ij}} : \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} = 0 \qquad \forall i, j \tag{5.4}$$

where the optimum vectors $\mathbf{x}_i^o$ sets the diminishing marginal private net benefits equal to zero. More precisely the optimal vectors of productive $\mathbf{x}_i^{op}$ and abating $\mathbf{x}_i^{oa}$ inputs must satisfy the following equalities:

$$FOC_{x_{ij}^p} : p\frac{\partial f_i(\mathbf{x}_i^p)}{\partial x_{ij}^p} = w^p \quad \text{and} \quad FOC_{x_{ij}^a} : w^a = 0 \qquad \forall i, j$$

According to the initial condition the profit maximizing vector $\mathbf{x}_i^{op}$ is nonnegative and equates the marginal value product of productive inputs with their competitive prices. However the later condition implies that in the absence of regulation the individual farmers use no abating inputs and thus $\mathbf{x}_i^{oa} = 0$.

Finally, the farmer i may target the maximum utility from profits, defined as $U_i(\pi_i(\mathbf{x}_i))$, for any choice of inputs $\mathbf{x}_i$. The utility function is strictly increasing and concave in profits since $\frac{\partial U_i(\cdot)}{\partial \pi_i} > 0$ and $\frac{\partial^2 U_i(\cdot)}{\partial (\pi_i)^2} \leq 0$ hold respectively (Cochard, 2003).

Regulator: Ambient pollution e is costly for society. Each individual in the society is affected by the total amount of emissions ($e = \sum_{i=1}^{n} e_i(\mathbf{x}_i)$) and the associated economic cost or damage can be defined as the sum of individual damages $D(e) = \sum_{h=1}^{I} d_i(e)$, for $h = 1, ..., I$ individuals.

In a risk-neutral society the regulator seeks to achieve the ex-ante efficient allocation of resources that maximizes social net benefit resulting from agricultural operation. Therefore the regulator's problem is:

$$\max_{\mathbf{x}_i} NSB = \sum_{i=1}^{n} \pi_i(\mathbf{x}_i) - D(e) \tag{5.5}$$

where $\sum_{i=1}^{n} \pi_i(\mathbf{x}_i)$ is the sum of farmers' profits. Based on definition (5.2) the regulator's problem can be also specified as:

$$\max_{\mathbf{x}_i^p, \mathbf{x}_i^a} \sum_{i=1}^{n} \pi_i(\mathbf{x}_i^p, \mathbf{x}_i^a) - D\left(\sum_{i=1}^{n} \{e_i(\mathbf{x}_i^p) - a_i(\mathbf{x}_i^a)\}\right) \qquad (5.6)$$

Equivalently, the regulator may wish to minimize the sum of abatement costs and environmental damage, an alternative expression of social welfare defined in terms of cost as: $\max_{\mathbf{x}_i^a}\{-\sum_{i=1}^{n} C_i(\mathbf{x}_i^a) - D(e)\}$ (Hansen, 2002).

The (5.5) defines the optimum vector of production choices $\mathbf{x}_i^*$ that maximizes the social welfare measure. The mn first-order-conditions with respect to x_{ij} are:

$$FOC_{x_{ij}} : \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} = \frac{\partial D(e)}{\partial e} \frac{\partial e(e_i)}{\partial e_i} \frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \qquad \forall i, j \qquad (5.7)$$

where at the social optimum the marginal profit from the use of each input equals the expected marginal damage from the use of the input.

In the same context the social optimum vectors of productive $\mathbf{x}_i^{*p}$ and abating $\mathbf{x}_i^{*a}$ inputs are defined respectively by the associated $FOCs$ of the maximization problem (5.6), where it holds:

$$FOC_{x_{ij}^p} \quad : \quad p\frac{\partial f_i(\mathbf{x}_i^p)}{\partial x_{ij}^p} = w^p + \frac{\partial D(e)}{\partial e} \frac{\partial D(e)}{\partial e_i} \frac{\partial e_i(\mathbf{x}_i^p)}{\partial x_{ij}^p} \qquad \forall i, j \qquad (5.8)$$

$$FOC_{x_{ij}^a} \quad : \quad \frac{\partial D(e)}{\partial e} \frac{\partial D(e)}{\partial e_i} \frac{\partial a_i(\mathbf{x}_i^a)}{\partial x_{ij}^a} = w^a \qquad \forall i, j \qquad (5.9)$$

According to these conditions the social optimum vector $\mathbf{x}_i^{*p}$ equates the marginal value product of productive inputs with the marginal social costs, while the social optimum vector $\mathbf{x}_i^{*a}$ equates the marginal damage saving with their competitive prices (Xepapadeas, 1997). Both conditions define a nonnegative vector of inputs, implying that $\mathbf{x}_i^{*p}, \mathbf{x}_i^{*a} > 0$ respectively.

Finally, an alternative definition of regulator's problem is provided by Xepapadeas (1992) where the regulator's problem (5.5) or (5.6) can be solved under the constraint $e = \sum_{i=1}^{n} e_i$, where the associated Lagrangian multiplier $\lambda < 0$ defines the marginal social cost of pollutant concentration.

5.1.2 The Model under Uncertainty

The previous definitions of individual and ambient pollution are deterministic since they ignore potential stochastic elements that may affect both emission generation and emission abatement. In this section uncertainty is introduced.[2] Based on Horan et al. (1998) nitrate leaching e_i from each non-point source i can be modeled as:

$$e_i = e_i(\mathbf{x}_i, u_i, \alpha_i) \tag{5.10}$$

where $\mathbf{x}_i = (\mathbf{x}_i^p, \mathbf{x}_i^a)$ is a vector of both material and immaterial production choices, u_i represents stochastic environmental variables at farm's site and α_i is a vector of site characteristics such as soil type, topography and location.[3]

In the same context the agricultural ambient pollution is defined as:

$$e = e(e_i, b, \omega, \beta) \tag{5.11}$$

where $e_i = (e_1, ..., e_n)$ is a vector of nitrate residuals generated by each agent, b denotes the natural generation of the pollutant, ω represents stochastic influences on transport and fate of the pollutant such as weather or technology, and β is a vector of watershed characteristics and parameters.[4] According to Malik (1993) pollution in the form of chemical spills is widely recognized to be stochastic.[5]

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[2] We follow the approach presented initially by Horan et al. (1998), Shortle et al. (1998) and further adopted by Horan et al. (2001 and 2002) and Cochard F. (2003).

[3] According to Cochard (2003) the emission function (5.10) can be simplified to $e_i = e_i(\mathbf{x}_i, u_i)$, since some of the parameters affecting individual nitrate leaching are deterministic (soil type, topology) while others are stochastic (e.g. rainfall).

[4] As previously (5.11) can be simplified to $e = e(e_i, \omega)$. Furthermore, measured ambient concentrations can be defined for simplicity as $e = \sum_{i=1}^{n} e_i + \omega$, denoting that group total is observed with error (Cochard, 2003).

[5] At this point it is worth mentioning a more specific model of ambient pollution, developed by Fleming and Adams (1997). The proposed geohydrology model includes a groundwater solute transport model that calculates groundwater nitrate concentration every 5 (dt_c) days at each node over the study area as:

$$e_{i,j,t}^{\lg w} = e_{i,j,t}^{gw} + \left(\frac{wl_{i,j,t}}{deep \times n_a \times 12}\right)\left(\frac{dt_c}{365} e_{i,j,t}^{sw} - e_{i,j,t}^{gw}\right)$$

where $e_{i,j,t}^{\lg w}$ and $e_{i,j,t}^{gw}$is the concentration of nitrate in groundwater after loading and before loading respectively, while $e_{i,j,t}^{sw}$ is the soil water nitrate concentration, $wl_{i,j,t}$ the depth of the soil water leached and n_a the porosity or actual pore space containing groundwater. Finally $deep$ is the depth of the aquifer and i and j represent directions.

Farmers: If the profit function is stochastic due to stochastic factors (θ) affecting crop yields[6], then it is represented by the expected profit function $E_i\{\pi_i(\mathbf{x}_i, \theta)\}$ that is strictly concave in input choices (Cochard, 2003). As a consequence, a risk-neutral farmer i aims at maximizing expected private net benefits and the problem is:

$$\max_{\mathbf{x}_i} E_i\{\pi_i(\mathbf{x}_i, \theta)\} \tag{5.12}$$

The first-order-condition with respect to x_{ij} that determines the profit maximizing vector of inputs $\mathbf{x}_i^o$ for each farmer in the competitive market is:

$$FOC_{x_{ij}} : E_i(\frac{\partial \pi_i(\mathbf{x}_i, \theta)}{\partial x_{ij}}) = 0 \qquad \forall i, j. \tag{5.13}$$

On the other hand, under risk-aversion farmers are supposed to be von Neuman-Morgenstern expected utility maximizers that choose input levels to achieve the maximum expected utility from profits, defined as $E_i\,\{U_i(\pi_i(\mathbf{x}_i, \theta))\}$ with E_i the farmer's mean operator over all the stochastic or inherently unknown variables (elements of $\omega, \alpha_i, \lambda$) (Shortle et al., 1998).

Regulator: Damages are stochastic if emissions and ambient pollution are stochastic and thus can be specified as: $D(e, \eta)$. The variable η introduces damage cost uncertainty, resulting from stochastic processes that influence the economic consequences of ambient pollution concentration levels (Horan et al., 1998) and make difficult to measure the true cost of damages (Fleming and Adams, 1997). In NPS problems the damage function is strictly increasing and convex as usual in environmental economics models, in which case $\frac{\partial D(e,\eta)}{\partial e} > 0$ and $\frac{\partial^2 D(e,\eta)}{\partial e^2} \geq 0$ hold respectively. Moreover, it holds $\frac{\partial D(e,\eta)}{\partial \mathbf{x}_i^a} < 0$ indicating that abating inputs decrease social damage.

In a risk-neutral society the regulator seeks to maximize the expected social net benefit of agricultural production. Therefore the problem can be defined as:

$$\max_{\mathbf{x}_i} \sum_{i=1}^{n} E_i\{\pi_i(\mathbf{x}_i, \theta)\} - E[D(e, \eta)] \tag{5.14}$$

[6]Crop fields can in general be dependent on input choices, site characteristics and stochastic events.

The first order condition $FOC_{x_{ij}}$ with respect to x_{ij} define the social welfare maximizing vector of input choices $\mathbf{x}_i^*$:

$$E_i(\frac{\partial \pi_i(\mathbf{x}_i,\theta)}{\partial x_{ij}}) = E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i,b,\omega,\beta)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i,u_i,\alpha_i)}{\partial x_{ij}}\right] \qquad \forall i,j \tag{5.15}$$

In a risk-averse society the regulator wishes to maximize the expected value of a social welfare function based on the utility of profits and damages, defined as $E\{W(U_1(\pi_1(\mathbf{x}_1,\theta)),...,U_n(\pi_n(\mathbf{x}_n,\theta)),D(e,\eta))\}$ (Horan et al., 2002) where E denotes the regulator's expectation operator over all the stochastic or inherently unknown variables (Shortle et al., 1998).

Finally, following Cochard (2003) modifications the set of farmers n is considered to be the optimal set - even though to neglect exit and entry considerations can be important in a long run approach.[7]

An alternative definition of regulator's problem is provided by Horan (2001), which however does not necessarily provide the ex ante efficient allocation. In this case the regulator wishes to maximize the objective function (5.14) under a "generic goal" that can either be defined as a lower bound on expected private net benefits or as an upper bound on private pollution control costs. According to Cochard (2003) the most simple constraint proposed in literature is $E(e) \leq \bar{e}$, implying that the expected ambient pollution must not exceed a specified level. The constraint takes the following general form:

$$E\{W(\mathbf{x}_i)\} \leq W_0 \tag{5.16}$$

The first-order-condition with respect to x_{ij}, $\forall i,j$ of the revised regulator's problem as defined by the associated Langrange function implies:

[7] The following condition defines the optimal number of producing agents (farmers) in the region.

$$\pi_n - \Delta_n E(D(e,\eta)) = 0$$

where $\Delta_n E(D(e,\eta))$ is the difference in expected damages when farmer n engages in production and when he does not. If the optimal number of farmers is n then the addition of an extra activity $(n+1)$ has a negative effect on the expected net benefits (Horan et al., 1998).

$$E_i(\frac{\partial \pi_i(\mathbf{x}_i,\theta)}{\partial x_{ij}}) - E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i,b,\omega,\beta)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i,u_i,\alpha_i)}{\partial x_{ij}}\right] - \lambda E(\frac{\partial W(\mathbf{x}_i)}{\partial x_{ij}}) = 0$$

where the Lagrangian multiplier λ is the shadow value of the constraint. The allocation of input choices satisfying this equality is considered to be cost-effective allocation, since it minimizes the private and social costs given the particular goal (Horan, 2001). However, based on Cochard (2003) the value λ is overdetermined since it is the same for every j, failing to deliver the social optimum and thus justifying the given characterization of "second best social objective".

It is important to note that if the regulator and farmers share the same information set, Ω then all random components are assumed to be jointly distributed according to the same density function $f(\mathbf{u},\omega,\eta\,|\Omega)$, where $\mathbf{u}$ is the of relevant variables. In this case the farmers' and the regulator's expectations E_i and E respectively, coincide (Horan et al., 1998). However, in practice neither the regulator nor the farmers access the information set, implying that regulator's information set (Ω^*) is different than the farmer i's information set (Ω_i). In such a case the jointly distribution of all random variables is given by $g_i(\mathbf{u},\omega,\eta\,|\Omega_i)$ for the each farmer i and $h(\mathbf{u},\omega,\eta\,|\Omega^*)$ for the regulator, denoting the difference in expectations between farmers and regulator ($E \neq E_i$) (Horan et al., 2002; Cabe and Herriges, 1992).

5.1.3 Market failure

After comparing the first-order-conditions (5.7) and (5.4) under certainty and conditions (5.15) and (5.13) under uncertainty, it is evident that there is deviation from the socially-optimal vector of input choices ($\mathbf{x}_i^*$) as defined by the problem of the social planner and the farmers' optimal vector of input choices ($\mathbf{x}_i^o$) as defined under an unregulated competitive market. Particularly the inequality $(\mathbf{x}_i^*) < (\mathbf{x}_i^o)$ holds in both contexts due to the strictly concavity of the profit function since $\frac{\partial \pi_i(\mathbf{x}_i^o)}{\partial x_{ij}} = 0$ while $\frac{\partial \pi_i(\mathbf{x}_i^*)}{\partial x_{ij}} = \frac{\partial D(e)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i^*)}{\partial x_{ij}} > 0$.[8] Therefore in the competitive market farmers use more inputs compared to the social optimum and thus over-pollute ($e_i^* < e_i^o$), since

[8] The same stands for the corresponding FOCs under uncertainty.

they do not take into account the external effects of their production choices.

The first fundamental welfare theorem, which implies that the market economy leads to a Pareto optimal result, is violated. In the absence of regulatory intervention the competitive market fails to induce farmers to operate in a way that would result in the socially optimal environmental pollution, leading to sub-optimal equilibrium. This is a market failure calling for regulatory intervention to bring competitive equilibrium closer to the social optimum without impeding agents' maximizing behavior. In this context Pillar II measures can be a regarded as regulation aiming at correcting this market failure.

5.2 Dynamic Non-Point-Source Pollution Model

In the previous model of NPS pollution the environmental costs or damages stemmed from the generation of ambient nitrate emissions e. Nevertheless, social damages do not only result from the flow of nitrate residuals on the field activity space but also by the accumulated stock of nitrate emissions in the natural resource located into the geographical region. The impact of nitrate leaching on water resources is a stock variable (Legras, 2004) and the element of time appears. Therefore the NPS pollution problem can be defined in a dynamic context, based on the model developed by Xepapadeas (1992).

In this section the specification of the farmer's payoff function remains quite the same as in the static NPS pollution model, while the environmental costs or damages are now defined as $D(S)$, an increasing and convex function of accumulated stock of nitrate emissions in the resource.

5.2.1 The Model under Certainty

If the rate of ambient accumulation of nitrate emission produced by farming activities exceeds the rate of natural cleaning ability then it is the stock of the pollutant (S) that is built into the environment (Xepapadeas, 1997). The pollution accumulation is described by the differential equation:

$$\dot{S} = e(t) - b\left(S(t)\right) \tag{5.17}$$

where $e(t)$ are the collective emissions generated each period t and $b\left(S(t)\right)$ defines the amount of pollution removed through natural processes. The later term can be specified as $bS(t)$, where b is a constant exponential natural pollution decay rate. Therefore (5.17) is rewritten as:

$$\dot{S} = \sum_{i=1}^{n} e_i(t) - bS(t) \tag{5.18}$$

Based on Huhtala and Laukkanen (2004) the accumulation of nitrates can also take the form $\dot{S} = (1 - \gamma(t))e(t) - bS(t)$, where $\gamma(t)$ denotes the rate of abatement undertaken by farmers. Moreover, by getting more specific if the pollutant accumulates into ecosystems such as shallow lakes then (5.18) is augmented by an additional term representing internal feedback loading of the pollutant into the ambient environment. In such cases when the pollution stock becomes too high, it sets off an internal positive feedback mechanism which impairs the ecosystem's ability to absorb and biodegrade loadings (Brock & Starrett, 1999). Based on Mäler & all (2003) the feedback term can be specified as $s\frac{S^2(t)}{S^2(t)+m^2}$ where s denotes the maximum rate of internal loading and m the anoxic level.

Regulator: The regulator pursues the maximum present value of social welfare by defining the optimal path of the vector $\mathbf{x}_i^*$ for each nonpoint farmer i, subject to a transition equation. Therefore the regulator's problem is:[9]

$$\begin{aligned} &\max_{\mathbf{x}_i} \int_0^{\infty} \exp(-\rho t) \left\{ \sum_{i=1}^{n} \pi_i(\mathbf{x}_i) - D(S) \right\} dt \\ &\text{s.t.} \quad \dot{S} = \sum_{i=1}^{n} e_i(\mathbf{x}_i) - bS \end{aligned}$$

[9] The objective function can be altenatively defined solely in terms of cost as: $\max_{\mathbf{x}_i} \int_0^{\infty} \exp(-\rho t) \left\{ -\sum_{i=1}^{n} C_i(\mathbf{x}_i^a) - D(S) \right\} dt$.

where ρ denotes the regulator's discount rate. In this maximization problem $\mathbf{x}_i$ is the control variable and S is the state variable.

To solve the problem the current value Hamiltonian is defined as:

$$\mathcal{H}(e, \mathbf{x}_i, \mu) = \sum_{i=1}^{n} \pi_i(\mathbf{x}_i) - D(S) + \mu \left(\sum_{i=1}^{n} e_i(\mathbf{x}_i) - bS \right) \tag{5.19}$$

where $\mu(t)$ is the costate variable and furthermore the dynamic social shadow cost of the stock of pollution S. Particularly it holds $\frac{\partial W^*}{\partial S} = \mu$, defining how the value function[10] W^* is affected by variations in the stock of the pollutant S. Furthermore $\mu < 0$ since the stock of pollution generates damages.

The necessary conditions for optimality, as determined by the Pontryagin principle, are:

$$\frac{\partial H(e, \mathbf{x}_i, \mu)}{\partial x_{ij}} = \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} + \mu \left(\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \right) = 0 \qquad \forall i, j \tag{5.20a}$$

$$\dot{\mu} = \rho\mu - \frac{\partial H(e, \mathbf{x}_i, \mu)}{\partial S} = (\rho + b)\mu + \frac{\partial D(S)}{\partial S} \tag{5.20b}$$

$$\dot{S} = \sum_{i=1}^{n} e_i(\mathbf{x}_i) - bS \tag{5.20c}$$

$$\lim_{t \to \infty} \exp(-\rho t)\mu(t)S(t) = 0 \tag{5.20d}$$

The later condition is the Arrow type transversality condition at infinite.[11] According to (5.20a) the socially optimal production choices $\mathbf{x}_i^*$ must equate each farmers marginal benefits with the marginal damage realized from an increase in input use by a small amount. These conditions define the short-run demand functions for production choices as a function of social shadow cost of pollutant stock: $\mathbf{x}_i^* = \mathbf{x}_i(\mu)$.

The regulator's problem can be solved either in the "*control-state*" space defining the socially optimum long-run equilibrium $(\mathbf{x}_i^*, S^*)$ for

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[10] The value function is defined as the maximum achieved social welfare: $W^* = \max_{\mathbf{x}_i} \int_0^\infty \exp(-\rho t) \left\{ \sum_{i=1}^{n} \pi_i(\mathbf{x}_i) - D(S) \right\} dt$.

[11] Under certain assumptions these conditions can also be considered as a necessary condition.

the productive choices vector and the pollutant accumulation, or in the "*state-costate*" space defining the optimal long-run equilibrium (S^*, μ^*) for the pollutant accumulation and the associated social cost. The later equilibrium is defined from the solution of the modified Hamiltonian dynamic system: $\dot{\mu} = 0$ and $\dot{S} = 0$, using conditions (5.20c) and (5.20b).[12]

Farmer: In an unregulated competitive market the farmer wishes to maximize the present value of net benefits:

$$\max_{\mathbf{x}_i} \int_0^\infty \exp(-rt) \{\pi_i(\mathbf{x}_i)\} \, dt \tag{5.21}$$

where r is the discount rate assumed to be common for all polluting agents, in general $r \neq \rho$ (Xepapadeas, 1992). Profit maximizing input vector $\mathbf{x}_i^o$ is chosen such as to satisfy the first-order-condition: $\frac{\partial \pi_i(\mathbf{x}_i^o)}{\partial x_{ij}} = 0$ and the associated long-run stock of pollutant is $S^o = \left(\sum_{i=1}^n e_i\right)/b$, obtained by setting $\dot{S} = 0$. It is notable that the objective function is independent of pollution stock S, implying that the input vector $\mathbf{x}_i^o$ and the corresponding individual emission level e_i^o are both independent of the pollutant's shadow cost (Xepapadeas, 1997). Such a behavior rule is called myopic, since agents systematically ignore the evolution of pollution stock and treat it as fixed $(\bar{S})$, defining input choices according to the rule: $\mathbf{x}_i = \mathbf{x}_i(\bar{S}, t)$ (Xepapadeas, 2005). Actually myopic farmers face a static problem defined by (5.3).

- Strategic Interaction: Open-Loop and Feedback Informational Structures

The previous definition of the dynamic NPS pollution problem relayed on the assumption that agents' payoff is independent of the pollution stock. Nevertheless in practice the stock of pollutants - such as nitrate leaching or salinity - could affect negatively the production of the agricultural product (Xepapadeas, 1997). Hence by relaxing this assumption the production function and consequently farmers' net benefits are redefined as:

$$y_i = f(\mathbf{x}_i, S) \quad \text{and} \quad \pi_i(\mathbf{x}_i, S)$$

[12] The short-run demand functions $\mathbf{x}_i^* = \mathbf{x}_i(\mu)$ have been substituted.into (5.20b)

where $\frac{\partial f(\mathbf{x}_i,S)}{\partial S}, \frac{\partial \pi_i(\mathbf{x}_i,S)}{\partial S} < 0$ hold respectively.

Finally, for convenience both the production and net benefit function can be also represented by a separable function of the form: $y_i = f(\mathbf{x}_i) - d_i(S)$ and $\pi_i(\mathbf{x}_i, S) = \pi_i(\mathbf{x}_i) - d_i(S)$, where $d_i(S)$ represents individual production damages resulting from pollution stock.

Regulator: The social welfare function is given as $\sum_{i=1}^{n} \pi_i(\mathbf{x}_i, S) - D(S)$, containing both general environmental and production damages. The Pontryagin conditions are given by:[13]

$$\begin{aligned} &\frac{\partial \pi_i(\mathbf{x}_i, S)}{\partial x_{ij}} + \mu \left(\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \right) = 0 \qquad \forall i, j \\ &\dot{\mu} = (\rho + b)\mu + \frac{\partial D(S)}{\partial S} - \sum_{i=1}^{n} \frac{\partial \pi_i(\mathbf{x}_i, S)}{\partial S} \\ &\dot{S} = \sum_{i=1}^{n} e_i(\mathbf{x}_i) - bS \end{aligned} \tag{5.22a}$$

After comparing (5.20b) with (5.22a) it is obvious that the resulting social shadow cost is higher than the one under the assumption $\frac{\partial \pi_i(\mathbf{x}_i)}{\partial S} = 0$, denoting lower long-run pollutant accumulation level when pollution dynamics enter agent's problem directly through the production function.

Farmer: Under this context unregulated agents can keep ignoring the evolution of pollution stock into their profit maximizing decisions and follow a myopic behavioral rule (defined above),or take (5.18) into account by using either an open-loop (OL) or feedback (FB) informational structure.

Under an OL information context farmers commit themselves to a particular input path (emission path) at the beginning of the game by taking into account only the initial conditions $(S(t_o), t_o)$ and ignoring thereafter the observed changes into the pollution stock. Input choices follow the rule: $\mathbf{x}_i = \mathbf{x}_i(S(t_o), t)$ and each agent i chooses the input path $\mathbf{x}_i$ that maximizes the present value net benefits by treating the input path (emission path) of the other agents as fixed at the best response. Therefore the maximization problem faced by

[13] According to Xepapadeas (1997) one could assume that $\frac{\partial D(S)}{\partial S} = 0$ in order to facilitate analysis without affecting the results.

farmer i is:

$$\max_{\mathbf{x}_i} \int_0^{\infty} \exp(-rt)\left\{\pi_i(\mathbf{x}_i, S)\right\} dt$$

$$s.t. \quad \dot{S} = e_i(\mathbf{x}_i) + \sum_{j \neq i}^{n-1} e_j(\bar{\mathbf{x}}_j) - bS \tag{5.23}$$

The solution of this problem defines the profit maximizing input vector $\mathbf{x}_i^{OL}$ and the long-run stock of pollutant S^{OL}. Moreover the private shadow cost of the pollution stock μ_i^{OL} for the agent i is given by the associated Pontryagin condition:

$$\dot{\mu}_i^{OL} = (\rho + b)\mu_i^{OL} - \frac{\partial \pi_i(\mathbf{x}_i, S)}{\partial S}$$

where each farmer takes into account only the adverse effects of his own input choices (emissions)and ignores the effects of its own pollution on the rest of the farmers.[14]

On the other hand, under a FB information context each profit maximizing agent takes into account the current state of the system, adopting a feedback rule: $\mathbf{x}_i = \mathbf{x}_i(S(t), t)$. Assuming a linear feedback rule, each farmer i perceives that one part of other farmers' input vector (and emissions) are autonomous while the remaining is dependent on the pollution stock. Therefore the input and emission strategy of the agents is given by:

$$\mathbf{x}_j = \bar{\mathbf{x}}_j + \mathbf{x}_j(S) \quad \text{and} \quad e_j(\mathbf{x}_j) = e_j(\bar{\mathbf{x}}_j) + e_j(\mathbf{x}_j(S))$$

where $\frac{\partial \mathbf{x}_j(S)}{\partial S} < 0$ and consequently $\frac{\partial e_j(\mathbf{x}_j)}{\partial S} < 0$, implying that each farmer i expects other farmers to reduce input usage and thus emissions production if the pollution stock increases (Xepapadeas, 1992). Thus each agent i faces the following problem:

$$\max_{\mathbf{x}_i} \int_0^{\infty} \exp(-rt)\left\{\pi_i(\mathbf{x}_i(S), S)\right\} dt$$

$$s.t. \quad \dot{S} = e_i(\mathbf{x}_i) + \sum_{j \neq i}^{n-1} \left[e_j(\bar{\mathbf{x}}_j) + e_j(\mathbf{x}_j(S))\right] - bS \tag{5.24}$$

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[14] Actually, this is a partial internalization of environmental damages.

defining the profit maximizing input vector $\mathbf{x}_i^{FB}$ and the long-run pollution stock S^{FB}. The associated private shadow cost of pollution stock μ_i^{FB} is given by:

$$\dot{\mu}_i^{FB} = (\rho + b - \sum_{j \neq i}^{n-1} \frac{\partial e_j}{\partial \mathbf{x}_j} \frac{\partial \mathbf{x}_j}{\partial S}) \mu_i^{FB} - \frac{\partial \pi_i(\mathbf{x}_i)}{\partial \mathbf{x}_j} \frac{\partial \mathbf{x}_j}{\partial S} - \frac{\partial \pi_i(\mathbf{x}_i)}{\partial S}$$

It can be shown that under certain assumptions the relationship between the social and private shadow costs of pollutant in the unregulated case is given by:

$$\mu^* > \mu_i^{OL} > \mu_i^{FB}$$

implying that in the absence of regulation unregulated agents undervalue pollution stock compared to the social optimum. Furthermore, the private shadow cost μ_i^{FB} under the FB rule is less than the associated μ_i^{OL} under the OL rule, since under a FB information structure each agent believes that the effects of his own extra emissions will be partly offset by the other agents' lower emissions. This provides an incentive to apply more inputs and thus overemit, justifying the inequality: $\mathbf{x}_i^o > \mathbf{x}_i^{FB} > \mathbf{x}_i^{OL} > \mathbf{x}_i^*$ and $S^o > S^{FB} > S^{OL} > S^*$.

Finally, it is important to mention that the OL equilibrium does not necessarily constitute equilibrium under different initial conditions $(S(t_o), t_o)$. Thus if a small deviation from the optimal path is realized, then the policy that was initially optimal may not necessarily be optimal for the remaining part of the game. On the other hand the defined FB equilibrium is subgame perfect equilibrium for any initial condition.

5.2.2 The Model under Uncertainty

Based on Xepapadeas (1992) uncertainty is introduced into the pollutant concentration model (5.18) through an additional term reflecting random natural decay rate. Therefore the evolution of the pollution stock $S(t)$ is described by an stochastic differential equation of the Itô's type:

$$dS(t) = \left[\sum_{i=1}^{n} e_i(t) - bS(t) \right] dt + \omega(S(t)) d\xi \qquad (5.25)$$

where $\xi(t)$ is a stochastic process, known as Wiener process or Brownian motion. In this case $(-bS)$ is the mean of the pollution removal process and $\omega^2(S) = \sigma S$ the instantaneous variance with $0 < \sigma < b$.

Profits and damages could also be made stochastic by including stochastic disturbances θ and η, implying functional forms like: $\pi_i(\mathbf{x}_i, \theta)$ and $D(S, \eta)$. However in this case the problem becomes more complicated since additional assumptions are needed regarding the evolution of θ and η as well as the correlation between θ, η and ξ. This is an area for further research.

Regulator: In a stochastic context the regulator wishes to maximize the expected social welfare subject to the stochastic differential equation (5.25). Therefore the regulator's problem is:

$$\max_{\mathbf{x}_i} E \int_0^\infty \exp(-\rho t) \left\{ \sum_{i=1}^n \pi_i(\mathbf{x}_i) - D(S) \right\} dt$$

$$\text{s.t.} \quad dS(t) = \left[\sum_{i=1}^n e_i(t) - bS(t) \right] dt + \omega(S(t)) d\xi$$

To solve the problem the generalized current value Hamiltonian is formed as:

$$\mathcal{H} = \sum_{i=1}^n \pi_i(\mathbf{x}_i) - D(S) + \mu \left(\sum_{i=1}^n e_i(\mathbf{x}_i) - bS \right) + \frac{1}{2} \mu_S (\sigma S)$$

where $\mu_S = \frac{\partial^2 W^*}{\partial S^2} < 0$ reflecting the regulator's risk aversion that can be also defined as $\mu_S = \frac{\partial \mu}{\partial S}$. In this case μ represents the expected social shadow cost of the pollutant's stock.

Under the assumption that the value function W^* is quadratic the optimality conditions imply:[15]

[15] This means that the problem can be written in a linear quadratic structure.

$$\frac{\partial H}{\partial x_{ij}} : \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} + \mu \left(\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \right) = 0 \qquad \forall i, j \tag{5.26a}$$

$$d\mu = \left(\rho\mu - \frac{\partial H}{\partial S} \right) dt + \omega(S) \frac{\partial \mu}{\partial S} d\xi$$

$$= \left[(\rho + b)\mu + \frac{\partial D(S)}{\partial S} - \frac{1}{2}\mu_S \sigma \right] dt + \omega(S)\mu_S d\xi \tag{5.26b}$$

$$dS(t) = \left[\sum_{i=1}^{n} e_i(t) - bS(t) \right] dt + \omega(S(t)) d\xi \tag{5.26c}$$

$$\lim_{t \to \infty} \exp(-\rho t) E \{\mu(t) S(t)\} = 0 \tag{5.26d}$$

The stochastic system of (5.26b) and (5.26c) provides the long-run equilibrium $(\mathcal{E}(\mathcal{S}^*), \breve{\mu}^*)$ for the expected stock of pollutant and the associated expected social cost of the pollutant stock. Based on Xepapadeas (1992) the stochastic equilibrium set $(\mathcal{E}(\mathcal{S}^*), \breve{\mu}^*)$ implies lower long-run pollutant accumulation level than the model under certainty due to regulator's risk aversion, as it is reflected by the term $(\frac{1}{2}\mu_S \sigma)$. Indeed risk aversion results in higher social shadow cost of pollutant accumulation level $\breve{\mu}^*$ compared to the level μ^* defined under certainty.

Farmer: In an unregulated competitive market the risk-neutral farmer i maximizes the present value of expected private net benefits. The farmer's problem is:

$$\max_{\mathbf{x}_i} \int_0^\infty \exp(-rt) \{\pi_i(\mathbf{x}_i)\} dt$$

Profit maximizing input vector $\breve{\mathbf{x}}_i^o$ for each nonpoint farmer are chosen such as to satisfy the nm first-order-conditions $\frac{\partial \pi_i(\mathbf{x}_i, \theta)}{\partial x_{ij}} = 0$. According to Xepapadeas (1992) in the unregulated competitive market the expected long-run pollutant concentration $\mathcal{E}(S^o)$ is equal to the corresponding level S^o defined under certainty context, implying that $\mathcal{E}(S^o) = S^o$, since farmers ignore stochastic factors associated with pollution accumulation.

5.2.3 Market failure

After comparing the first-order-conditions resulting from the regulator's and farmer's problem in the context of certainty and uncertainty respectively, it is evident that there is deviation from the socially optimal level of the stock of pollutant $\mathcal{S}^*$ (or $\mathcal{E}(S^*)$ under uncertainty) and the optimum level of stock of pollutant S^o (or $\mathcal{E}(S^o)$ under uncertainty) under an unregulated competitive market. Particularly the inequalities $\mathcal{S}^* < S^o$ and $\mathcal{E}(\mathcal{S}^*) < \mathcal{E}(S^o)$ hold respectively. Therefore it is evident that in the competitive market each farmer uses more inputs compared to the social optimum and thus over-pollutes ($e_i^* < e_i^o$ or $\mathcal{E}(e_i^*) < \mathcal{E}(e_i^o)$ respectively), leading to higher nitrate pollutant accumulation into the ambient environment. However, it is important to mention that the following general inequality holds: $\mathcal{E}(S^o) = S^o > \mathcal{S}^* > \mathcal{E}(\mathcal{S}^*)$, implying that under uncertainty the market failure is higher resulting in $\mathcal{E}(S^o) - \mathcal{E}(\mathcal{S}^*) > S^o - \mathcal{S}^*$ (Xepapadeas, 1992).

5.3 Non-point-Source Pollution Problem under Agents Heterogeneity

Polluting agents in the region are assumed to be heterogeneous and can be distinguished by their type or distinctive characteristics, represented by a scalar parameter Θ_i.[16] The heterogeneity characteristic can embody the farmer's ability, proximity to a receiving body, soil composition (Xepapadeas, 1997), soil retention capacity. It can either be interpreted as the costs of abatement (Spulber, 1988), or the efficiency in pollution generation reflected by an index of efficiency in input use $\mathbf{x}_i$ (Millock et al., 2002).[17] A continuum of agents with regard to a scalar parameter Θ can be considered, where all farmers belong to a certain interval $\left[\underline{\Theta}_i, \bar{\Theta}_i\right]$. The a priori beliefs of the regulator and agents about the distribution of the Θ_i parameters may be represented by the cumulative distribution functions $H_i(\Theta_i)$ which are common knowledge (Spulber, 1988) and a known, continuous, strictly positive density function $h(\Theta_i)$ (Millock et al.,

[16] This is basically an adverse selection case.

[17] In the case of mobile NP sources the parameter Θ_i can represent for instance the energy efficiency of a car (Millock et al., 2002).

2002) which is equal to $h(\Theta_i) = \frac{dH_i(\Theta_i)}{d\Theta_i} > 0$. The regulator takes the parameter Θ_i to be independently but not necessarily identical distributed, while each agents beliefs about the other polluters characteristics are independent of their own type (Spulber, 1988). Finally let $H(\Theta_i) = \prod_{i=1}^{n} H_i(\Theta_i)$ be the distribution function and $d\left(H(\Theta_i)\right) = \prod_{i=1}^{n} d\left(H_i(\Theta_i)\right)$ the density.

Farmer: In Spulber (1988) the parameter Θ_i is incorporated into the cost function that is defined as $C(\mathbf{x}_i, \Theta_i)$, with $\frac{\partial C(\mathbf{x}_i,\Theta_i)}{\partial \Theta_i} > 0$ implying that the costs of reducing pollution increase with the technology parameter Θ_i. In (Millock et al., 2002) the production function is specified as: $y_i = f(\mathbf{x}_i, \Theta_i)$, with $\frac{\partial^2 f(\mathbf{x}_i,\Theta_i)}{\partial \mathbf{x}_i \partial \Theta_i} > 0$ implying that with higher Θ_i the maximum yields can be achieved with fewer units of inputs - higher quality soils utilize chemicals better. Therefore the farmer i's profit function $\pi_i(\mathbf{x}_i(\Theta_i), \Theta_i)$ can alternatively be defined as:

$$pf(\mathbf{x}_i) - C(\mathbf{x}_i(\Theta_i), \Theta_i) \quad \text{or} \quad pf(\mathbf{x}_i(\Theta_i), \Theta_i) - \mathbf{w}_j \mathbf{x}_i(\Theta_i)$$

where the associated first-order-conditions are similar to the previous definitions (i.e. $\frac{\partial \pi_i(\mathbf{x}_i(\Theta_i),\Theta_i)}{\partial \Theta_i} \frac{\partial e_i(\mathbf{x}_i(\Theta_i),\Theta_i)}{\partial x_{ij}} = 0$).

Under this context the pollution function is a stochastic function of input choices $\mathbf{x}_i$ and the type Θ_i of the discharger, represented by:

$$e_i = e_i(\mathbf{x}_i(\Theta_i), \Theta_i) = e_i(\mathbf{x}_i(\Theta_i)) - \Theta_i - \varepsilon$$

where ε is a random variable with zero mean reflecting observation errors of individual emissions (Xepapadeas, 1997). It is plausible that a higher Θ_i indicates that the farmer is more efficient in pollution generation, emitting less than low Θ_i farmers for given input level (Xepapadeas, 1997), implying a negative relation between Θ_i and e_i.

Nonetheless, it is worth mentioning that if the same quantity Θ_i is used in both the production and pollution function then the correlation between the heterogeneity parameter Θ_i and individual discharges e_i can be also positive. According to Millock et al. (2002) if pollution is an inevitable effect of production then the output externality case $e_i = e_i(\Theta_i \mathbf{x}_i)$ is relevant and it holds $\frac{\partial e_i(\mathbf{x}_i,\Theta_i)}{\partial \Theta_i} > 0$, indicating a positive correlation. On the other hand, if unutilized inputs results in pollution then the residue externality case $e_i = (1 - \Theta_i)\mathbf{x}_i$ is relevant and it holds $\frac{\partial e_i(\mathbf{x}_i,\Theta_i)}{\partial \Theta_i} < 0$, indicating a negative correlation between Θ_i and e_i, supporting the initial argument.[18]

Therefore aggregate pollution is defined as:

$$e = \int_{\underline{\Theta}_i}^{\bar{\Theta}_i} \left[e_i(\mathbf{x}_i(\Theta_i), \Theta_i) \right] h(\Theta_i) d\Theta_i$$

Regulator: Based on Xepapadeas (1997) the regulator's welfare measure is modified to incorporate the welfare value of regulatory intervention, defined as $\tilde{\lambda} T_i(\Theta_i)$ with $T_i(\Theta_i)$ the payment under the regulation (i.e. tax revenues) and holds $\tilde{\lambda} > 0$ since revenues resulting under the regulation are used to reduce distortionary measures (i.e. taxes) and thus raise the welfare gains. Therefore the expected net social benefit is:[19]

$$NSB = \int_{\underline{\Theta}_i}^{\bar{\Theta}_i} \left\{ \pi_i(\mathbf{x}_i(\Theta_i)) - D(e) + \tilde{\lambda} T_i(\Theta_i) \right\} h(\Theta_i) d\Theta_i \tag{5.27}$$

where according to Spulber (1988) $\tilde{\lambda}$ is the multiplier associated with participation constraint $\pi_i(\Theta_i) = \pi_i(x_i(\Theta_i)) - T_i(\Theta_i) \geq 0$. It is notable that the profits after the regulation $\pi_i(\Theta_i)$ are nonnegative since the regulator does not wish to drive farmers out of production (Xepapadeas, 1997). After some manipulations the welfare measure can be rewritten as:[20]

$$NSB = \int_{\underline{\Theta}_i}^{\bar{\Theta}_i} \left\{ (1 + \tilde{\lambda}) \pi_i(\mathbf{x}_i(\Theta_i)) - D(e) - \tilde{\lambda} \pi_i(\Theta_i) \right\} h(\Theta_i) d\Theta_i \tag{5.28}$$

The associated mn first-order-conditions with respect to x_{ij} are:

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[18] According to Millock et al., (2002) the derivative: $\frac{de_i(\mathbf{x}_i(\Theta_i))}{d\Theta_i} = \frac{\partial e_i(\mathbf{x}_i(\Theta_i))}{\partial \Theta_i} + \frac{\partial e_i(\mathbf{x}_i(\Theta_i))}{\partial \mathbf{x}_i} \frac{\partial \mathbf{x}_i(\Theta_i)}{\partial \Theta_i}$ determines how pollution varies with Θ_i.

[19] Based on Millock et al., (2002) the social damage can be further specified as $D(e) = D\left(\int_{\underline{\Theta}_i}^{\bar{\Theta}_i} \left[e_i(\mathbf{x}_i(\Theta_i), \Theta_i) \right] h(\Theta_i) d\Theta_i \right)$.

[20] The regulator's objective in Spulber (1988) is defined quite differently as: $NSB = \int_{I^n} \left\{ (1 + \tilde{\lambda}) \left[S(\sum_{i=1}^n y_i) - C_i(\mathbf{x}_i, \Theta_i) - D(e) \right] - \tilde{\lambda} \sum_{i=1}^n R_i(e_i) \right\} h(\Theta_i) d\Theta_i$, where $S(\sum_{i=1}^n y_i)$ is defined as the lump-sum transfers from agents to consumers through effluent taxes, and $\sum_{i=1}^n R_i(e_i)$ are the total ex ante returns to private information, that is, the costs incurred by the regulator to induce truthtelling.

$$FOC_{x_{ij}} : (1+\tilde{\lambda}) \left\{ \frac{\partial \pi_i(\mathbf{x}_i(\Theta_i))}{\partial x_{ij}} - \left[\frac{\partial D(e)}{\partial e} \frac{\partial e(\cdot)}{\partial e_i} \frac{\partial e_i(\cdot)}{\partial x_{ij}} \right] \right\} h(\Theta_i) = 0 \tag{5.29}$$

It is important to redefine the regulator's problem under incomplete information, where the regulator does not know the polluters' heterogeneity parameter Θ_i and individual emissions e_i. The type Θ_i is the adverse selection **p**arameter since it is private information known only by the farmers, while individual emissions are characterized as the moral hazard parameter since they cannot be inferred from the observed input use (Xepapadeas, 1999). In this context the regulator's distribution function $H(\Theta_i)$ satisfies the monotonous hazard rate assumption: $\frac{d}{d\Theta_i}\left(\frac{H(\Theta_i)}{h(\Theta_i)}\right) \geq 0$. Such a pollution problem can be addressed through revelation mechanisms, that is equivalent to any regulation mechanism under the revelation principle (Xepapadeas, 1997), given that agents pursue truth-telling strategies.

6

Instruments for Designing Non-Point-Source Pollution Control Policies

NPS pollution is a major source of environmental quality problems in developed countries which have advanced pollution policies and a growing cause of environmental degradation in developed countries (Shortle et al., 1998; Horan et al., 2002). Even though there is a substantial agreement that more aggressive NPS control policy is needed, there is less agreement about the kinds of actions that represent good policy (Shortle et al., 1998).

However, public concern about adverse impacts of agricultural production practices has drawn attention towards policies for environmental improvements in the area of NPS pollution. In general the environmental policy schemes discussed in this chapter are:

- **Effluent-based schemes.** Even though measures based on individual emissions are not widespread in practice, they are extensively presented in this survey since they form the theoretical foundation of the most applied measures of environmental policies. Such instruments include linear emission charges or emission-reduction subsidies, as well as performance or design standards, that can be uniformly or nonuniformly applied. Under imperfect monitoring such Pigouvian taxes are imposed either on a fraction of farmers after random inspections or on all farmers based on imprecise estimates of individual emissions, or on information provided by polluting agents themselves. Moreover, emission-based instruments are considered under an asymmetric information and dynamic context, where in the later case time flexible and steady-state Pigouvian tax rates are defined respectively under alternative behavioral rules.
- **Input-based schemes.** If individual discharges are not directly observed, NPS instruments can be based on observed production choices. Such measures include charges or restrictions on inputs that increase a detrimental externality and / or

subsidies on inputs that reduce it. They can be applied nonuniformly or uniformly on a subset of inputs or even on a single input. Moreover, nonuniform policy schemes can be imposed broadly on all farmers or target certain categories of polluters within a geographical region. Under uncertainty input policy scheme must be modified to account of the substitution effects of taxed and untaxed inputs on expected damages, as well as to account of the input and output market price effects of the NPS regulations. Finally, input-based schemes are also defined in an asymmetric information and dynamic framework, where a dynamic input-based scheme can be also designed both under a linear Markov perfect tax rule and in the context of quantity-quality problems.

- **Output-based schemes.** Even though output taxes are considered to have a Pigouvian role, this survey does not focus on the theoretical developments regarding such instruments, since they are known to be inefficient in the long-run.

- **Ambient-based schemes.** In reality neither individual discharges nor individual productive choices are directly observed, then policy schemes are based on ambient pollution. Such policy schemes can be either imposed collectively to all farmers within a geographical region or randomly on one or more farmers. They involve either uniform or farm-specific fines (subsidies) and can be formulated as budget or non-budget balancing. Linear and nonlinear, state-dependent schemes, as well as a variance-based instrument can be also considered. Ambient-based schemes can further defined under heterogeneous expectations between the regulator and farmers, under asymmetric information, and in a dynamic context. This category also includes NPS instruments that consist of a combination of collective ambient taxes and individual emission charges, known as mixed-based schemes.

In the following we present the NPS instruments as they have been developed in the literature. It is worth noting that the analysis these types of instruments can help to obtain insights in actual policy design, an issue very important for the design of efficient agri-environmental policies in the EU.

6.1 Emission-based Schemes

If individual emissions are observable and the regulator is capable to calculate the ex ante distribution of emissions as accurately as the polluter through the observed vector of agent's inputs (Hansen, 2002) then effluent or emission-based policies can be applied to handle the pollution problem. For the case of agricultural NPS pollution an emission tax is in principle imposed on nitrate leaching generated by farms. Although these type of measures are not widespread in practice, in this section we presented the developments on emission or effluent-based measures in the context of certainty and uncertainty, as well as in a static and dynamic framework, since they form the foundations of environmental policies as well as the theoretical foundation of the most applied measures.

6.1.1 Under certainty

In a setting of complete certainty about the source and the individual contribution of each individual polluter to the ambient pollution level and complete information about the functions and every parameter of the problem (Cochard, 2003), the pollution problem is definitely a point-source problem. Notable examples are the Swedish NO_x charge on heat and power producers, and the Japanese SO_x charge, which are based on metered emissions. (OECD, 1994).

- *Static context:*

In this case the pollution problem can be dealt through Pigouvian taxes, that are charges per unit of individual emissions. These charges can be linear to individual emissions, with a tax rate t_i. The total tax payments are $T_i(e_i) = t_i e_i$ and thus the after-tax objective function of farmer i is: $\pi_i(\mathbf{x}_i) - T_i(e_i)$. The associated nm first-order-conditions are:

$$FOC_{x_{ij}} : \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} - t_i \frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} = 0 \qquad \forall i, j$$

The comparison of the above $FOCs$ with the FOC (5.7), corresponding to the social optimum, implies that the emission tax rate t_i must be set equal to

$$t_i = \left[\frac{\partial D(e)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right] \Big/ \left[\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right] \qquad \forall i, j \tag{6.1}$$

to reproduce the socially optimal discharge even if the farmer i uses more than one input ($j > 1$) since in this context either the term $\frac{\partial D(e)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}$ or $\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$ is deterministic (Cochard, 2003).

The regulator can also provide linear subsidies for reducing individual emissions below an individual target $\bar{e}_i$. In this case the subsidy scheme is $S_i(e_i) = \kappa_i(\bar{e}_i - e_i)$ and the after-subsidy objective function is redefined as $\pi_i(\mathbf{x}_i) + S_i(e_i)$. The subsidy rate κ_i delivering the social optimum is identified with the ratio (6.1) defined previously for the tax rate.

The regulator can not only handle the NPS pollution problem through emission-based, fiscal instruments but also through command and control regulation, such as performance standards. Particularly an emission limit $\tilde{e}_i$ can be imposed on each farmer, leaving polluters the maximum freedom to comply with the standard by either reducing output or increasing abatement . The farmer's problem in this case is defined as:

$$\max_{\mathbf{x}_i} \pi_i(\mathbf{x}_i) \qquad \text{s.t. } e_i \leq \tilde{e}_i$$

In this case the associated Langrangian multiplier λ denotes the shadow cost of the emission limit - the marginal change in the agent's payoff due to an increase of the emission standard. If the emission limit $\tilde{e}_i$ is set at the welfare maximizing level then the performance standards is equivalent to the Pigouvian tax (Xepapadeas, 1997).

Nevertheless, this type of regulation requires individual monitoring and knowledge of compliance costs, which in practice can be expensive of technically infeasible justifying the use of design standards (Xepapadeas, 1997). Under this policy measure the regulator requires farmers to use a specific technology or practices, defined as $\tilde{x}_{ij}$, that can either be pollution prevention technologies that reduce the use of polluting inputs per unit of output or pollution-treatment technologies that reduce nitrate pollution (Abler and Shortle, 1995)[1].

[1] Such a design standard can imply the use of reduced tillage, establishment of buffer strips, construction of manure storage facilities, land retirement (Wu and Segerson,

Therefore in this context the farmer maximizes individual profits under the constraint: $x_{ij} \geq \tilde{x}_{ij}$. A known example of design standards are the Best Management Practices (BMPs) to reduce erosion or runoff, used by US. agricultural pollution control programs (Helfand and House, 1995).[2] Finally, even though there are private benefits from the adoption of such technologies these benefits might be insufficient to produce the correct incentives for installation of new technology, due to high costs of adoption, the irreversibility of investment and uncertainty about returns from adoption (Isik, 2004).

- *Dynamic context:*

Under certainty the regulator commits to a path of emission tax rates $\tau_i(t)$ and the total tax payment of farmer i is given by $T_i(e_i) = \tau_i(t)e_i$. The associated first-order-conditions are:

$$FOC_{x_{ij}} : \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} - \tau_i(t)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} = 0 \qquad \forall i, j$$

After comparing the upper $FOCs$ with (5.20a) it is evident that the optimal dynamic effluent tax rate is set equal to the dynamic social shadow cost of pollution stock as defined previously by (5.20b). In particular the dynamic tax rate is:

$$\tau_i(t) = -\mu(t) \tag{6.2}$$

securing that the socially optimal individual discharge and pollution stock is realized (Xepapadeas, 1992). The policy scheme stands for the subsidy rate κ_i with the reverse sign.

Under the assumption that agents' net benefits are dependent on pollution stock and that agents adopt either an open-loop or feedback behavioral rule, the optimal dynamic tax rate can be further specified as:

$$\tau_i^{OL}(t) = -\mu(t) + \mu_i^{OL}(t) \quad \text{and} \quad \tau_i^{FB}(t) = -\mu(t) + \mu_i^{FB}(t) \tag{6.3}$$

2003), heat sensors to determine soil moisture, soil erosion control to minimize surface runoff (Owen et al., 1998), drip irrigation, integrated pest management and site-specific farming (Isik, 2004).

[2] It is notable that some USA States offer reduced property taxes to farmers that adopted soil-conserving BMPs, while other states developed a mandatory program for farmers to adopt erosion-reducing management practices (Helfand and House, 1995).

The particular tax schemes bridge the gap between the social and private shadow cost of pollution stock that causes the equilibrium pollution stock in the OL (S^{OL}) and FB (S^{FB}) information structure to exceed the social optimal pollution stock level (S^*) (Mäler et al., 2003). Nevertheless, since the private cost of pollution stock is not the same across farmers these instruments must be farm-specific (Xepapadeas, 1997), implying that the regulator not only needs to know individual emissions and individual shadow cost but also which strategy is followed by agents so that to apply the appropriate tax scheme (Legras, 2004). Moreover these measures are time dependent, fact that requires continuous change of the tax rate, making the regulator's informational burden even higher. Finally, the private shadow cost of pollution stock is a function of tax rate, introducing a simultaneity problem that allows only the implicit definition of the tax rates (Xepapadeas, 1997).

Consequently since time-dependent emission taxes may not be practically feasible, the dynamic Pigouvian tax may be modified to a second best tax and set equal to the interest corresponding to the discounted flow of marginal damages (Xepapadeas, 1997). Thereupon the second-best dynamic Pigouvian tax is:

$$\tau_S = \rho \int_0^\infty \exp(-\rho t) \left\{ \frac{\partial D(S)}{\partial S} \frac{\partial S(e_i)}{\partial e_i} \frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \right\} dt \qquad (6.4)$$

Such a tax rate can either be time invariant, considering that the desired emissions are kept constant for the whole time period, or even allow for discrete changes (or shaped-in policies) over time. In the latter case the second-best tax rate can be improved over time, however this is strongly dependent on the adjustment cost necessary to introduce the changes.

According to Xepapadeas (2005) the time invariant taxes rates can also be determined by the corresponding steady state values. Hence the particular steady-state tax rates under the myopic, OL and FB behavioral rule are given as:

$$\tau_i(t) = -\mu^\infty = \frac{\partial D(S)}{\partial S} \frac{1}{(\rho + b)}$$

$$\tau_i^{OL}(t) = -\mu^\infty + \mu_i^{OL\infty}$$

$$\tau_i^{FB}(t) = -\mu^\infty + \mu_i^{FB\infty}$$

Even though both steady-state and time-flexible policy schemes approach the same saddle point, the corresponding paths under the former measures that determine the transition to the steady state are not identical to the socially-optimal time paths (Mäler et al., 2000). Moreover, the time needed to converge to the equilibrium point under the two emission schemes does not coincide.

It is worth mentioning that under a dynamic context if the regulator ignores the dynamics of the system and proceeds in the maximization of the objective function (5.14) subject to (5.10), then the imposed emission tax rate is equal to the lagrangean multiplier λ resulting in suboptimalities (Xepapadeas, 1992).

Summarizing it is important to mention that under the subsidy scheme the number of active farmers is higher than under the Pigouvian tax - even though in this survey the set of farmers n is considered to be the optimal set. If the set of farmers n was not assumed to be the optimum set then the application of a subsidy scheme would not only increase the size of the agricultural sector but potentially motivate the increase in the ambient nitrate pollution e in the long-run. According to Xepapadeas (1997) fiscal instruments provide more effective incentives in the long-run for invention, innovation and diffusion of clean technologies, since emission reductions realized through environmentally-superior technologies will reduce tax bills or increase subsidies. On the other hand no incentive for innovation or adoption of environmentally-supperior technologies once the standard has been satisfied. Finally, even though emission taxes have the potential to reduce distortionary taxes and thus raise the welfare gains, they can also discourage employment and investment by creating extra costs in the labour and capital markets.

6.1.2 Under uncertainty

- *Static context:*

Under complete information about the functions and every parameter of the problem (Cochard, 2003) the expected total tax payments of farmer i are $T_i(e_i) = t_i E\{e_i\}$. This Pigouvian tax can not only be linear but also nonlinear or piecewise linear (Schmutzler and Goulder, 1997) or state-dependent (Cochard, 2003). Under a linear tax rate the associated expected after-tax payoff function is $E_i\{\pi_i(\mathbf{x}_i, \theta)\} - T_i(e_i)$ and the corresponding $FOCs$ are:

$$FOC_{x_{ij}} : E_i(\frac{\partial \pi_i(\mathbf{x}_i, \theta)}{\partial x_{ij}}) - t_i E\left[\frac{\partial e_i(\mathbf{x}_i, u_i, \alpha_i)}{\partial x_{ij}}\right] = 0 \qquad \forall i, j$$

The comparison with the FOC (5.15) implies that under uncertainty the tax rate t_i must be set equal to:

$$t_i = E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i, b, \omega, \beta)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i, u_i, \alpha_i)}{\partial x_{ij}}\right] \Big/ E\left[\frac{\partial e_i(\mathbf{x}_i, u_i, \alpha_i)}{\partial x_{ij}}\right] \tag{6.5}$$

This tax rate induces the socially optimal individual discharge and pollution stock level only if $j = 1$, meaning that the agricultural activity uses a single input. According to Cochard (2003) when the farmer i uses more than one input ($j > 1$) then the tax rate t_i is overdetermined, since neither the term $\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i,b,\omega,\beta)}{\partial e_i}$ or $\frac{\partial e_i(\mathbf{x}_i,u_i,\alpha_i)}{\partial x_{ij}}$ is deterministic.

These policy schemes can be imposed to all farmers in the regulated region. Nevertheless, if the monitoring costs are prohibitive then the regulator can relay on random audit scheme, where only a fraction $q \in (0, 1)$ of polluting agents in the region is monitored and thus penalized with a Pigouvian tax on their actual emissions (Cochard, 2003). The expected after-tax payoff function is defined as:$E_i\{\pi_i(\mathbf{x}_i, \theta)\} - qt_iE\{e_i\}$ and the corresponding FOC is similar to the previous definition with the only difference that the auditing probability enters the optimality conditions.

$$E_i(\frac{\partial \pi_i(\mathbf{x}_i, \theta)}{\partial x_{ij}}) - qt_i E\left[\frac{\partial e_i(\mathbf{x}_i, u_i, \alpha_i)}{\partial x_{ij}}\right] = 0 \qquad \forall i, j$$

It can be easily seen that the tax rate t_i (6.5) remains the same multiplied with the ratio $\frac{1}{q}$. According to Cochard (2003) the smaller q is the higher the tax rate t_i is, putting in danger the financial hypostasis of the operations and increasing the risks of bankruptcy. Even though greater monitoring effort amounts to a greater number of firms that are monitored (Schmutzler and Goulder, 1997) and decreases the bankruptcy risks, it might not be desirable from society's point of view to incur additional monitoring costs.

Based on Schmutzler and Goulder (1997) the random audit scheme can be reviewed as follows. In this case the basis of the emission taxes is the information provided by the individual polluting agents. This should be considered equivalent to self-reporting - that is nearly universally needed for regulation of water pollution and toxic/ hazardous chemical releases (Malik, 1993). However, since only a fraction of polluting agents is monitored and thus only their actual emissions are observed, farmers have an incentive to understate individual discharge. Even though actual emissions are e_i polluters declare less emission, say e_{di}, concealing the amount $\upsilon_i = e_i - e_{di}$. In this context pure emission taxes are not optimal and the regulator has to rely on complementary measures, such as occasionally spot-checks to verify the honesty of firms' reports (Malik, 1993) and a system of fines imposed in addition to taxes already paid if the actual with reported emissions differ. Thus these complementary measures give some incentives for truthtelling and improve welfare. In this case the total tax payment is $T_i(e_i) = t_i\,(e_i - \upsilon_i) + f(e_i, \upsilon_i, c)$, where $f(\cdot)$ the expected fine for dishonest reports given the level of monitoring effort c fines.[3] The associated tax rate is:

$$t_i = \left(E\left\{ \frac{\partial D(\cdot)}{\partial e} \frac{\partial e(\cdot)}{\partial e_i} \frac{\partial e_i(\cdot)}{\partial x_{ij}} \right\} - f(e_i, \upsilon_i, c) \frac{\partial e_i(\cdot)}{\partial x_{ij}} \right) \Big/ \frac{\partial e_i(\cdot)}{\partial x_{ij}} \qquad (6.6)$$

Even though the detection probability q is treated as exogenous given, it can also be dependent on polluter's past behavior (Cochard, 2003), as well as ambient pollution since if declared emissions are small relative to actual emissions then the monitoring effort increases and detection becomes likely (Schmutzler and Goulder, 1997).

Audit schemes are expected to be important under Pillar II policies, which require a certain type of voluntary or mandatory behavior from the farmers' point of view. It is known that the rural development regime uses aid and direct payments to improve the competitiveness of EU agriculture in a sustainable way, provided that the whole farm complies with environmental requirements such as the maximum amount of 170 kg N of livestock manure applied per hectare each year in vulnerable zones under the EU Nitrate Directive 91/676/EEC. In this context the introduced system of farm audits

[3] In Schmutzler and Goulder (1997) the problem solved by the regulator is similar to the one defined in the section (4), allowing its adaptation in the context of this survey. The author assumes that the regulator maximizes the total certainty equivalent $\pi_i(\mathbf{x}_i) - D(e) - C(c)$ taking also in mind his cost of monitoring effort $\mathcal{C}(c)$ (page 58).

involves a reduction in direct payments by 5% and 15% if the farmer fails to comply with the rules through negligence. In the event of deliberate non-compliance the reduction is at least 20%.

Similarly Cremer and Gahvari (2002) have developed a tax per unit of emissions, defined as $\theta_i = t_i(1 - \varrho_i) + f(t_i, \varrho_i, c)$, where $(1 - \varrho_i)$ is the fraction of reported discharges and $f(\cdot)$ is the expected fine per unit of emissions that depends also on statutory tax rate and the fraction of concealed emissions.[4] In this context the tax payment is $T_i(e_i) = \theta_i e_i$ and the agent i's payoff function is $E_i\{\pi_i(\mathbf{x}_i, \theta)\} - \theta_i e_i(\mathbf{x}_i)$. The random audit scheme implies some emission monitoring and fine collection costs per unit of emissions, defined as $\sum_{i=1}^{k<n} \mathcal{C}(t_i, \varrho_i, c)$ since , that need to be incorporated into the regulator's problem since they imply an additional source of cost to the society only a fraction of agents is monitored. Therefore the welfare measure is altered to $\max_{\mathbf{x}_i} \sum_{i=1}^{n} E_i\{\pi_i(\mathbf{x}_i, \theta)\} - E[D(e, \eta)] - \sum_{i=1}^{k<n} \mathcal{C}(t_i, \varrho_i, c)e_i$ and the Pigouvian tax rule (6.6) is further modified to include the monitoring cost per unit of emissions induced by changes in input choices. Thereupon the effective and statutory emission tax is:

$$t_i = \left\{ E\left(\frac{\partial D(\cdot)}{\partial e}\frac{\partial e(\cdot)}{\partial e_i}\frac{\partial e_i(\cdot)}{\partial x_{ij}}\right) + \sum_{i=1}^{k<n} \mathcal{C}(t_i, \varrho_i, c) - f(\cdot)\frac{\partial e_i(\cdot)}{\partial x_{ij}} \right\} \Big/ (1 - \varrho_i)\,\frac{\partial e_i(\cdot)}{\partial x_{ij}}$$

which leads to less emission reductions than under the perfect observability case since the emission tax falls short of the full social marginal damage of emissions due to the welfare lose associated with monitoring (Cremer and Gahvari, 2002). With imperfect observability, there are two sources of distortion: externalities and the resource costs associated with monitoring, however the emission taxes cannot correct both sources of distortion at once (Cremer and Gahvari, 2002).

Under costly and thus imperfect monitoring, the regulator can base Pigouvian taxes on imprecise estimates of individual emissions (Schmutzler and Goulder, 1997). The associated tax payment is

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[4] In Cremer and Gahvari (2002) the problem solved by the regulator is quite different compared to the ones defined in the section (4). The authors assume that the regulator maximizes the representative consumer's utility subject to his budget constraint, which includes monitoring and fine collection costs (page 394). For the purpose of this survey the regulator's problem was adapted to avoid complications.

$T_i(\hat{e}_i) = t_i\hat{e}_i$, where $\hat{e}_i = e_i + \varpi$ represents the observed signal and ϖ is the error distribution with mean zero and variance σ^2 (Schmutzler, 1996). In this case risk-aversion (r) on polluters' side is expected to behold since there is undesired unpredictability of emission tax payments and this should be viewed as a drawback of the particular input-based policy scheme. Therefore the expected benefits net of taxes are equal to $E_i\{\pi_i(\mathbf{x}_i, \theta)\} - t_i\hat{e}_i - \frac{1}{2}r\,(t_i)^2\,\sigma^2$, where $\left(\frac{1}{2}r\,(t_i)^2\,\sigma^2\right)$ is a risk term, implying that individual emission tax payments are partly random and cannot be influenced by polluting agents (Schmutzler, 1996).[5] The associated tax rate is:[6]

$$t_i = \left(-E\left\{\frac{\partial D(\cdot)}{\partial e}\frac{\partial e(\cdot)}{\partial e_i}\frac{\partial e_i(\cdot)}{\partial x_{ij}}\right\}\right) \Big/ \left(\frac{\partial e_i(\cdot)}{\partial x_{ij}} + r\sigma^2\Gamma\right)$$

According to Schmutzler (1996) if the regulatory agency takes into account the risk term into the welfare measure and $\sigma^2 > 0$ then the emission tax is lower than the marginal benefit of emission reduction and the first best cannot be achieved.

Therefore the following can be concluded:

> Under imperfect monitoring the regulator can impose Pigouvian taxes either only a fraction of farmers after random inspections or on all farmers based on imprecise estimates of individual emissions or on information provided by polluting agents themselves.

Nonetheless, the regulator can simply impose a lump sum Pigouvian tax, applied uniformly to all farmers no matter their individual polluting performance. For instance, municipal waste charges for households are generally levied at flat rates, with each household

[5] In Schmutzler (1996) the problem solved by the regulator is quite different compared to the ones defined in the previous sections. The author assumes that the regulator maximizes the total certainty equivalent subject to the incentive constraint that the farmer maximizes his certainty equivalent (page 254). The adaptation this regulator's problem is outside the scope of this study however we followed their solution concept to define the tax rate. Therefore in this case the regulator total certainty equivalent is $TCE = E_i\{\pi_i(\mathbf{x}_i, \theta)\} - E[D(e, \eta)] - \frac{1}{2}r\,(t_i)^2\,\sigma^2$ and the farmer's certainty equivalent is $CEF = E_i\{\pi_i(\mathbf{x}_i, \theta)\} - t_i\hat{e}_i$, the term $\frac{1}{2}r\,(t_i)^2\,\sigma^2$ is not included since he cannot influence it. The problem is $\max_{\mathbf{x}_i} TCE$ s.t. CEF, where the constraint is replaced by $t = E_i(\frac{\partial \pi_i(\mathbf{x}_i,\theta)}{\partial x_{ij}}) \big/ E\left(\frac{\partial e_i(\cdot)}{\partial x_{ij}}\right)$ given by the $FOC_{x_{ij}}$ of the farmers problem.

[6] Where Γ represents the term $\left(E_i(\frac{\partial \pi_i(\mathbf{x}_i,\theta)}{\partial x_{ij}})\frac{\partial^2 e_i(\cdot)}{\partial (x_{ij})^2}\right) \Big/ \left(\frac{\partial e_i(\cdot)}{\partial x_{ij}}\right)^2$.

paying a fixed sum, unconnected to the quantity of waste actually supplied (OECD, 1994). According to Millock et al. (2002) this is the case where no emissions monitoring technology is feasible. The tax payment can be specified as $T_i(e_i) = \bar{T}$ and the associated after-tax payment as $E_i\{\pi_i(\mathbf{x}_i, \theta)\} - \bar{T}$, where $\bar{T}$ is a fixed fee common for all polluters such as a per acre land tax.The advantage of this policy scheme is that no monitoring and thus no observability is required from regulator's side, while the significant drawback is that the economic viability of the less productive farmers is threatened and may be forced to stop production. The particular policy instrument decreases ambient pollution only at the extensive margin, leaving the most polluting units unaffected and thus it cannot induce the social optimum (Millock et al., 2002). Only in the special case that marginal damages are constant regardless of the source of the additional pollution a uniform effluent tax can come close to the social optimum, which however is not the case of agriculture where marginal damages vary - for instance nitrates entering a river at different places with different assimilative capacities have different impact on ambient pollution(Helfand and House, 1995).

The regulator can adopt a less strict policy, giving polluting agents the capability to face a nonuniform tax scheme based on individual emissions under the condition that they have installed monitoring equipment. The installation of monitoring technology permits the regulator to know with certainty the pollution levels of individual agents and thus differentiate taxation. In this context farmers that do not adopt the monitoring keep paying the fixed lump sum fee $\bar{T}$ while the adopters face a linear tax per unit of emissions they generate and are subsidized for any overestimate of pollution, before they installed monitoring. The tax payment of an adopter is: $T_i(e_i) = t_i e_i(\mathbf{x}_i) - \bar{T}_1$ and the payoff function is $E_i\{\pi_i(\mathbf{x}_i, \theta)\} - T_i(e_i) - \nu$, where ν are monitoring costs and $\bar{T}_1$ is either a lump-sum tax (> 0) or subsidy ($<$ 0). Such a policy scheme provides incentives to low-polluting agents to reveal themselves by adopting voluntarily monitoring equipment and according to Millock et al. (2002) this is equivalent to voluntary self-reporting of individual emissions to regulator rather than paying a fixed penalty.

The regulator can also proceed in mandatory adoption of monitoring equipment Millock et al. (2002). The installation of monitoring technology permits the regulator to know with certainty the pollution levels of each agents and thus differentiate taxation. This is actually the case of PS pollution and in this context each agent pays a Pigouvian tax $T_i(e_i) = t_i e_i(\mathbf{x}_i)$, where the tax rate shrinks as the cost of monitoring technology increases since the polluters cover these costs. However, monitoring costs can be incurred either by the regulatory agency or by the polluters themselves (Schmutzler and Goulder, 1997). Thus if agents were receiving subsidies to install monitoring technology then this would lead to more monitoring than the optimum. Consequently pollution is at lower level and the Pigouvian tax should be lower.

Summarizing, based on Millock et al. (2002) the regulator can choose among three different types of emission-based policy schemes relatively to the availability of monitoring technology:

1. No monitoring technology is available - a uniform lump sum tax is applied to all farmers.

2. Monitoring technology is mandatory - nonuniform Pigouvian taxes are imposed on all farmers.

3. Monitoring technology is voluntary - a lump-sum tax/subsidy and nonuniform Pigouvian taxes are imposed only on adopters of the monitoring technology, while the nonadopters pay a uniform lump sum tax.

It notable that the previous policy schemes implying self-reporting and monitoring - Millock et al., (2002), Cremer and Gahvari, (2002), Schmutzler and Goulder (1997) - were based on the assumption that monitoring technology is perfect and thus allows accurate revelation of discharge levels. No assumption was made about unreliable, imperfect monitoring technology which implies that the regulator may not infer correctly the existing pollution state e_i and that a firm may be erroneously fined. According to Malik (1993) a such policy scheme consists of a discharge standard $\tilde{e}_i = e_L$ coupled with a penalty on pollution generation either (t_H) if polluters report violating the standard and a penalty (t_L) for agents meeting the standard.[7] To ensure that agents report honestly their performance the regulator

[7] The regulatory authority imposes the penalty t_L also to agents that do not report their performance status, since such a behavior is considered as an indirect confession of violating the standard.

proceeds in random audits and imposes fines for dishonest reports. It is notable that only agents declaring emission level e_L are inspected, implying that $q_L > 0$ and $q_H = 0$ hold respectively for the auditing probabilities, and the expected fine $f_{L/H}$ is imposed when contrary to the report an audit indicates violation of the standard. However, imperfect monitoring implies that there is a probability ($\mathfrak{q}_{HL}$) that the regulator can erroneously fine a farmer for submitting a dishonest report, implying that even though a farmer has met the standard the regulator infers the opposite after an audit. Therefore the expected tax payment can alternatively be:

$$\begin{array}{ll} \text{If } e_i \leq e_L & T_i(e_i) = t_L + q_L \left[\mathfrak{q}_{HL} f_{L/H}\right] \\ \text{If } e_i > e_L & T_i(e_i) = t_H \quad \text{or} \quad t_L + q_L \left[\mathfrak{q}_{HH} f_{L/H}\right] \end{array}$$

where $\mathfrak{q}_{HH}$ is the probability that the regulator correctly infers the state occurred and $t_H > t_L = 0$ at the optimum. According to Malik (1993) truthful recording is achieved if the expected fine for submitting a false report ($f_{L/H}$) is strictly higher than the penalty (t_H). In the same context in the absence of self-reporting the regulator imposes a fixed penalty to agents found violating the standard. However, the tax payment depends on the accuracy of the tax monitoring technology. Thus if $e_i \leq e_L$ the expected tax payment is $T_i(e_i) = q\left\{\mathfrak{q}_{LL} t_L + \mathfrak{q}_{LH} t_H\right\}$, while if $e_i > e_L$ the expected tax payment is $T_i(e_i) = q\left\{\mathfrak{q}_{HH} t_H + \mathfrak{q}_{HL} t_L\right\}$.

- *Dynamic context:*

Under risk aversion the optimal dynamic effluent tax is equal to the expected social shadow cost of pollutant accumulation $\breve{\mu}^*$ at each instant of time that is given by:

$$\breve{\tau}_i(t) = -\breve{\mu}(t)$$

is higher than the associated dynamic tax μ^* under certainty due to the risk aversion premium $\left(\frac{1}{2}\mu_S \sigma\right)$ (Xepapadeas, 1992). Moreover the emission tax rates under the OL and FB behavioral rule are: $\breve{\tau}_i^{OL}(t) = -\breve{\mu}(t) + \breve{\mu}_i^{FB}(t)$ and $\breve{\tau}_i^{FB}(t) = -\breve{\mu}(t) + \breve{\mu}_i^{OL}(t)$ respectively. Finally, due to high informational requirements the regulator can impose a time invariant, semi-time invariant or steady-state tax rate.

6.1.3 Under asymmetric information

Assume initially that the regulator has complete information and knows the type Θ_i of each discharger. The emission tax payment is defined as $T_i(e_i(\Theta_i)) = t_i e_i(\mathbf{x}_i(\Theta_i), \Theta_i)$, where $T_i(e_i(\Theta_i))$ is such that the participation constraint $\pi_i(\Theta_i) \geq 0$ is not violated. Therefore discharger i's after-tax problem is given by $\pi_i(\mathbf{x}_i(\Theta_i)) - t_i e_i(\mathbf{x}_i(\Theta_i), \Theta_i)$ and the associated emission tax rate is equal to:

$$t_i = \left[\frac{\partial D(e)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i(\Theta_i), \Theta_i)}{\partial x_{ij}}\right] \Big/ (1+\tilde{\lambda}) \left[\frac{\partial e_i(\mathbf{x}_i(\Theta_i), \Theta_i)}{\partial x_{ij}}\right] \tag{6.7}$$

which is less than the marginal damages, leading to extra emissions. According to Xepapadeas (1997) this is an expected outcome when the welfare value of collected taxes is taken into account. This tax rate resulted from the comparison of (5.29) with the $FOC_{x_{ij}}$ resulting from the farmers problem.

Under incomplete information the regulator does not know the polluters' heterogeneity parameter Θ_i and individual emissions e_i. Such a pollution problem can be addressed through revelation mechanisms, that is equivalent to any regulation mechanism under the revelation principle (Xepapadeas, 1997), where the regulator specifies an effluent-based policy scheme $\{e_i, t_i\}$ consisting of individual effluent levels and individual taxes for each polluter based on messages received by all farmers regarding the heterogeneity cost parameter Θ_i (Spulber, 1988). Therefore the expected effluent level and tax payment for farmer i as a function of the announced parameters Θ_i of all agents, given that other agents pursue truth-telling strategies, are:[8]

$$e_i(\Theta_i) = e_i(\Theta_i, \Theta_{-i}) \quad \text{and} \quad t_i(\Theta_i) = t_i(\Theta_i, \Theta_{-i})$$

Based on Xepapadeas (1997) it is assumed that the farmer i's incentive scheme depends totally on the responses of the particular farmer and thus can be reduced in $\{e_i(\Theta_i), t_i(\Theta_i)\}$. However, under such a reduced-form mechanism the full information optimum is not attainable since effluent levels do not depend on the entire vector of heterogeneity parameters (Spulber, 1988).

[8] Where $\Theta_{-i} = (\Theta_1, ..., \Theta_{i-1}, \Theta_{i+1}, ..., \Theta_n)$.

Under an incomplete information framework the reported parameter value, defined as $\widetilde{\Theta}_i$, may vary from the true parameter value Θ_i since the individual agents may take advantage of their private information in order to achieve lower tax payments and thus higher net benefits from production. Consequently the associated after-tax profit function from reporting parameter value $\widetilde{\Theta}_i$, when the true parameter is Θ_i is given as $\pi_i(\Theta_i, \widetilde{\Theta}_i) = \pi_i(\mathbf{x}_i(\Theta_i)) - t_i(\widetilde{\Theta}_i)$, where the emission-based tax payment is:

$$T_i(e_i(\widetilde{\Theta}_i)) = t_i e_i(\mathbf{x}_i(\widetilde{\Theta}_i)) = t_i \left\{ e_i(\mathbf{x}_i(\widetilde{\Theta}_i)) - \widetilde{\Theta}_i \right\}$$

Nevertheless the regulation mechanism can induce truthful behavior if both $\pi_i(\Theta_i, \Theta_i) \geqq \pi_i(\Theta_i, \widetilde{\Theta}_i)$ and $\pi_i(\Theta_i, \Theta_i) \geqq 0$ hold respectively for all $\Theta_i, \widetilde{\Theta}_i$ in $\left[\underline{\Theta}_i, \bar{\Theta}_i\right]$ (Spulber , 1988). Alternatively truth-telling can be optimal if the true parameter value guarantees the maximum payoff, implying that $\Theta_i \in \arg\max_{\widetilde{\Theta}_i} \pi_i(\Theta_i, \widetilde{\Theta}_i)$, where $\frac{\partial \pi_i(\Theta_i, \widetilde{\Theta}_i)}{\partial \Theta_i} - \frac{\partial t_i(\Theta_i)}{\partial \Theta_i} = 0$ is the the first order condition for incentive compatibility (Xepapadeas , 1997).

In this context the emission tax rate**,** under truthtelling, is defined as:

$$t_i = \left[\frac{\partial D(e)}{\partial e} \frac{\partial e(e_i)}{\partial e_i} \frac{\partial e_i(\cdot)}{\partial x_{ij}} + \frac{\tilde{\lambda}}{1+\tilde{\lambda}} \frac{H(\Theta_i)}{h(\Theta_i)} \frac{\partial^2 \pi_i(\mathbf{x}_i(\Theta_i))}{\partial \left(\mathbf{x}_i\right)^2} \right] \Big/ (1+\tilde{\lambda}) h(\Theta_i) \left[\frac{\partial e_i(\cdot)}{\partial x_{ij}} \right]$$

According to Xepapadeas (1997) under incomplete information the most efficient agents emit relatively more compared to the complete information framework, while the less efficient agents reduce their emissions, receiving information rents to secure truthful revelation of their type and the discharges of the marginal agent $\underline{\Theta}_i$ are the same as the optimal emissions under complete information.

Finally, it is worth mentioning that full information policies, such as the "equal marginal cost" rule, are feasible under incomplete information if the policy scheme can be supplemented by a costly government subsidy for truth-telling by agents. In this case taxes are reduced so that farmers obtain optimal rents and full information optimum is feasible if and only if net social welfare from production exceeds the total information rents (Spulber, 1988).

According to Cochard (2003) such a revelation policy scheme is inapplicable due to the formidable amount of required information, implying that the scheme should be modified to the second best that does not require private information or even relay on uniform tax rates. Nevertheless, the author underlines that under special cases the first-best solution can be achieved despite the fact that the regulator is uninformed of the polluting agents' type Θ_i. This requires that the expression (6.5) is independent of i or j's, meaning that all the environmental functions: social damage $D(e)$, aggregate pollution e and individual discharges e_i, should be linear and the associated derivatives should be the same across the regulated agents. Under these circumstances the regulator can achieve the social optimum through an emission tax rate that is uniform across dischargers.

This is actually the case of the lump-sum tax $\bar{T}$ developed by Millock at al. (2002) previously. It is an extreme case where the regulator cannot observe the parameter Θ_i and does not define a revelation mechanism to infer the parameter Θ_i. It is worth mentioning that under this policy scheme only the agents with $\Theta_i \geq \Theta_m^0(T)$ will continue to operate under the lump-sum tax, where $\Theta_m^0(T)$ is the marginal value satisfying with equality the participation constraint $\pi_i(\Theta_i) = \pi_i(\mathbf{x}_i(\Theta_i)) - T_i(e_i(\Theta_i)) \geq 0$. For $\Theta_i < \Theta_m^0(T)$ it holds $\pi_i(\Theta_i) < 0$, while when $\Theta_m^0(T) = \underline{\Theta}_i$ then the ambient pollution is at its maximum level. According to the authors the instrument works best when profits and pollution levels are negatively correlated, since the units closed down are these with the lowest contribution to output - least profitable - and the highest pollution, while the remaining have high productivity and low pollution. In the opposite case the most polluting agents remain unaffected and exiting farmers may have both high pollution and high output levels.[9]

6.2 Input-based Schemes

In the context of NPS pollution problems, individual discharges are not directly observed and thus emission-based schemes are not in generally feasible. Nevertheless the regulator can circumvent the inherent informational constraints through input-based schemes that are more feasible for NPS pollution control (Helfand and House, 1995) and have been recognized as substitutes for direct taxes on negative externalities (Shortle et al., 1998; Legras, 2004).

[9] The same conclusions hold under mandatory monitoring.

Input-based incentives include charges or restrictions on inputs that increase a detrimental externality and subsidies on inputs that reduce it (Shortle et al., 1998). Such indirect measures could replicate the marginal conditions for an efficient solution (Shortle et al., 1998) if a set of differential taxes or standards are imposed on each polluting agent. However, in practice informational or / and administrational constraints allow only the implementation of uniform or nonuniform input-based schemes applied on a single or a subset of inputs, which achieve the social optimum only under quite unlikely circumstances (Helfand and House, 1995).

This section presents developments in input-based mechanisms for NPS pollution problems. It should be noted that in the context of the EU agricultural policy, input-based schemes imply the development of policies on inputs such as fertilizers, pesticides, water, and land in the context of land-set-aside programmes.

6.2.1 Under certainty

- **Static context:**

Input-based schemes have been suggested as a means to induce changes in farmers' behavior (Johnson et al., 1991) and there is economic evidence that they can be effective in bringing changes in resource allocation, depending of course on how they are structured (Shortle et al., 1998).

Under complete information the regulator imposes a linear tax on each input choice $\mathbf{x}_i$ and thus the total tax payment for the source of pollution i is $T_i(\mathbf{x}_i) = \mathbf{t}_i\mathbf{x}_i = \sum_{j=1}^{m} t_{ij}x_{ij}$, where $\mathbf{t}_i$ is a vector of nonuniform taxes rates corresponding to each input j affecting agricultural pollution. The after-tax objective function of farmer i is: $\pi_i(\mathbf{x}_i) - T_i(\mathbf{x}_i)$ and the associated nm first-order-conditions are:

$$FOC_{x_{ij}} : \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} - \frac{\partial T_i(\mathbf{x}_i)}{\partial x_{ij}} = 0 \qquad \forall i, j$$

After comparing with the FOC (5.7), corresponding to the social optimum, the input-based tax scheme must satisfy:

$$t_{ij} = \frac{\partial D(e)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \qquad \forall i, j \tag{6.8}$$

For each polluter the regulator defines m tax rates. So if the regulated area consists of n farmers then the necessary number of input-based instruments is $m \times n$. In contrast the number of effluent-based schemes is n and each agent faces a single emission-based instrument.

In the same context the regulator can define a set of performance standards ($x_{ij} \leq \bar{x}_{ij}$) for each input choice and for each polluting agent. Obviously, if the initial application rate is greater than the limit then the application rate must be reduced to the limit, otherwise the usage-decision is not affected (Wu and Babcock, 2001). In the special case where the limits are set equal to zero then the particular input choice is banned (Mapp et al., 1994). An x_{ij} percentage reduction in pollution can be achieved by rollbacks in inputs of x_{ij} percent only if the relevant pollution function is homogeneous of degree one, otherwise if the pollution function exhibits decreasing (increasing) returns to scale then the percentage required reduction in inputs will be greater (less) than the targeted percentage reduction (Helfand and House, 1995). Finally, in a world of perfect information both input taxes and input standards can be used to achieve the social optimum and this equivalence depends on the assumption that the agency knows each individual farm's physical attributes and can impose differential taxes or standards (Wu and Babcock, 2001).

In US several restrictions on water use have been imposed as a result of drought conditions in some states, while in Denmark under the Aquatic Environment Action plan of 1987 there are standards for manure storage capacities and the application of manure on agricultural land. Nevertheless, the most notable production input constraint is land set-aside, implying withdrawal of land from production that can either be mandatory or voluntary (OECD, 1993). Under Agenda 2000 a long-term set-aside mechanism (ten years) for arable land is proposed in place of the existing rotational set-aside (EC, 2004). Farmers entitled to direct payments must set aside part of their land, on which however energy crops such as oilseeds or bio-mass can be cultivated. From the land set-aside mechanism is excempted land used for organic production or for the production of materials not intended for human or animal consumption. However, according to OECD (1993) the net environmental effect of such production is uncertain since it may encourage input more intensive input use on the remaining production base.

More specifically the regulator can define the following input tax/subsidy scheme, where a vector of g input tax rates $\mathbf{t}_i^p$ is imposed on productive inputs $\mathbf{x}_i^p$ and a vector of $(m-g)$ subsidy rates $\mathbf{t}_i^a$ is provided on abatement inputs $\mathbf{x}_i^a$. This scheme is applied to each agent and in this context the profit function is given by:

$$\pi_i(\mathbf{x}_i^p, \mathbf{x}_i^a) = pf(\mathbf{x}_i^p) - \left(\mathbf{w}_j^p + \mathbf{t}_i^p\right)\mathbf{x}_i^p - \left(\mathbf{w}_j^a - \mathbf{t}_i^a\right)\mathbf{x}_i^a$$

For instance, input taxes can be imposed on fertilizer use such as nitrogen, potassium and phosphate (Austria, Finland, Norway, Sweden), while subsidies can be provided for less polluting fuels such as natural gas (Canada), as well as for irrigation (Japan) (OECD, 1993). In particular, in Austria there has been a levy on fertilizer use since 1986, imposed per kg of pure nutrients contained in inorganic fertilizers (i.e. nitrogen, potassium, phosphate) (OECD, 1993). In the same context in Norway a general tax of around 15% on fertilizers is introduced and a list of approved fertilizers for agricultural use is renewed every five years. Finally, in Finland a tax on phosphate fertilizers has been operation since 1990, while in Sweden a tax on both nitrogen and phosphate is implemented (OECD, 1993). Finally, it is notable that in Japan irrigation subsidies are not regarded as environmentally undesirable and no salinity problems have been detected since their introduction. However, the provision of production input-oriented subsidies, such as grants for draining wetlands or cleaning woodlands, may have direct and indirect negative environmental effects (i.e. wildlife habitat destruction) (OECD, 1993).

After comparing the corresponding $FOCs$ with (5.8) and (5.9) respectively, the input-based tax and subsidy rate must be equal to:

$$t_{ij}^p = \frac{\partial D(e)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i^p)}{\partial x_{ij}^p} \quad \text{and} \quad t_{ij}^a = \frac{\partial D(e)}{\partial e}\frac{\partial D(e)}{\partial e_i}\frac{\partial a_i(\mathbf{x}_i^a)}{\partial x_{ij}^a} \qquad \forall i, j$$

By taxing inputs that increase a detrimental externality and subsidizing inputs that reduce it the regulator could replicate the marginal conditions for an efficient solution (Shortle et al., 1998). Such a policy scheme requires that the net emission function (5.1) is known and that there is perfect correlation between observed inputs and emissions (Xepapadeas, 1997).

The previous policy schemes require input-based instruments to be applied to all inputs influencing emissions, and be varied across farmers in the regulated region,[10] due to heterogeneity in production and environmental effects (Classen and Horan, 2001). Such instruments provide the social optimum and guarantee efficient NPS pollution control at least cost. Therefore in the case of agriculture:

> A first-best input-based policy scheme targets production choices influencing nitrate leaching such as fertilizer applications, water use, technology and land use (Classen and Horan, 2001), charging simultaneously each user of an input a different tax for that input, as optimality would require if the users are not identical (Helfand and House, 1995).

However, even if these input-based mechanisms implement optimum in dominant strategies[11] they are informationally demanding (Hansen, 2002) - since the regulator must acquire detailed information on each agent's input usage and how these inputs affect the emission function. Input-based mechanisms are strongly dependent on the regulator's ability to deal simultaneously with all inputs contributing to the externality (Larson et al., 1996). Moreover, even if the marginal damages are constant across sources (Helfand and House, 1995) different sets of input taxes are required since emission functions differ, implying that first-best solutions occur under more stringent conditions (Larson et al., 1996). Finally, political constraints or diminishing marginal efficiency gains may also make impossible the regulation of all polluting inputs in a first best way (Larson et al., 1996). Hence:

> Given the often extreme spatial variation in the agricultural resource base, efficient input-based NPS pollution control may be administratively costly[12] and difficult (Classen and Horan, 2001), or even impossible.

In practice these information costs and/or other considerations require that the input-based incentives are modified to the second-best policies, focusing to a subset of input choices that are both

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[10] Or even vary for each soil type (Helfand and House, 1995).

[11] A strategy is dominant if no farmer has incentives to react to changes in other farmers' input (and emission) choices.

[12] Administrative costs include the resource costs of designing, enforcing, monitoring and managing the policy but do not include subsidies.

relatively easy to observe and correlated with ambient impacts, often imperfectly (Shortle et al., 1998; Schmutzler and Goulder, 1997). Such input-based schemes target only a limited set of inputs ($k < m$) (Classen and Horan, 2001) closely related to ambient emissions. The tax payment is applied nonuniformly across polluting agents and is rewritten as $T_i(\mathbf{x}_i) = \sum_{j=1}^{k<m} t_{ij}x_{ij}$, where $j = 1, ..., k$ inputs are regulated while $m-k$ inputs remain unregulated. In this context, the first-order conditions for after-tax profit maximization are (Shortle et al., 1998):

$$\frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} = t_{ij} \text{ for } j = 1, ..., k \text{ and } \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} = 0 \text{ for } j = 1, ..., m-k$$

In practice agricultural policies have to be based on factors that affect pollution only indirectly and are relatively easily observed (Huhtala and Laukkanen, 2004). Water and nitrogen appear to be the key variables both for crop and nitrate production (Helfand and House, 1995) and are highly correlated with water pollution. Thus the regulation of both nitrogen and water use is effective in reducing nitrogen leaching (Wu and Babcock, 2001). In particular, charging individual tax prices for irrigation water may be possible, especially when it is administered by a water district and imposed directly on a user by the district (Helfand and House, 1995). Regarding regulations on fertilizers and pesticides use, they may not be a perfect measure of environmental harm but they are strongly correlated with water pollution (Abler and Shortle, 1995). Nevertheless, if the regulator takes into account the additional complexity and costs of getting the taxes on both inputs “right” then water taxes may be the best way to reduce nitrate leaching - even though the final choice might be affected by the distribution of farmers in different production or pollution classes (Larson et al., 1996).

Finally, it is important to mention that economic-biophysical models, describing interactions between production technologies and environmental pollution, can be used for designing the taxes on such observed inputs taking also into account stochastic changes in "site quality" (Xepapadeas, 1999).

Uniform input-based scheme:

Even though nonuniform input incentives are theoretically capable of achieving the social optimum, in real terms they could be difficult to implement since as Shortle et al., (1998) point out:

1. The regulated input could be subject to resale generating incentives for arbitrage between agents. For instance it could be impossible to maintain separate prices for a fertilizer for different users (Helfand and House, 1995) since farmers facing lower taxes could buy large quantities and resell to those who would otherwise face higher taxes. Such an arbitrage could defeat the efficiency of the firm-specific tax system.

2. Under the farm-specific tax system the amounts used by individuals have to be monitored (or verified) and priced individually (Helfand and House, 1995). This involves substantial information costs since there are many resource management decisions that have a great impact on the environment and a great number of them is very unlikely to be observable - self-produced or immaterial inputs (Cochard, 2003).

Therefore NPS can be considered as:

> A situation where marginal pollution costs and marginal profit from chemical use vary over space but differential taxes or standards cannot be employed (Wu and Babcock, 2001).

Given the above problems input-based schemes are modified to the second-best and are applied on a limited set of inputs at uniform rates across producers (Classen and Horan, 2001). Such regulations involve identical taxes, restrictions or more precisely pre-acre restrictions that can be either imposed broadly or target certain categories of polluters. In particular, broad policy schemes are applied to all polluting sources (Helfand and House, 1995), while targeted policies are applied on certain soil types or certain production systems (Mapp et al., 1994). Even though uniform schemes cannot attain the social optimum almost with certainty (Helfand and House, 1995) and are inefficient in NPS problems (Underwood and Caputo, 1996), since they eliminate potential gains from differential treatment of polluters according to their relative impacts on ambient conditions (Shortle et al., 1998),[13] they may be a preferable measure in the agricultural

[13]Under uniform input-oriented policy schemes high control cost or low damage cost

case due to the complexity and cost of getting separate instruments "right" for both inputs and soil types.

Regarding the broad policies alternatives, a uniform set of input taxes imposed on all land types allow the producers to allocate input choices (i.e. nitrogen) across crops and soils as desired, while restrictions set at per-acre level basis induce input reductions on many soil types which are not problematic and producers no longer have the flexibility of meeting the restriction by primarily reducing applications (i.e. nitrogen) on acres (i.e. dryland acres) where inputs usage and potential environmental damages are both less than under irrigated production (Mapp et al., 1994). Total restrictions are considered to be identical to a set of marketable input permits, limiting production of a polluting input but permitting sales of the input within the constraint (Helfand and House, 1995).

Often broad policies involve the regulation of a single input, since coordination among all the agencies - authorized to monitor or regulate inputs such as pesticides, fertilizers, water without authority over each other (Helfand and House, 1995) - to get taxes "right" can be difficult at best (Larson et al, 1996), suggesting that only one input should be taxed uniformly across soil types (Helfand and House, 1995). According to Helfand and House (1995) if only one input is to be taxed or constrained then water appears to be the preferable input to regulate in terms of achieving NPS reduction goals at lower cost (Fleming and Adams, 1997). Even though the tax bill under the water tax is twice the tax bill under the nitrogen tax, the larger welfare losses result from the nitrogen tax and the control on nitrogen application (Helfand and House, 1995). This is because such instruments allow little substitution between inputs, and the pollution function is more elastic with respect to water use than nitrogen applications, requiring thus relatively higher nitrogen taxes or restrictions (Larson et al., 1996).

Finally regarding targeted policies, the regulator targets soils where pollution is most likely to occur. In the case of nitrate leaching the coarser, more permeable soils should be targeted (Mapp et al., 1994). The effectiveness of such a policy depends on the distribution of soils in the regulated area and may not produce the anticipated reduction

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polluters will end up devoting to many resources to pollution control, while low control cost or high damage cost polluters will devote too few resources to pollution control.

in nitrate losses, justifying targeting restrictions on particular production systems that may produce significant quantities of pollution (Mapp et al., 1994). In the case of nitrate pollution, such a policy targets all furrow irrigated acreage, restricting nitrogen application. As a response producers are expected to shift to the adoption of sprinkler irrigation, reducing furrow irrigation (Mapp et al., 1994).

In such cases:

> The easily implemented second-best instruments must be weighted against the increased social cost they impose (Helfand and House, 1995).

Even though partially targeted policies appear to be overall attractive in a heterogeneous setting and have economic advantages over uniform or broad policies in a number of settings (Fleming and Adams, 1997), in a second best world a uniform or broad policy may be more appropriate. In practice the implementation of a spatial policy requires a significant amount of information on physical processes and involves high cost of monitoring and enforcing, making questionable whether the gains (cost savings) of a spatial policy are sufficient to cover the cost of data acquisition (Fleming and Adams, 1997). On the other hand, uniform or broad policies are often preferred because they are easier to administrate and seemingly more fair to the agricultural producers and resource owners (Classen and Horan, 2001).

In a second best context the regulator needs also to define the type of the instrument to be applied since under spatial heterogeneity a tax is preferred under some physical settings and a standard is preferred in others (Wu and Babcock, 2001). In a first best world regulators would vary both the type and the level of the instrument as marginal benefits and costs vary across the landscape, however, such a policy is difficult to be implemented in a agricultural NPS setting and thus the regulator needs to define which measure is more efficient in terms of expected difference in social surplus. In this context:

> The advantage of a tax over a standard depends on the relative slopes of the marginal cost and profit function (Wu and Babcock, 2001).

Therefore, a combination of steep marginal costs and flat marginal profits results in larger dead weight loss under the uniform tax, favoring a uniform standard. The relative advantage of the tax also decreases when either the slope of marginal cost, the positive correlation between the marginal costs and marginal profits, or the variance in farmers' responses to the tax increases. On the other hand, the relative efficiency of the input-use tax increases whenever marginal profits are more sensitive to chemical use than marginal pollution costs on land where marginal profits are high, marginal profits are less sensitive than marginal costs on land where marginal profits are low, marginal profits are negatively correlated with marginal pollution costs, and the variation in farmers' responses to the tax is small. Additionally, under high chemical application efficiencies marginal costs are likely to be low with flat slopes and marginal profits are likely to be high, favoring the uniform tax (Wu and Babcock, 2001).

Nevertheless, the conventional finding of tax or standard superiority based on the relative-slope rule of marginal profits and marginal pollution costs of chemical use can be reserved in the presence of corner solutions, since under spatial heterogeneity a uniform tax rate results in some farmers not using the chemical or a uniform standard has no effect on farmers whose profit-maximizing chemical-use levels are below the uniform standard[14] ((Wu and Babcock, 2001).

Finally, under spatial or agent heterogeneity a policy that involves price regulations is superior to a policy that focuses on quantity controls (Kampas and White, 2004). In particular, input taxes are more desirable than input restrictions (or an outright ban) since they give producers a choice over how to respond to the instrument (Helfand and House, 1995) and encourage farmers to be more selective in their input use and to substitute an alternative when cost effective (Underwood and Caputo, 1996). Nevertheless, it is worth mentioning that a negative correlation between the prices of chemicals and the efficiency of the nitrogen application, suggests that a standard is likely to be more efficient when the chemical price is low due to the

[14] For example in US.Central Nebraska Basin the 24% of corn acres has nitrogen application rates below 50lbs/acre and the 17% above 150 lbs/acre, thus a nitrogen-performance standard above 50 lbs/acre would have no effect at least 20% of corn acreage (Wu and Babcock, 2001).

"insurance nitrogen" incentive (Wu and Babcock, 2001).[15]

- Dynamic context:

Dynamic input taxes are designed in such a way so that the farmers' choices for determining the optimal amount of input use are the same to the socially optimal choices (Xepapadeas, 2005). In this context the regulator can apply time-dependent, time invariant, or semi-time invariant, as well as steady-state tax rates to deal with agricultural NPS pollution. These first- and second-best input-based schemes are actually the same policy schemes defined in the previous section for the emission-based policy schemes, with the only difference that the tax rates are multiplied by the derivative $\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$. In particular the time-flexible tax rates for input j are given by:[16]

$$\begin{aligned}
\tau_{ij}(t) &= -\mu(t)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \\
\tau_{ij}^{OL}(t) &= -\left(\mu(t) - \mu_i^{OL}(t)\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \\
\tau_{ij}^{FB}(t) &= -\left(\mu(t) - \mu_i^{FB}(t)\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}
\end{aligned}$$

The associated steady-state tax rates for input j are equal to:

$$\begin{aligned}
\tau_{ij}(t) &= -\mu^{\infty}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \\
\tau_{ij}^{OL}(t) &= -\left(\mu^{\infty} - \mu_i^{OL\infty}\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} \\
\tau_{ij}^{FB}(t) &= -\left(\mu^{\infty} - \mu_i^{FB\infty}\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}
\end{aligned}$$

where the total input tax payment for farmer i in period t is $T_i(\mathbf{x}_i) = \boldsymbol{\tau}_i(t)\mathbf{x}_i(t)$, with $\boldsymbol{\tau}_i = (\tau_{i1}, \tau_{i2}, .., \tau_{im})$ the vector of dynamic input tax rates.

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[15] A low price gives farmers an incentive to apply "insurance nitrogen" to guard against years in which large amounts of soil-stored nutrient are lost leading to low application efficiency.

[16] These tax schemes resulted after comparing the associated first-order-conditions $\forall i, j$ under a myopic, OL and FB behavioral rule: $\frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} - \tau_{ij}(t) = 0$, $\frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} - \tau_{ij}(t) + \mu_i^{OL}(t)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} = 0$ and $\frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} - \tau_{ij}(t) + \mu_i^{FB}(t)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} = 0$ with (5.20a).

Nevertheless, under strategic interaction a dynamic input policy scheme can be also based on a linear Markov perfect tax rule, that is linear in inputs and dependent on the current pollution stock level (Legras, 2004). In this case the total tax payment is given by:

$$T_i(\mathbf{x}_i, S) = g(S)\mathbf{x}_i \qquad \text{with} \qquad g(S) = AS + B$$

where $g(S)$ is the optimal tax rate that has a component independent of the pollutant stock (B) and a stock-dependent component (AS). The after-input tax problem for the farmer is:

$$\max_{\mathbf{x}_i} \int_0^{\infty} \exp(-rt) \left\{\pi_i(\mathbf{x}_i, S) - g(S)\mathbf{x}_i\right\} dt$$
$$\text{s.t.} \quad (5.23) \text{ or } (5.24)$$

where after following the standardized procedure the associated Pontryagin conditions are derived under the open-loop and feedback informational structure respectively. To define the values of A and B under the each behavioral rule employed a linear quadratic expression for the benefit function is used.[17] It is found that $B^{OL} < B^{FB}$ and $A^{OL} > A^{FB}$.

Under an OL informational structure the obtained dynamic input tax rate: $g^{OL}(S) = A^{OL}S + B^{OL}$ is based on the current state of the stock which is affected by the past input decisions, even if the current input decisions are optimal (Legras, 2004). Such a tax rule sends the polluters a message that the more inputs they use now, the higher their emission are and thus the higher the pollution stock and their future tax liability. Therefore farmers are expected to realize that their emissions will affect the future tax rate since they contribute to the pollution accumulation. On the other hand, under a FB informational structure the FB tax rule punishes farmers more heavily than the OL at low pollution stock levels, while the opposite holds at high pollution stock levels. This is logical since under the FB strategy farmers over-emit now because there is a belief that the other farmers will cut their emissions to counterbalance the increase

[17] For details see Legras (2004) page 10. It is worth mentioning that the same procedure can be replicated by our model under the assumption that $\frac{\partial^2 e_i}{\partial (x_{ij})^2} = 0$. In the opposite case the analysis becomes too complicated.

in the stock pollution and thus the fixed tax (B^{FB}) parameter is set high to induce them to lower their emissions from the beginning (Legras, 2004).

Finally, it is important to refer to how the structure of dynamic input-based tools is modified under quantity-quality problems. Irrigation agriculture puts great pressure on the quantity and quality of water resources (Karagiannis and Xepapadeas, 2001). In particular, overextractions of water resources for irrigated agriculture is another major problem of European environment, since these overextractions have resulted in pollution of groundwater resources because of percolation of agricultural pollutions (i.e. nitrate pollution) and seawater intrusion in coastal aquifers (i.e. salinity). Such extensive use of groundwater can have detrimental effects upon the physical and chemical characteristics of the resource, putting into danger the sustainability of its use and thus the sustainability of agricultural production. Therefore a regulator needs take into account both the evolution of the natural resource and the pollution stock, while designing a time-dependent or independent input tax scheme imposed on irrigation water.

Based on Xepapadeas (2005) the regulator's problem is modified to:

$$\begin{aligned}
&\max_{\mathbf{x}_i,\mathbf{x}_{-i}} \int_0^{\infty} \exp(-\rho t) \left\{ \sum_{i=1}^{n} \pi_i(\mathbf{x}_{-i1}, \mathbf{x}_i, S) - D(S) \right\} dt \\
&\text{s.t.} \quad \dot{R} = F(R) - \mathbf{x}_{i1} \qquad (6.9) \\
&\qquad\quad \dot{S} = \sum_{i=1}^{n} e_i(\mathbf{x}_{-i1}, \mathbf{x}_{i1}) - bS
\end{aligned}$$

where $\mathbf{x}_{i1} = (x_{11}, x_{21}, ..., x_{n1})$ is the vector of individual water extractions, $\sum_{i=1}^{n} x_{i1}$ is total extraction, and $\mathbf{x}_{-i1}$ the vector of the rest input choices made by $i = 1, ..., n$ farmers. The damage function $D(S)$, $D' > 0$, $D'' \geq 0$, reflect environmental damages due to agricultural pollution, such as damages in wetlands. The expression (6.9) describes the evolution of the natural resource (R) within the groundwater aquifer, with $F(R)$ the natural recharge rate.

After following the established procedure the social shadow cost of pollution stock ($\mu(t)$) and water resource ($\psi(t) > 0$), as well as the private shadow cost of pollution stock and water resource under the open-loop and feedback behavioral rule are defined. Therefore, under the quantity-quality problem the time flexible tax rates for the input $j = 1$ (irrigation water) under the myopic, open-loop and feedback behavioral rule are given by:

$$\tau_{ij}(t) = \psi(t) - \mu(t)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$$
$$\tau_{ij}^{OL}(t) = \left(\psi(t) - \psi_i^{OL}(t)\right) - \left(\mu(t) - \mu_i^{OL}(t)\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$$
$$\tau_{ij}^{FB}(t) = \left(\psi(t) - \psi_i^{FB}(t)\right) - \left(\mu(t) - \mu_i^{FB}(t)\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$$

where the tax scheme has two elements, the first term accounts for water overextractions while the later for excess pollution.

In the same context the steady-state tax rates for the input $j = 1$ are:

$$\tau_{ij}(t) = \psi^{\infty} - \mu^{\infty}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$$
$$\tau_{ij}^{OL}(t) = \left(\psi^{\infty} - \psi_i^{OL\infty}\right) - \left(\mu^{\infty} - \mu_i^{OL\infty}\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$$
$$\tau_{ij}^{FB}(t) = \left(\psi^{\infty} - \psi_i^{FB\infty}\right) - \left(\mu^{\infty} - \mu_i^{FB\infty}\right)\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$$

Finally, under certain assumptions it can be shown that the inequality: $x_{ij}^{o} > x_{ij}^{FB} > x_{ij}^{OL} > x_{ij}^{*}$ holds regarding the ranking of extraction paths, implying that at the social optimum the pollution stock level is smaller and water conservation is greater compared to the corresponding values in the absence of regulation.

6.2.2 Under uncertainty

- Static context:

Under a differentiated input-based tax scheme the expected total tax payment is $E\left\{T_i(\mathbf{x}_i)\right\}$ and the after-tax payoff function: $E\left\{\pi_i(\mathbf{x}_i,\theta)\right\} - E\left\{T_i(\mathbf{x}_i)\right\}$. Thus the associated nm first-order-conditions are:

$$FOC_{x_{ij}} : E\left\{\frac{\partial \pi_i(\mathbf{x}_i,\theta)}{\partial x_{ij}}\right\} - E\left\{\frac{\partial T_i(\mathbf{x}_i)}{\partial x_{ij}}\right\} = 0 \qquad \forall i,j$$

After comparing the above $FOCs$ with (5.15), the optimal non-uniform input tax scheme must be equal to the expected increase in damages from a marginal increased use of each input j and can either be linear or nonlinear (Cochard, 2003). In particular:

$$E\left\{\frac{\partial T_i(\mathbf{x}_i)}{\partial x_{ij}}\right\} = E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right] \qquad \forall i, \forall j \tag{6.10}$$

According to Shortle et al. (1998) the optimal marginal input tax rate can also include a covariance term acting as a risk premium or reward depending on the sign. Hence:

$$t_{ij} = E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right] + cov\left\{\frac{\partial D(e,\eta)}{\partial e}, \frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right\} \tag{6.11}$$

This is actually a system of *farm-specific, per-unit input tax/subsidy rates*, since the tax rate may be positive or negative depending on the signs and relative magnitudes of the two effects, providing thus, the correct marginal incentives for input use.[18]

However, such input tax/subsidy structures are exceptionally complex, which makes necessary that the policy scheme is modified to the second-best. Under this context input taxes are applied uniformly to a subset of inputs and the total tax payment is $T_i(\mathbf{x}_i) = \sum_{j=1}^{k<m} t_j z_i$ with z_i the vector of $k < m$ regulated inputs and h_i the vector of the remaining $m - k$ unrestricted inputs. [10]

Based on Shortle et al., (1998) the second-best uniform tax rate t_u imposed on the restricted input u takes into account (i) the average marginal impact of restricted input substitution on expected damages and profit levels, (ii) the average change in expected damages due to a marginal increase in the use of the input u and (iii)

[18] If the number of farmers is not optimal then an additional instrument would be necessary to guarantee that the optimal number of farmers. Such a tool is a lump sum tax charged to extra-marginal farmers which will act as an entrance or license fee. (Shortle et al., 1998).

[10] A restricted input must also be *verifiable* Nyborg (2000). This implies that this input is covered by a formal tax base and can be enforced by a third party such as a legal court, otherwise the farmers could refuse paying the tax. Nevertheless, a bilateral agreement between the regulator and farmers can decrease both the use of restricted and unrestricted inputs (Nyborg, 2000).

the average marginal impact of unrestricted input substitution on expected damages.[19] Therefore the tax rate t_u is given by:

$$t_u = \sum_{i=1}^{n}\sum_{i\neq j}^{k}\left\{E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial z_{ij}}\right]-E\left(\frac{\partial \pi_i(\mathbf{x}_i)}{\partial z_{ij}}\right)\right\}\rho_{iu}\omega_{iju}+ \\ +\sum_{i=1}^{n}E\left[\frac{\partial D(\cdot)}{\partial e}\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial z_{ij}}\right]\rho_{iu}+\sum_{i=1}^{n}\sum_{i\neq j}^{k}\left\{E\left[\frac{\partial D(\cdot)}{\partial e}\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial z_{ij}}\right]\right\}\rho_{iu}\gamma_{iju}$$

where (ρ_{iu}) is the proportion of the aggregate change in the usage of the regulated input u used by the farmer i, (ω_{iju}) the substitution rate of farmer i's use of jth restricted input for the uth restricted input due to a marginal increase in the tax rate on the uth restricted input, and (γ_{iju}) the substitution rate of farmer i's use of jth restricted input for the uth unrestricted input due to a marginal increase in the tax rate on the uth restricted input.[20]

Finally, whether the particular tax (subsidy) rates on restricted inputs are higher or lower than the optimal marginal input tax rate (6.11) depends on the direct effect of the regulated inputs on the environmental quality and their substitution relationship with untaxed factors (Shortle et al., 1998).

It is possible that agricultural NPS regulations are likely to produce input and output market price effects, requiring that the second-best input policy scheme must be modified to account both for substitution and pecuniary externalities (Classen and Horan, 2001). In such a case the tax-induced substitution and output effects on producers in sub-region i affect market prices and thus producers in other sub-regions $-i$. Such pecuniary externalities occur both for uniform and non-uniform input based tax rates and have market price effects that affect both farmers' profits and existing environ-

[19] The procedure followed by the authors in order to define the second-best-optimum tax is different from the standard process used in this survey, and follows more closely the optimal taxation approach for a case where the regulator is a Stakelberg leader choosing optimal taxes, and farmers are followers that take taxes as given and maximize profits. In particular optimal tax rates are determined by plugging the input choices that satisfy the conditions: $\frac{\partial \pi_i(\mathbf{x}_i)}{\partial z_{ij}} = t_j$ for $j = 1, ..., k$ and $\frac{\partial \pi_i(\mathbf{x}_i)}{\partial h_{ij}} = 0$ for $j = 1, ..., m-k$ for any t_j into the regulator's objective function and then choosing tax rates to maximize the expression: $\max_{t_u} \sum_{i=1}^{n} E_i\{\pi_i(\mathbf{z}_i(t), \mathbf{h}_i(t))\} - E[D(e,\eta)]$.

[20] In particular it holds $\rho_{iu} = (\partial z_{iu}/\partial t_u) / \left(\sum_{i=1}^{n} \partial z_{iu}/\partial t_u\right)$, $\omega_{iju} = (\partial z_{ij}/\partial t_u) / (\partial z_{iu}/\partial t_u)$ and $\gamma_{iju} = (\partial h_{ij}/\partial t_u) / (\partial z_{iu}/\partial t_u)$.

mental externalities in other sub-regions, by altering social pollution control costs and hence the level of pollution control in these areas.[21]

Hence, the *second-best optimal non-uniform tax* designed for fertilizer use is specified as:[22]

$$t_i = \lambda^i E\left\{\frac{\partial e_i}{\partial g_i}\right\} - \lambda^i \left[E\left\{\frac{\partial e_i}{\partial g_i}\right\} g_i - E\{e_i\}\right]\omega_1^{ii} + \sum_{k\neq i} \upsilon^i E\left\{\frac{\partial e_i}{\partial g_k}\right\}\zeta^{ik}$$
$$-\sum_{k\neq i}\lambda_i\left(E\left\{\frac{\partial e_i}{\partial g_k}\right\} g_k - E\{e_k\}\right)\omega_1^{ki}\zeta^{ik} - \sum_{k\neq i} t_k\zeta^{ik} \quad \forall i$$

The first term is the first-best tax evaluated at the second-best optimum, while the remaining terms represent the additional substitution (second term) and pecuniary externalities (final three terms) created by the tax. In particular the third and forth term represent the marginal impacts of input and output price effects due to t_i on the other sub-regions' pollution control costs, and the final term indicates that as the taxes in other sub-regions are increased the tax in sub-region i must increase in order to compensate for reduced effectiveness. It is notable that in the absence of price effects the final three terms vanish.

The *second-best optimal uniform tax* designed for fertilizer use is:

$$t = \left(\upsilon^i E\left\{\frac{\partial e_i}{\partial g_k}\right\} - \upsilon^i\left[E\left\{\frac{\partial e_i}{\partial g_k}\right\} g_k - E\{e_k\}\right]\omega_1^{kk}\right)\gamma$$

where $\gamma = (dx_{k2}/dt)\,/\,(\sum_{i=1}^n dx_{is}/dt)$. This is actually a single tax rate used to achieve an environmental goal, determined such that (5.16) holds as an equality in sub-region k and induce overcompliance in all other sub-regions. If this overcompliance produces appreciable market effects then significant pecuniary externalities may be imposed on sub-region k.

[21] The regulator's problem, from which the particular nonuniform and uniform second-best-optimum tax rates were defined, is quite different than the one used in this survey. In particular the authors proceed in the maximization of the net private surplus subject to an environmental objective. For details see Classen and Horan (2001) page 6-5.

[22] Where g_i is the fertilizer use per acre in the sub region i that is equal to $(x_{i2})\,/\,(x_{i1})$ with x_{i2} the aggregate fertilizer use in this subregion. Moreover it holds that $\omega_j^{ki} = (dx_{kj}/dt_i)\,/\,(dx_{i2}/dt_i)$ and $\zeta^{ik} = (dx_{k2}/dt_i)\,/\,(dx_{i2}/dt_i)$. Finally λ^i is the Lagrangian multiplier associated with the maximization of individual net surplus subject to (5.16). For details see Classen and Horan (2001).

- Dynamic context:

In the dynamic setup the structure of the input-based policy schemes in an expected value maximization context, applied to farmers, are the same as those defined in the previous subsection where these measures where analyzed under certainty, with the only modification that the dynamic multiplier ($\mu(t)$) should be replaced by the corresponding expected value ($\breve{\mu}(t)$).

6.2.3 Under asymmetric information

If the regulator has complete information about the farmers heterogeneity parameter Θ_i then, the input-based tax defined by problem (5.28) is given by:

$$t_i = \left[\frac{\partial D(e)}{\partial e} \frac{\partial e(e_i)}{\partial e_i} \frac{\partial e_i(\mathbf{x}_i(\Theta_i), \Theta_i)}{\partial x_{ij}} \right] \Big/ (1 + \tilde{\lambda})$$

which is less than the marginal damages, leading to extra emissions. In this case the tax payment is $T_i(\mathbf{x}_i(\Theta_i)) = t_i \mathbf{x}_i(\Theta_i)$ and the farmer's problem is defined as $\pi_i(\Theta_i) = \pi_i(\mathbf{x}_i(\Theta_i)) - T_i(\mathbf{x}_i(\Theta_i))$.

Under incomplete information the heterogeneity parameter is the adverse selection variable Θ_i and the associated input-based mechanisms are defined in terms of adverse selection models (Hansen, 2002). In this case the regulator defines the individual tax rate $t_i(\Theta)$ based on agent i's input use and revealed information regarding the type Θ_i of all farmers $(\overline{\Theta}_1, ..., \overline{\Theta}_n)$. This information allows the regulator to define the input level $\mathbf{x}_i^*(\overline{\Theta}_1, ..., \overline{\Theta}_n)$ of each agent so that any cheating agent is caught (Cochard, 2003). Therefore the expected input level and tax payment for farmer i as a function of the announced cost parameters of all farmers, given that other farmers pursue truth-telling strategies, are:

$$\mathbf{x}_i(\Theta_i, \Theta_{-i}) \quad \text{and} \quad t_i(\Theta_i) = t_i(\Theta_i, \Theta_{-i})$$

After following the same procedure as defined in the previous subsection, under incomplete information and under the simplifying assumption that farmer i's incentive scheme depends totally on the responses of the particular farmer, the input-based tax rate can be defined as:

$$t_i = \left[\frac{\partial D(e)}{\partial e} \frac{\partial e(e_i)}{\partial e_i} \frac{\partial e_i(\cdot)}{\partial x_{ij}} + \frac{\tilde{\lambda}}{1+\tilde{\lambda}} \frac{H(\Theta_i)}{h(\Theta_i)} \frac{\partial^2 \pi_i(\mathbf{x}_i(\Theta_i))}{\partial (\mathbf{x}_i)^2} \right] \Big/ (1+\tilde{\lambda}) h(\Theta_i)$$

Moreover, based on Xepapadeas (1997) the following input-based tax schemes can be defined for the observed input x_{ij}, either as a nonlinear or linear policy scheme respectively:

$$t_i(x^*_{ij}(\Theta_i)) = T_i(x^*_{ij}(\Theta_i)) = \pi_i(x^*_{ij}(\Theta_i)) - \pi_i(\Theta_i) \quad (6.12)$$

$$t_i(\overline{\Theta}, x_{ij}) = t(\overline{\Theta}) + \left[\frac{\partial \pi_i(x^*_{ij}(\overline{\Theta}))}{\partial x_{ij}} \right] (x_{ij} - x^*_{ij}(\overline{\Theta})) \quad (6.13)$$

where x^*_{ij} is the optimal level of observed input defining $\Theta_i(x^*_{ij})$. The linear taxes on observed inputs as defined by (6.13) induce both truthtelling $\Theta = \overline{\Theta}$ and the optimal use of the observable input $x_{ij} = x^*_{ij}(\Theta)$, thus the release of optimal individual emissions. Furthermore, (6.13) can be implemented using two approaches: a menu of linear tax schedules $t_i = \Gamma + \Delta x_{ij}$ from which farmers choose the one that fits to their type, or a tax schedule $t_i(x_{ij}, x^*_{ij}) = \psi + \delta(x_{ij} - x^*_{ij})$ imposed by the regulator to induce farmers cut input use to the recommended level corresponding to their type.

It should be noted that there will be no difference between the full information and constrained information optimal if both production and pollution functions are of the fixed proportions type (Millock et al., 2002).[23] According to Cochard (2003) such a revelation policy scheme is inapplicable due to the formidable amount of required information, implying that the scheme should be modified to the second best that does not require private information or even relay on uniform tax rates. Nevertheless, the author underlines that under special cases the first-best solution can be achieved despite the fact that the regulator is uninformed of the polluting agents' type Θ_i. This requires that the expression (6.10) is independent of i or j's, permitting the regulator to define purely individual schemes, consist-

[23] For example the linear functions can take the following form: $E[D(e,\eta)] = de + \eta$, $e = \sum_{i=1}^{n} ce_i + \epsilon$ and $e_i = \sum_{j=1}^{m} s_j x_{ij} + \alpha_i$, where d, c and s_j are fixed values and same across the population. Therefore the tax rate is given by $t_j = dcs_j$ and it is uniform across polluters (Cochard, 2003).

ing of a set of tax rates t_{ij} that are uniform across polluting agents, though they are different across inputs.[24]

6.3 Output-based Schemes

Within the price-based approaches the regulator can also impose taxes on outputs that are closely related to emissions but often imperfectly correlated with them (Schmutzler and Goulder, 1997). The idea behind the implementation of such measures is that the regulator can rely on output quantities to receive information about the emission levels and thus impose output taxes to control nitrate emissions (Schmutzler, 1996). Thus output taxes have a Pigouvian role (Cremer and Gahvari, 2002), they can substitute emission taxes and thus induce first-best outcomes. It should be noted in the context of the EU policies that land-set-aside is primarily a policy instrument to control output supply which at the same time can be beneficial to the control of agricultural pollutants (Kampas and White, 2004).

Nevertheless, it should be noted that output-based schemes have been shown to be inefficient instruments in the long-run (Kampas and White, 2004).

Even though output taxes would be more sensible to be applied in sectors where small owner-managed operations are dominant (i.e. agricultural sector), their undeniable disadvantage is that they can only influence emissions insofar they depend on output level encouraging only a very specific kind of pollution reduction (Schmutzler, 1996). Moreover, if the regulator cannot measure output precisely then the mechanism to nitrate leaching control cannot be based on ex post output, since farmers are the residual claimants and first handlers of the harvest crop and can understate their output either by consuming it directly or by misrepresenting actual output (Chambers and Quiggin, 1996). Finally, input substitution, employing different technologies and abatement imply that a given level of output may result in different levels of emissions and thus the considered equivalence between output and emission taxes breaks down (Cremer and Gahvari, 2002).

Nevertheless, output taxes may be a useful tool in cases where

[24] This can be simply represented by $(t_{11} = t_{21} ... = t_{n1}, t_{12} = t_{22} ... = t_{n2}, ..., t_{1m} = t_{2m} ... = t_{nm})$ but $(t_{i1} \neq t_{i2} ... \neq t_{im})$.

input taxes induce substitution from the targeted, less dangerous input to a non-targeted, environmentally dangerous substance, since they can indirect reduce both inputs even though they do not induce necessarily the most desirable adjustment (Schmutzler, 1996).

6.4 Ambient-based Schemes

In reality neither individual discharges nor individual productive choices are directly observed in a NPS pollution problem, rendering thus emission-and input-based schemes inadequate for regulating NPS emissions in a effective way. Nevertheless, NPS pollution problems can be handled through policy schemes based on ambient pollution which is observable or can be measured at a reasonable cost. Such ambient-based measures shift the location of monitoring from the choices of agents suspected of contributing to environmental degradation, to the environmental media (Horan et al., 1998) and have been proposed as a means of reducing administrative information requirements and monitoring costs associated with NPS pollution control, and eliminating polluters' incentives for moral hazard (Horan et al., 2002).

For the case of agricultural NPS pollution an ambient policy scheme is in principle imposed on the collective nitrate leaching generated by farms and measured at some spatial locations - in a static setup - or on the nitrate pollution accumulated at a particular receptor point (i.e. lake, groundwater aquifer) - in a dynamic context. Such schemes can be either imposed collectively to all farmers within a geographical region or randomly on one or more farmers. Moreover, ambient instruments can involve uniform or farm-specific fines (subsidies) and can be characterized as budget or non-budget balancing schemes. Even though, ambient instruments have considerable theoretical appeal because they would seem to reduce the complexity of policy design relative to input-based incentives (Shortle et al., 1998), they have many and notable drawbacks outlined by economic literature that can undermine their effectiveness or even their implementation.

In this section the available theoretical developments on ambient-based instruments in the context of certainty and uncertainty, as well as in a static and dynamic framework are provided.

6.4.1 Under certainty

- Static context:

Under an ambient-based scheme the total tax payment is $T_i(e)$ and the after-regulation payment of farmer i is $\pi_i(\mathbf{x}_i) - T_i(e)$. The associated nm first-order-conditions are:

$$FOC_{x_{ij}} : \frac{\partial \pi_i(\mathbf{x}_i)}{\partial x_{ij}} - \frac{\partial T_i(e)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}} = 0 \qquad \forall i, j \qquad (6.14)$$

After comparing with (5.7), the condition that needs to be satisfied by the ambient-based instrument, so that profit maximizing, regulated farmers apply the socially optimal vector of input choices $\mathbf{x}_i^*$ is given by:

$$\frac{\partial T_i(e)}{\partial e} = \frac{\partial D(e)}{\partial e}$$

implying that at the margin the optimal individual regulatory scheme must be equal to marginal social damages.

In the environmental economics literature five variants of the ambient-based instrument are usually met. In particular:

1. Group incentive instrument:

It involves either a tax (t_i) plus a lump-sum fine (F_i) if the optimal pollution level e^* is exceeded, or a subsidy (κ_i) and a bonus (B_i) otherwise.

$$T_i(e) = \begin{cases} t_i(e - e^*) + F_i & \text{if} \quad e > e^* \\ \kappa_i(e - e^*) - B_i & \text{if} \quad e \leq e^* \end{cases} \qquad (6.15)$$

where under optimal choose of the ambient instrument

$$t_i = \kappa_i = \frac{\partial D(e)}{\partial e} \qquad (6.16)$$

Actually this is a general scheme combining all the following ambient-based schemes, which result under simplifying assumptions.

2. Tax / subsidy scheme:

By setting both F_i and B_i equal to zero the ambient tax / subsidy scheme is obtained that combines a tax (t_i) and a subsidy (κ_i) depending upon whether the total pollution level is above or below the optimal level (Cochard et al., 2004). In particular, a subsidy is provided if the group total is below the optimal level and a tax is imposed if it is above the optimal level and each individual pays the entire damage (Spraggon, 2002).

$$T_i(e) = \begin{cases} t_i(e - e^*) & \text{if} \quad e > e^* \\ \kappa_i(e - e^*) & \text{if} \quad e \leq e^* \end{cases}$$

However, the abatement decisions of one polluter affect (reduce) the tax payments of the other polluters, giving an incentive to form coalitions among to polluters in order to increase abatement above its individually optimal level (Hansen, 1998). If all agents coordinate on the collusive outcome then the group total (ambient pollution) is inefficiently lower than the optimal level and the regulator will be forced to pay a large amount of subsidies to the agents (Spraggon, 2002). Under coalition formation ambient tax / subsidy mechanisms are inefficient.

3. Tax scheme:

By setting F_i, B_i and κ_i equal to zero the ambient tax scheme is obtained. Such a scheme can be treated as a simple Pigouvian tax on deviations from optimal aggregate pollution, since it involves only a tax if the optimal pollution level is exceeded (Cochard et al., 2004), while no subsidy is provided when the group total is below (Spraggon, 2002).

$$T_i(e) = \begin{cases} t_i(e - e^*) & \text{if} \quad e > e^* \\ 0 & \text{if} \quad e \leq e^* \end{cases} \tag{6.17}$$

The particular mechanism alleviates the collusion problem defined previously and optimality can be achieved, since when $e \leq e^*$ the expected tax reduction from lowering the aggregate pollution level is zero, making it not optimal for coalitions of farmers to overabate in order to reduce pollution below the optimal level and thus receive a subsidy. Furthermore, since the coalition is sustained only if the net rent to members is positive the regulator can ensure that no coalition is formed and that the mechanism implements optimum, via a cutoff level $\bar{e}$ that is lower than (or equal to) the optimal pollution level and higher than the smallest cut-off point resulting in a negative payoff to all possible coalitions $\bar{e}_c$ (i.e. $\bar{e}_c < \bar{e} \leq e^*$), so that further damage reduction does not result in reduced tax payment (Hansen, 1998).

Finally, based on Cochard (2003) the regulator can take advantage of the fact that farmers form collusions in order to achieve the social optimum. In particular, the regulator can persuade farmers to cooperate so that to cut individual input choices and thus emissions to the social optimal, by committing himself not to implement any regulatory measure (like the tax described previously) if they form the grand coalition. This is actually a type of voluntary environmental agreement (described in the following sections), which by relying on the background threat of a tax offers to the group of polluters the chance to take into account the externality they generate.

4. Subsidy scheme:

By setting F_i and t_i equal to zero the ambient subsidy scheme is obtained, that involves a subsidy and a lump-sum bonus in case the total pollution level is equal or below the optimal level (Cochard et al., 2004). The subsidy payment depends on the deviations between the desired and measured ambient concentration levels at a "receptor point", where the smaller the deviations, the greater the distributed subsidy (Xepapadeas, 1991).

$$T_i(e) = \begin{cases} 0 & \text{if} \quad e > e^* \\ \kappa_i(e - e^*) - B_i & \text{if} \quad e \leq e^* \end{cases}$$

This instrument is undesirable since if the group of farmers is able to collude and proceed in overabatement then the regulator is likely to face budget constraints. (Spraggon, 2002).

5. Group fine scheme:

Whenever the actions of the agents result in an aggregate pollution level above the cut-off/optimal level, a lump-sum fine paid by all the agents (Camacho and Requate, 2004).

$$T(e) = \begin{cases} F_i & \text{if} \quad e > e^* \\ 0 & \text{if} \quad e \leq e^* \end{cases}$$

The purpose of this fixed penalty - since is independent of the deviation between observed and desired pollution levels - is to counterbalance any gains from free riding if individual polluters emissions which are unobservable exceed the desirable level (Xepapadeas, 1999). It is notable that collective penalties can be used as a credible threat in negotiations of policy schemes, to induce self-reporting and early containment of damage (Millock and Zilberman, 2004).

Existing Ambient - type instruments:

Even though no actual implementation of an ambient-based instruments to regulate NPS pollution, as described by theory, has reported until now (Cochard et al., 2004) there are practices having similarities to ambient-based schemes. Regarding collective punishments real examples are certainly found in the military and schools, but environmental ambient taxes are difficult to find since it is seen as a quite drastic form of policy (Millock and Zilberman, 2004). Nevertheless, as examples of collective punishment can be considered the California drainage water policy that threatened to stop water supply unless farmers had cleaned up their site and developed alternatives to the drainage canal, as well as the threat to increase the land tax in Florida for all the farmers if the aggregate phosphorus reduction goal is not met (Millock and Zilberman, 2004). In Denmark, under the Aquatic Environment Action plan of 1987, there is an overall target of a 50% fall in nitrogen leaching (OECD, 1993). In the same context, the Netherlands set a target that nitrate concentrations in groundwater beneath farmland should not exceed 50mg/l at a depth of two meters below water table in areas where groundwater can be used for drinking water (OECD, 1993).Moreover, based on Xepapadeas (1999) Austria has established groundwater protection zones in which if the water quality is reduced then farmers have to comply with certain management practices or change land use and the Dutch water pollution charge involves a fixed tax payment for households and small firms, which is independent of their actual emissions (Xepapadeas, 1999). Finally, there is the experience in Germany, Thailand and Japan of "industrial associations for specific types of industries or for specific locations which are given a chance to attain their own a certain ambient level of water or air quality, otherwise charges or even a direct regulation can be imposed by the government regulatory agency" (Franckx, 2002).

These ambient instruments, first introduced by Segerson (1988), can be designed in terms of farm-specific/input basis.[25] Neverthe-

[25] Moreover, such ambient schemes could be applied spatially, implying that each production (soil) zone faces its own unique tax scheme if the groundwater nitrate concentration at an observation well site within the zone exceeds the desired level, and producers

less, under $m > 1$ productive choices all these ambient-based forms are overdetermined since there is not a single tax (subsidy) rate and lump-sum fine (bonus), satisfying all the equations necessary to guarantee that the Cournot-Nash equilibrium is efficient (Horan et al., 1998). In particular they are in general overdetermined for two reasons: (i) the single tax (subsidy) rate cannot correct for each input's marginal contribution to ambient pollution levels, and (ii) each input has a different marginal contribution to damages (Horan et al., 2002). However, this problem could be handled either if the covariance between marginal damages and marginal ambient pollution was zero for each input for each farmer $\left(cov(\frac{\partial D(e,\eta)}{\partial e}, \frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}) = 0\right)$ or if agents made a single production choice ($m = 1$) (Horan et al., 2002).

For the "small number regulation problems" , NPS pollution is characterized by heterogeneity among agents. To design and implement farm-specific or spatial ambient instruments is highly informationally demanding since their correct specification requires firm information (Hansen, 2002). Hence:

> The implementation of an ambient tax requires considering information and monitoring costs, implying that there is a trade-off between costs and tax mechanism design (Larson et al., 1996).

This is why ambient-based taxes are applied symmetrically across individuals, since symmetric taxes potentially reduce the informational burden. Therefore, Segerson's ambient schemes are modified to the second best and uniform ambient taxes (subsidies) are imposed (paid) on every potential polluter once measured ambient levels exceed (fall short) some desired cutoff level, by monitoring only the ambient level of pollutant at some receptor points and without seeking information about the productive choices of individual dischargers (Xepapadeas, 1995). It should be noted that uniform ambient taxes based on ambient water quality can achieve an efficient level of non-point pollution, under heterogeneous farmers if and only if marginal benefits of abating pollution are constant (Larson et al.,

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within a soil zone face the same tax (Fleming and Adams, 1997). However, the spatial tax should take into account that some local nitrate is generated by a source upstream in the aquifer (Fleming and Adams, 1997).

1996).

In the environmental economics literature two further categories of policy schemes to regulate NPS pollution can be distinguished:

1. Damage based schemes
2. Linear or nonlinear state-dependent schemes

that both can satisfy the efficiency conditions as a Cournot-Nash equilibrium regardless of the dimensionality of agents' productive choices (Horan et al., 1998).

Under damage based schemes the base of the mechanism is shifted from a measure of polluter actions (ambient pollution) to a measure of the effects of these actions, that is, environmental damage (Hansen, 1998). A damage based version of the ambient tax decentralizes the planning problem to polluters, contrary to the previous mechanisms where the whole planning problem was held by the regulator, and requires solving the regulator's information problem (Hansen, 1998) and handling the multiple dimensionality of polluters' choice set (Cochard et al., 2004). The regulator needs to know only the damage function (Hansen, 1998) and the need to acquire knowledge of the polluters' production functions even when the damage function is non-linear is eliminated (Hansen, 2002). The total tax payment is given by:

$$T_i(e) = t_i D(e)$$

where after comparing regulator's optimality condition (5.7) with agents' FOCs it is evident that the tax rate is one ($t_i = 1$) (Hansen, 1998), implying that the *damage based ambient tax* gets the following simple form:

$$T(e) = D(e)$$

where the associated FOC corresponds to the FOC for optimum (5.7), achieving thus the efficient outcome as a Cournot-Nash equilibrium (Horan et al., 1998). The tax payment is equal to total damages providing agents an incentive to consider at the margin the impact of each input on social damages (Horan et al., 1998). Farmers perceive that an increase in tax payment corresponds to the increase in damage and they are automatically penalized for the damage caused by increased emissions (Hansen, 1998). Actually this scheme is a *state dependent, nonlinear tax* suggested by Horan et al.

(1998), Hansen (1998) and Horan et al. (2002), which is applied uniformly across farmers and eliminates the regulator's need to calculate firm specific tax rates.

Moreover, the damage based scheme - after a slight modification proposed by Hansen et al. (1998) - reduces or even handles totally the probability of coalition formation. Therefore the modified scheme is:

$$T(e) = \begin{cases} D(e) & \text{if} \quad e > e^* \\ 0 & \text{if} \quad e \leq e^* \end{cases}$$

where no tax reduction is provided from lowering ambient pollution below the optimal level.

However, even though the damage based mechanism solves a serious information problem faced by the regulator via decentralization:

> Shifting the base of the mechanism from ambient concentration to environmental damage weakens the instrument since it introduces the possibility that the optimum Nash equilibrium is not stable and the optimum is only implemented in non-dominant Nash strategies (Cochard et al., 2004).

The efficiency of the mechanism is questionable when polluters' optimal emissions are interdependent and when polluters have limited information about their strategic environment (Cochard et al., 2004; Hansen, 2002). In particular, under a tax rate defined as a function of ambient pollution level, the marginal tax effects of abatement perceived by each farmer become dependent on other farmers' optimal abatement actions. Thus agent i's $FOCs$ depend on other agents' abatement efforts and farmer i can only identify his own profit ash-equilibrium strategies (Cochard et al., 2004) when the mechanism is initially imposed since this requires knowledge of other agents' Nash-equilibrium strategies (Hansen, 1998). The mechanism only implements equilibrium in Nash strategies, a fact that may reduce stability even with Nash conjectures (Hansen, 2002). Concluding, even though this tax scheme does not depend on firm level information, its equilibrium may be unstable and it may induce polluters to base their actions on non-Nash conjectures causing equilibrium to stray from optimum (Hansen, 2002).[26]

[26] It is worth mentioning that interdependences between farmers are also triggered

Nevertheless this problem can be handled through the implementation of a state dependent, linear ambient tax scheme (Cochard, 2003). Actually this scheme is the modified Segerson's linear scheme where the tax payment is proportional to aggregate pollution and not dependent on the deviation $(e - e^*)$. Therefore the total tax payment is $T(e) = te$ and the tax rate is given by:

$$t = \frac{\partial D(e)}{\partial e} \quad \forall i$$

The ambient tax rate imposes the full social marginal damage on each polluter, who thus has an incentive to cut back individual emissions to the socially optimal level, yielding the first best aggregate emission level (Millock and Zilberman, 2004). Both the tax rate (t) and tax base (e) are state dependent (Horan et al., 1998). As with the nonlinear tax scheme the linear tax is applied uniformly across all farmers (Shortle et al., 1998). It is clear that the instrument evaluated at the efficient level of input use (i.e. $T(e) = \frac{\partial D(e^*)}{\partial e} e$) is able to provide the socially optimal incentives, but this requires the computation of e^* that necessitates complete information (Cochard, 2003).[27]

Regarding the issue of the weakness of the Nash equilibrium concept due to the collective nature of ambient instruments, Cochard (2003) has identified the conditions under which a general ambient-based scheme provides a Nash equilibrium in dominant strategies. It is clear from the previous analysis that the $FOCs$ of each regulated agent, defined by (6.14), should be independent of the other agents' decisions. This implies that both $\frac{\partial T_i(e)}{\partial e}$ and $\frac{\partial e(e_i)}{\partial e_i} \frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}$ must not be affected by the other farmers' decisions. This requirement is satisfied if the tax scheme is linear - such as the linear, state dependent tax scheme defined previously - and if in addition the aggregate emission function is also linear.

- Collective nature and non-budget balancing feature of

under input-based schemes, which are purely individual schemes and not collective as the ambient-based schemes (Cochard, 2003).

[27] If the initial assumption that the set of farmers is optimal is relaxed then a lump-sum tax is needed to be used in addition to both linear and nonlinear ambient taxes to ensure that extra-marginal farmer does not expect to earn positive after tax profits. On the other hand, a lump-sum subsidy is necessary for marginal and intramarginal firms (Shortle et al, 1998).

ambient-based schemes

An ambient-based scheme is characterized as collective instrument if tax (subsidy) payments are imposed to all the farmers within the regulated geographical region whenever aggregate emissions exceed (fall short) the optimal level.

An alternative definition of the collective ambient subsidy scheme described previously is provided by Xepapadeas (1997), where no subsidy is provided when the group total is below the optimal pollution level, making coalition formation unprofitable.[28] Under this instrument each agent receives a subsidy if there is no deviation from the desired level e^*, while if a deviation is observed then the provided subsidy is reduced via collective penalties. Therefore the total subsidy payment is given by:

$$T_i(e) = \begin{cases} \phi_i RSB + k_i \Gamma(e) - \delta & \text{if} \quad e > e^* \\ \phi_i RSB & \text{if} \quad e = e^* \end{cases}$$

If the target is attained then the provided subsidy is equal to the farmer i's abatement share of total optimal abatement $\left(\phi_i = \mathbf{x}^a_{ij} / \sum_{i=1}^n \mathbf{x}^{a*}_{ij}\right)$.[29] In this case the regulator actually divides the real social benefits due to the reduction of aggregate emissions (Spraggon, 2002) among the firms in a form of a subsidy. On the other hand, if deviations are observed collective penalties are imposed to provide the desired outcome, reducing the provided subsidy. The reduced subsidy is the initial subsidy minus a part of the social value of excess pollution[30] ($k_i\Gamma(e)$) and a fixed penalty (δ) independent of the size of deviations (Xepapadeas, 1991).[31] In order to ensure that the particular ambient subsidy scheme provides the socially-desired solution, the regulator must choose carefully the values of k_i and δ to eliminate any benefits from shrinking (reduced abatement compared to optimal abatement effort).

However, such an ambient-based scheme may not be entirely satisfactory solution since the collective penalties are triggered even by

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[28] The same scheme is provided in a dynamic context by Xepapadeas (1991).

[29] Where $RSB = SB - TR$, with SB the social value of abatement and TR the transaction costs between the regulator and polluting agents. For details see Xepapadeas (1997) page 152.

[30] Where $\Gamma(e) = -\frac{\partial D(e)}{\partial e}(e - e^*)$ is the social cost of excessive pollution.

[31] Actually the collective penalty is a combination of an ambient tax type plus a lump-sum fine type.

small deviations from desired ambient levels, placing financial strain on the whole group of farmers especially when abatement efforts are close to the desired level and gains from free riding are not substantial (Xepapadeas, 1991). Furthermore individuals who take costly actions to improve their environmental performance may resent the idea of finding themselves liable to penalties due to environmental shirking on the part of others, introducing a problem regarding the political and ethical acceptability of ambient schemes which rely on collective punishments (Shortle et al., 1998).

In this context random fining mechanisms have been proposed, where only one of the agents (or more) is randomly chosen and charged the total amount of the social cost of the reduced abatement if the observed pollution level exceeds the social desirable one (Camacho and Requate, 2004). Therefore the total subsidy payment is given by:

$$T_i(e) = \begin{cases} \phi_i RSB & \text{if} \quad e > e^* \quad \text{with prob. } 1 - \zeta_i \\ \phi_i RSB + \Gamma(e) & \text{if} \quad e > e^* \quad \text{with prob. } \zeta_i \in (0,1) \\ \phi_i RSB & \text{if} \quad e = e^* \end{cases}$$

where ζ_i is the probability that the farmer i is penalized with $\sum_{i=1}^{n} \zeta_i = 1$. This assumes that the agent(s) is (are) wealthy enough to pay the fine (Xepapadeas, 1991), which however high not be the case for the majority of agents operating in the agricultural sector.

These collective and random punishment mechanisms are non-budget balancing as Xepapadeas (1991) has indicated (Camacho and Requate, 2004). A scheme is non- budget balancing (NBB) if total subsidy payments to polluters are different from society's value of abatement or equivalently if the total tax payments of polluters are not equal to the social cost of pollution. Each farmer is liable for the whole environmental damage (Cochard, 2003), leading to a multiple of damage costs collected (Herriges et al., 1994). On the other hand, a budget-balancing scheme (BB) is such that the environmental damage is shared between the members of the group and it might produce a more acceptable solution by reducing the financial burden on firms even though it may increase regulator's burden (Xepapadeas, 1991).[32]

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[32] According to Spulber (1988) the budget-balancing feature implies that the sum of

Under the proposed budget-balancing, random fine mechanism if the ambient pollution exceeds the optimum at least one randomly chosen polluter is penalized (Xepapadeas, 1991) even if he does not constitute the source of the excess pollution, implying that its abatement effort has not been monitored (Herriges et al., 1994). The imposed penalty has two components: the subsidy is removed totally, and an additional fine[33] (F_g) is imposed. The collected penalties minus the social welfare loss due to the higher ambient concentration ($\Gamma(e)$) are redistributed to the remaining firms, justifying the budget-balancing character of the mechanism, since it allocates the total subsidy to the members. If $\left[\phi_g RSB + F_g + \Gamma(e)\right] > 0$ then the rest farmers receive an increased subsidy compared to the subsidy ($\phi_i RSB$) corresponding to the no shrinking case (Xepapadeas, 1991). Hence the total payment under the budget-balancing ambient scheme is:

$$T_i(e) = \begin{cases} \phi_i RSB + \\ \phi_{ig}\left[\phi_g RSB + F_g + \Gamma(e)\right] & \text{if} \quad e > e^* \quad \text{with prob } 1-\zeta_i \\ -F_g & \text{if} \quad e > e^* \quad \text{with prob } \zeta_i \in (0,1) \\ \phi_i RSB & \text{if} \quad e = e^* \end{cases}$$

where g is the farmer penalized and $\phi_{ig} = \mathbf{x}_{ij}^{a} / \sum_{i=1}^{n-1} \mathbf{x}_{ij}^{a*}$. However, the successful application of this instrument requires participation of all dischargers in the region and there might be some legal problems in enforcing an ambient measure involving random penalties (Xepapadeas, 1997). Finally, according to Cochard (2003) no BB scheme could be designed to induce agents perform the social optimal level of abatement as a Nash equilibrium, only a NBB scheme can deal with such an incentive problem.[34]

- Dynamic context:

There are ambient taxes developed in a dynamic setting, taking into account the dynamic process of pollution accumulation and thus

transfers across agents sums to zero.

[33] The fine is such that the Pareto optimality requirement is satisfied and the scheme is efficient (Xepapadeas, 1991).

[34] Even though it has been assumed that the number n of farmers is optimal it is worth mentioning that a BB scheme was also developed by Horan et al., (1998) in order to handle suboptimal entry/ exit.

the stock nature of NPS nitrate pollution (Legras, 2004). In particular the dynamic ambient based scheme is given by:

$$\phi(\Delta S) = \phi(S(t) - S^*(t)) \quad \text{with } \phi' > 0, \phi'' \geq 0$$

where charges per unit of deviation between socially desired and observed ambient pollutant level are imposed at each instant of time on every potential discharger and these charges depend on the strategies of the polluters (Xepapadeas, 1992). On the other hand if no deviation is observed then no penalties are imposed, implying that if $(S - S^*) < 0$ then $\phi(\Delta S) = 0$.

The after-tax payoff function of farmer i is given by:

$$\max_{\mathbf{x}_i} \int_0^\infty \exp(-rt) \left\{ \pi_i(\mathbf{x}_i, S) - \phi(\Delta S) \right\} dt$$

where the designed intertemporal incentive schemes are analogue to Segerson's static Pigouvian tax, accounting of the endogenous nature of the externality and are appropriate for the regulation of water quality in a water body polluted by agriculture (Xepapadeas, 1992).

The incentive scheme under the myopic strategy, where dynamics of the pollutant accumulation are ignored, achieves the social optimum if it charges the full social cost of deviations between observed and desired ambient concentration levels at any instant of time (Xepapadeas, 1992). Hence the myopic dynamic ambient tax is equal to:

$$\tau(t) = -\mu(t)(S(t) - S^*(t))$$

Moreover, the fact that the policy scheme depends on individual emissions via the observed stock $S(t)$, introduces strategic interaction among agents. Therefore under an OL and FB information structure the time flexible efficient tax scheme is given respectively by:

$$\tau^{OL}(t) = -\mu(t)\,(r + b)\,(S(t) - S^*)$$

$$\tau^{FB}(t) = -\mu(t) \left[(r + b) - \sum_{j \neq i}^{n-1} \frac{\partial e_j}{\partial \mathbf{x}_j} \frac{\partial \mathbf{x}_j}{\partial S} \right] (S(t) - S^*(t))$$

These schemes are applied uniformly across farmers. and it is obviously that the charge depends on (i) pollutant's shadow cost (μ), (ii) discount rate (r), (iii) natural pollution decay rate (b), and (iv) information structure (Xepapadeas, 1992). As agents adopting the FB behavioral rule tend to over-apply inputs and consequently to overemit due to their perception that the other agents will emit less to counterbalance the increase in the stock pollution, the ambient tax has to be more stringent for the feedback formulation (Legras, 2004). Obviously the required ambient tax under the OL information structure is less than the required tax rate under FB (Xepapadeas, 1992).

It is notable that sometimes NPS pollution can be attributed to polluting activities that go back many years, implying that in each period active polluters are not fully or at all responsible for the current pollution level and that responsible agents may have disappeared (Cochard , 2003).[35] Therefore, even if their current input choices and emissions are optimal farmers can be asked to pay an ambient tax due to a deviation from the optimal pollution stock path observed in the present but originating in past overdischarges (Legras, 2004). Farmers keep paying the charge for the whole adjustment period to the optimal path (Xepapadeas, 1992).[36]

To balance the budget of the present schemes a system of fines and rewards is needed at any instant of time such that long-term efficiency is attained and the present value of net payments equal to the present value of social cost of deviations (Xepapadeas, 1992). The dynamic BB ambient schemes are defined as the static schemes, with the only difference that the penalty ($\Gamma(t)$) which depends of the size of deviations is given by $\Gamma(t) = \mu(t)(S(t) - S^*(t))$, that is the valuation of the excess concentration of the pollutant. The dynamic shadow cost of the pollutant concentration $\mu(t)$ can be either regarded as the subsidy per unit of reduction of net emissions or as the tax per unit of net emissions (Xepapadeas, 1991). However, the optimal abatement level is not always the unique Nash

[35] Changes in observed conditions may have little relationship to contemporary actions since nitrates and pesticides may take years to move from fields to wells (Shortle et al., 1998).

[36] If the regulator ignores the dynamic processes of pollutant accumulation on policy design and defines a policy scheme of the type: $\phi(t) = \lambda(t)(S(t) - S^*(t))$, where $\lambda(t)$ is the Lagrangean multiplier and not the Hamiltonian costate variable, then inefficiencies will be observed in the long-run.

equilibrium, since other Nash equilibria might exist characterized by over- or under-abatement due to the agents perception that the other agents will proceed in shrinking if they adopt the optimal policy.

Finally, the successful application of such a dynamic BB scheme has highly informational requirements since the regulator needs to know production and abatement technologies, damages from ambient pollutant concentration and the characteristics of the pollutant accumulation process (Xepapadeas, 1991).

6.4.2 Under uncertainty

- Static context:

Under a general ambient-based scheme - given by (6.15) in expected values - the farmer i's after-regulation payoff is (Spraggon, 2002):

$$\begin{aligned} E\{\pi_i\} &= E\{\pi_i(\mathbf{x}_i,\theta)\} - [t_i(E\{e\}-e^*) + F_i](1-\Pr(e \le e^*)) \\ &\quad - [\kappa_i(E\{e\}-e^*) - B_i)](\Pr(e \le e^*) \end{aligned}$$

with $\Pr(e \le e^*)$ the individual agent's expectation that the realized ambient pollution level e is lower than the socially optimal value e^*. Following Horan et al. (1998) it is considered that no bonus is provided ($B_i = 0$) to facilitate the definition of t_i and F_i. Thus the expected profits can be written as:[37]

$$E\{\pi_i\} = E\{\pi_i(\mathbf{x}_i,\theta)\} - [t_i(E\{e\}-e^*) + F_i(1-\Pr(e \le e^*))]$$

The associated nm first-order-conditions of the regulated farmer are:

$$FOC_{x_{ij}} : E\left\{\frac{\partial \pi_i(\mathbf{x}_i,\theta)}{\partial x_{ij}}\right\} - t_i E\left\{\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial x_{ij}}\right\} - F\frac{\partial \Pr}{\partial x_{ij}} = 0 \qquad \forall i,j$$

After comparing with (5.15) the efficient tax (subsidy) rate and fine much be chosen such that the following equality is satisfied for all the farmers (i) and inputs (j):

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[37] Equivalently it could be assumed that $F_i = 0$, since it is logical to consider that both $t_i = \kappa_i$ and $F_i = B_i$.

$$t_i E\left\{\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial x_{ij}}\right\} + F_i \frac{\partial \Pr}{\partial x_{ij}} = E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right] \qquad \forall i, \forall j \tag{6.18}$$

Since the group incentive scheme (6.15) comprises ambient-based instruments suggested by Segerson (1988), the optimal tax (subsidy) rates and / or lump sum fines corresponding to each subcase can be defined via the condition (6.18) under the appropriate simplifying assumptions. In particular if:

$$F_i = 0, \quad t_i = E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right] \Big/ E\left\{\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial x_{ij}}\right\} \qquad \forall i$$

$$t_i = 0, \quad F_i = -\,E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right] \Big/ \frac{\partial \Pr}{\partial x_{ij}} \qquad \forall i$$

$$t_i = \text{arbitrary}, \quad F_i = \left\{-E\left[\frac{\partial D}{\partial e}\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial x_{ij}}\right] + t_i E\left\{\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial x_{ij}}\right\}\right\} \Big/ \frac{\partial \Pr}{\partial x_{ij}} \qquad \forall i$$

where the initial scheme is the ambient tax instrument, followed by the group fine scheme and the ambient tax / subsidy instrument.

Finally, by getting more precise since farmers proceed both in productive $\mathbf{x}_i^p$ and abatement $\mathbf{x}_i^a$ input choices, the tax rate and group fine that implement optimum as a Nash equilibrium under the ambient tax / subsidy scheme must be chosen so that (Hansen, 1998):

$$t_i = \left[E\left\{\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(\cdot)}{\partial e_i}\frac{\partial e_i(\cdot)}{\partial x_{ij}^p}\right\} - \left(\frac{\partial \Pr}{\partial x_{ij}^p}\Big/\frac{\partial \Pr}{\partial x_{ij}^a}\right) E\left\{\frac{\partial D}{\partial e}\frac{\partial e}{\partial e_i}\frac{\partial a_i}{\partial x_{ij}^a}\right\}\right] \Big/ \left[E\left\{\frac{\partial e}{\partial e_i}\frac{\partial e_i}{\partial x_{ij}^p}\right\} - \left(\frac{\partial \Pr}{\partial x_{ij}^p}\Big/\frac{\partial \Pr}{\partial x_{ij}^a}\right) E\left\{\frac{\partial e}{\partial e_i}\frac{\partial a_i}{\partial x_{ij}^a}\right\}\right] \qquad \forall i$$

$$F_i = \left\{-E\left[\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial a_i(\mathbf{x}_i)}{\partial x_{ij}^a}\right] - t_i E\left\{\frac{\partial e}{\partial e_i}\frac{\partial a_i}{\partial x_{ij}^a}\right\}\right\} \Big/ \frac{\partial \Pr}{\partial x_{ij}^a} \qquad \forall i$$

It should be noted that when Segerson type ambient-based schemes are considered which they can achieve the efficient solution for NPS as a Nash equilibrium with minimal monitoring or information gathering requirements (Horan et al., 2002), this occurs under very restrictive conditions (Shortle et al., 1998). In particular the required optimality conditions are satisfied with equality for all i farmers and j input choices under the assumptions of: risk-neutrality, identical beliefs about the distribution of stochastic events (Horan et al., 2002), linear damage function (Cochard et al., 2004),[38] single productive choices ($m = 1$) or absence of covariance between marginal damage and marginal ambient pollution $\left(cov(\frac{\partial D(e,\eta)}{\partial e}, \frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}) = 0\right)$ for all farmers and inputs if $m > 1$.

Nonetheless, according to Hansen (1998) the Nash-solution concept can be strengthened if the emission function is linear. Under this assumption Segerson's mechanisms imply constant tax rates that are not affected by other farmers' input choices (and emissions), providing farmers an incentive to adopt optimal input choices irrespective of others actions. Moreover, Segerson's mechanisms could be potentially more relevant for practical application if the regulator makes public the information he must elicit about farmers' profits and emission functions (Hansen, 1998). In such a case farmers have complete knowledge about other farmers actions, allowing them to calculate their own Nash-equilibrium strategy. An explicit or "cheap talk" agreement among farmers in order to play the known Nash-equilibrium strategies would be self-enforcing. However, even if the regulator could collect the required information, this information would have to be transmitted back to each polluter and such a dissemination is likely be costly and it is also unlikely that farmers would be able to process easily and accurately(Horan et al., 2002).

State dependent Schemes: Under m-dimensional input choices Segerson's ambient measures fail to implement the social optimum in dominant strategies, suggesting the implementation of regulatory schemes which depend on the state of nature, and satisfy the efficiency requirements under less restrictive conditions. Hence under risk neutrality the expected linear and nonlinear, state dependent ambient-based schemes are defined as:

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[38] Under a linear damage function the correct specification of the mechanism requires only knowledge of the damage function (Cochard et al., 2004), while under nonlinear damage function the regulator needs to have knowledge of each polluters' profit and emission function (Hansen, 1998).

$$E\{T(e)\} = tE\{e\} \quad \text{with} \quad t = E\left\{\frac{\partial D(e,\eta)}{\partial e}\right\} \quad \forall i \tag{6.19}$$

$$E\{T(e)\} = t = E\{D(e,\eta)\} \quad \forall i \tag{6.20}$$

The tax rate under the linear scheme is the total marginal environmental damage caused by changes in the ambient concentration, determined ex post given the ex ante efficient levels of inputs use and it is conditional on the realization of all random variables (Shortle et al., 1998). On the other hand, the nonlinear ambient tax scheme - or else known as damage based scheme - charges an amount equal to total expected damages, providing agents an incentive to consider the impact of each input on expected damages (Horan et al., 1998). Both instruments are ex ante efficient since they are determined after the realization of all random variables and are applied uniformly across all polluters (Shortle et al., 1998).

The regulator can also design an ambient "standards and pricing" mechanism that involves an environmental standard and a pricing version of the damage based tax mechanism in order to meet an expected damage level $\tilde{D}(e,\eta)$ corresponding to the standard $(\bar{e})$ that the regulator wants to meet (Hansen, 1998). Under this context the expected tax payment is:

$$T(e) = \lambda_t E\left[\tilde{D}(e,\eta) - D(\bar{e},\eta)\right]$$

where λ_t is the Lagrangean multiplier which can be adjusted by the regulator if the expected net tax payment is significantly different than zero. Even though temporary coalitions are possible under this scheme, when λ_t is less than the equilibrium value λ^*, they do not affect long-run equilibrium and may be welfare improving in the short-run. Finally, in equilibrium this mechanism corresponds to the nonlinear, state dependent scheme with $E\{D(\bar{e},\eta)\} = \lambda^* E\left\{\tilde{D}(e,\eta)\right\}$ (Hansen, 1998).

Risk-aversion: Due to the stochastic nature of ambient pollution the assumption of risk-neutrality can be dropped. According to Horan et al. (2002) with risk-averse farmers, an ambient tax cannot be longer used alone to attain the ex-ante efficient solution. The ambient-based scheme needs to be modified and include additional instruments that account for the risk associated with stochastic ambient tax.[39] Such an efficient mixed regulatory scheme consist of an ambient tax and input taxes, where the total tax payment is given by:

$$T_i(e) = t_{1i}(e) + t_{2i}(\mathbf{x}_i)$$

The variable part $t_{1i}(e)$ represents the efficient ambient tax as designed for risk-neutral polluters that is independent of any utility-related but not environmentally related risk effects. On the other hand, the variable part $t_{2i}(\mathbf{x}_i)$ is a risk-premium or reward to account for the utility-related risk effects that result from the stochastic ambient tax and which are associated with the use of each polluting input. The sign of $t_{2i}(\mathbf{x}_i)$ depends on the sign of the covariance $cov\left(E_i\left\{\frac{\partial U_i(\pi_i(\mathbf{x}_i,\theta))}{\partial x_{ij}}\right\}, \frac{\partial T_i(e)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right)$. However, such a scheme has extensive informational requirements since both the regulator and farmers need to know the other polluters utility functions.[40] Moreover, the monitoring requirements under risk aversion are increased since the regulator needs to monitor both the ambient pollution levels but also the individual input choices (Horan et al., 2002).

Separable abatement effects: state dependent ambient-based and variance-based measures

Based on Hansen (2002) the linear and nonlinear, state dependent tax schemes can be also defined under the specialized assumption that abatement input choices $(\mathbf{x}_i^a)$ have separable effects on ambient concentrations. In particular, abatement choices do not only affect the mean ambient concentration but also its variance, implying that the individual and ambient emission functions can be specified by:

$$e_i = \bar{e}_i(\mathbf{x}_i^a) + \sqrt{var_i(\mathbf{x}_i^a)\mathfrak{n}} \quad \text{with} \quad e = \sum_{i=1}^{n} e_i(\mathbf{x}_i^a, v_i) + v_{n+1}$$

[39] Horan et al. (2002), as well as Cochard (2003) provide a explanation why an additional measure is needed to implement social optimum, via the comparison of the regulators *FOCs* and the risk-averse farmer's *FOCs* under the regulation.

[40] It is reminded that under risk aversion farmers are supposed to be von Neuman-Morgenstern expected utility maximizers that choose input levels to achieve the maximum expected utility from profits given by $E_i\{U_i(\pi_i(\mathbf{x}_i,\theta))\}$.

where $v_1, ..., v_i, ..., v_{n+1}$ are random variables independently distributed and $\mathfrak{n}$ is a stochastic variable following the normal distribution. The terms $\bar{e}_i(\mathbf{x}_i^a)$ and $var_i(\mathbf{x}_i^a)$ denote the mean and variance of farmer i's contribution to the ambient concentration level as a function of the polluter's abatement input set.

In this case the optimal tax rate under the *linear, state dependent ambient scheme* is equal to:[15]

$$t_i = q + s\frac{\partial var_i(\mathbf{x}_i^a)/\partial x_{ij}^a}{\partial \bar{e}_i(\mathbf{x}_i^a)/\partial x_{ij}^a}$$

where q is the damage effect of a unit reduction in mean ambient concentration and s the damage effect of the corresponding change in the variance of ambient concentration.

It is notable that the optimal linear, state dependent tax scheme is no longer uniformly applied across polluters since the ratio of abatement effects $\left((\partial var_i(\mathbf{x}_i^a)/\partial x_{ij}^a)/(\partial \bar{e}_i(\mathbf{x}_i^a)/\partial x_{ij}^a)\right)$ on mean and variance of ambient concentration differs across farmers (Hansen, 2002). However the specification of such tax rates is not possible since it requires the additional knowledge of each individual farmer's optimal abatement level, as well as the mean and variance functions (Hansen, 2002). Therefore, the linear tax scheme can be modified to the second-best, which does not account of variations between farmers but it may be feasible in terms of required information. Finally, regarding the *nonlinear, damage based ambient tax* the regulator can implement the optimum even though he does not know the mean and variance (Hansen, 2002).

Variance-based measures: When abatement has two effects on ambient concentration a *variance-based tax scheme* that utilizes a separate, linear tax rate for each effect is suggested. Based on Hansen (2002) the tax mechanism is given by:

$$T(e) = q\,[\bar{e}_i(\mathbf{x}_i^a)]^e + s\,[var_i(\mathbf{x}_i^a)]^e$$

where $[\bar{e}_i(\mathbf{x}_i^a)]^e$ and $[var_i(\mathbf{x}_i^a)]^e$ represents the estimated mean and variance respectively, of the stochastic process generating ambient pollution that can be estimated based on the series of ambient pollution observations over time.

[15] For details regarding the procedure followed in order to define this particular tax rate see Hansen (2002).

The superiority of the measure compared to the Segerson's tax scheme and the damage-based tax lies in the fact that it implements the optimum in dominant strategies without firm level information (versus Segerson's tax),[41] ensures that Nash conjectures are rational and that optimal input and emission strategies are independent of other farmers' inputs and emissions choices (versus damage-based tax). Indeed, the Segerson's ambient tax rate is overdetermined when $m > 1$ since several tax dimensions (here mean and variance) are regulated jointly through a single tax, while the damage-based tax may induce non-Nash conjectures since the abatements effects on variance are incorporated through a nonlinear tax on ambient concentrations. Furthermore, under certain types of stochastic independence of farm emissions - due to common stochastic effects such as weather and NP emission situations like nitrate leaching from farms - the variance-based mechanism can be properly adjusted so that the implementation of the optimum is retained in dominant strategies.[42] All these features make the variance-based instrument more applicable than the other tools.

Ambient-based schemes under heterogeneous expectations between the regulator and farmers

All the previous measures are defined under homogenous expectations between farmers and regulator, regarding the distribution of stochastic events influencing ambient pollution level. However, as mentioned before[43] expectations between farmers (E_i) and regulator (E) can be different since both sides have different priors[44] regarding the joint distribution of all random variables, and in turn different priors on the ambient concentrations that result from a given level of production and abatement activity (Cabe and Herriges, 1992).

Under this context the tax rate under Segerson's farm-specific, ambient tax scheme is given as:

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[41] The optimal tax rates q and s neither vary across firms nor depend on firm level information (Hansen, 2002).

[42] If the regulator observes the stochastic variable that represents multiplicative common effects and its distribution then specification of correct uniform tax rates is feasible (Hansen, 2002).

[43] In the definition of the static NPS model under uncertainty.

[44] It is reminded that $g_i(\mathbf{u},\omega,\eta\,|\Omega_i)$ is the density function for the each farmer i and $h(\mathbf{u},\omega,\eta\,|\Omega^*)$ for the regulator (Horan et al., 2002).

$$t_i = E\left\{\frac{\partial D(e,\eta)}{\partial e}\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right\}\Big/ E_i\left\{\frac{\partial e(e_i)}{\partial e_i}\frac{\partial e_i(\mathbf{x}_i)}{\partial x_{ij}}\right\}$$

It is obvious that if producers and the regulator shared the same prior means regarding the transfer coefficients then the marginal tax rate would be simply the marginal damage cost from increased concentration (Cabe and Herriges, 1992) and the social optimum would be implemented. However, under discrepancy between the prior beliefs ($E \neq E_i$) incentives designed on the assumption of homogeneous expectations do not have the desired properties (Shortle et al., 1998) and inefficiencies are expected. In particular, if farmer i believes that the marginal contribution of his input choices to damages is small or even zero, then the ambient taxes either have little or no impact on ambient pollution (Horan et al., 2002) or they have to be set large enough in order to efficiently reduce ambient pollution (Cabe and Herriges, 1992).[45]

Based on Horan (2002) the adjusted state-dependent linear and nonlinear ambient taxes are actually schemes (6.19) and (6.20) multiplied by:

$$\frac{g_i(\mathbf{u},\omega,\eta\,|\Omega_i)}{h(\mathbf{u},\omega,\eta\,|\Omega^*)} \quad \forall i$$

to correct for differences in expectations about stochastic events and uncertainty about environmental processes. In particular, this term transforms agents' expectations so that, ex-ante, they evaluate uncertainty in the same way as the regulator, according to the regulator's density function $h(\mathbf{u},\omega,\eta\,|\Omega^*)$.

A notable difference when we compare with the identical ambient schemes under homogeneous expectations, where the equality $h(\mathbf{u},\omega,\eta\,|\Omega^*) = g_i(\mathbf{u},\omega,\eta\,|\Omega_i)$ holds, is that under heterogeneous expectations $(h(\mathbf{u},\omega,\eta\,|\Omega^*) \neq g_i(\mathbf{u},\omega,\eta\,|\Omega_i))$ the state-dependent measures are no longer applied uniformly across farmers but they are designed farm-specifically (Horan et al., 2002). However, such a nonuniform modification has extreme informational requirements that may seriously limit the regulator's ability to design an ex ante efficient ambient tax scheme and the ability of farmers to achieve the efficient

[45] In the later case the tax rate may set at a level that profits are non-positive, terminating thus the suspect activity.

equilibrium, even if the tax rates are designed correctly. In particular, the regulator requires perfect information about each farmer's joint density function, which may not be feasible because polluters are not likely to report this information truthfully (Horan et al., 2002). Furthermore, he needs to know the prior distribution on the transport mechanism (Cabe and Herriges, 1992). Moreover, the realized values of all random variables must be known in order to compute the correction term of the state-dependent tax, raising an additional difficulty (Horan et al., 2002).

Finally, Cabe and Herriges (1992) suggest two solutions to mitigate the problems arising from expectations' discrepancy: the regulator either educates polluters about his conditional probability function for uncertain environmental relations, or proceeds in monitoring to acquire information on the physical processes influencing fate and transport of pollutants so that to update prior beliefs and thus increase the flexibility of the tax policy. Nonetheless, in practice both solutions seem to be difficult since if firms perceive that their own belief structure is superior then they have no incentive to adopt agency's belief structure, while monitoring and information dissemination would likely be costly and it is unlikely that farmers will be able to easily and accurately process it (Horan et al., 2002). It is worth mentioning that such problems are not met under input-based schemes since farmers do not need information about fate and transport (Cochard, 2003).

- *Dynamic context:*

Under uncertainty the dynamic ambient-based scheme is given by:

$$\phi(\mathcal{E}(\Delta S)) = \phi(S(t) - \mathcal{E}(\mathcal{S}^*(t))) \quad \text{with } \phi' > 0, \phi'' \geq 0$$

where a charge equal to the expected social shadow cost of the pollutant's stock is imposed per unit of deviation from the socially-optimal expected value of the ambient pollution stock $\mathcal{E}(\mathcal{S}^*(t))$ at each instant of time.

By getting more precise, the dynamic ambient taxes applied under the myopic, open-loop and feedback behavioral rule are given respectively by:

$$\breve{\tau}_i(t) = -\breve{\mu}(t)(S(t) - \mathcal{E}(\mathcal{S}^*(t)))$$
$$\breve{\tau}_i^{OL}(t) = -\breve{\mu}(t)\,(r+b)\,(S(t) - \mathcal{E}(S^*(t)))$$
$$+\frac{1}{2}\breve{\mu}_{iS}^{OL}\sigma(\mathcal{E}(\Delta S(t))) - \frac{\partial \pi_i(\mathbf{x}_i)}{\partial S}$$
$$\breve{\tau}_i^{FB}(t) = -\breve{\mu}(t)\left[(r+b) - \sum_{j\neq i}^{n-1}\frac{\partial e_j}{\partial \mathbf{x}_j}\frac{\partial \mathbf{x}_j}{\partial S}\right](\mathcal{E}(\Delta S(t)))$$
$$+\frac{1}{2}\breve{\mu}_{iS}^{FB}\sigma(\mathcal{E}(\Delta S(t))) - \sum_{j\neq i}^{n}\frac{\partial \pi_i(\mathbf{x}_i)}{\partial \mathbf{x}_j}\frac{\partial \mathbf{x}_j}{\partial S} - \frac{\partial \pi_i(\mathbf{x}_i)}{\partial S}$$

It can be easily noticed that the dynamic ambient schemes under OL and FB behavioral rule consist both of two parts. Particular the FB time-flexible ambient tax $\breve{\tau}_i^{FB}(t)$ can be decomposed into: the incentive scheme as defined in the certainty context and into the element $\left(\frac{1}{2}\breve{\mu}_{iS}^{FB}\sigma(\mathcal{E}(\Delta S(t)))\right)$ that reflects the effect of environmental uncertainty (Xepapadeas, 1992). Since $\breve{\mu}_S^{FB}$ represents the farmer i's risk aversion and is negative the second term acts as a stabilizing factor that reduces the tax payment.

Under uncertainty the dynamic ambient scheme is weakened since deviations from the optimal path are not entirely attributed to polluters' emission policies (Xepapadeas, 1992). For instance, farmers who take costly actions to improve their environmental performance can find themselves subject to larger rather than smaller penalties due to natural variations in pollution contributions from natural resources or stochastic variations in weather (Shortle et al., 1998). Conversely, individuals who behave badly may end up being rewarded by the good actions of nature (Shortle et al., 1998). Therefore in order to strengthen the political and ethical acceptability of ambient instruments a "confidence belt" is introduced (Xepapadeas, 1992). Under this modification charges are imposed if and only if deviations outside the belt are observed since this could not be considered result of random fluctuations. In such a way the regulator can avoid imposing penalties or paying subsidies due to random deviations from the desired path, even if all polluters follow the optimal policy (Xepapadeas, 1992).

In a dynamic context and under uncertainty the ambient-based budget-balancing incentive mechanism is given by:

$$T_i(e) = \begin{cases} \phi_i RSB + \phi_{ig}\left[\phi_g RSB + F_g + \Gamma(S(t))\right] & \text{if} \quad S > \mathcal{E}(S^*(t)) \quad \text{with prob } 1-\zeta_i \\ -F_g & \text{if} \quad S > \mathcal{E}(S^*(t)) \quad \text{with prob } \zeta_i \in (0,1) \\ \phi_i RSB & \text{if} \quad S = \mathcal{E}(S^*(t)) \end{cases}$$

where $\Gamma(S(t)) = \breve{\mu}(t)(S(t) - \mathcal{E}(S^*(t)))$ is the expected valuation of the excess concentration of the pollutant. In this context if monitoring shows cheating and the ambient standard $\mathcal{E}(S^*(t))$ is not exceeded then it is considered that there is monitoring error and no fine is imposed (Xepapadeas, 1991).

It is worth mentioning that there is a critic relatively to the instrument's ability to induce compliance with the environmental objectives under different risk attributes. According to Herriges et al. (1994) the BB system of subsidies and random penalties provides the correct incentives for optimal abatement only if polluters are sufficiently risk-averse. Optimal abatement effort will not yield at a Nash equilibrium under risk neutrality since the budget balancing implies that farmer captures the full marginal shirking benefit and suffers only a fraction of the marginal costs. In such a case the fine is not enough to offset the shirking cost savings and the problem cannot be held by increasing the farmer i's compliance incentive without decreasing the compliance incentive of the other farmers.[46] Nevertheless, under sufficiently risk averse farmers, random penalties can provide an extra incentive for compliance and combined with high enough fines to all polluters Xepapadeas' mechanism is applied successfully - as long as liquidity constraints[47] are not binding (Herriges et al., 1994). The random nature of the fine ensures compliance by increasing the variability of shirking cost for each polluter and risk aversion outweighs the dependence of incentives across polluters so the net expected shirking benefits are negative (Herriges et al., 1994).

Summarizing, there is another critic of dynamic ambient-based measures relate to the time lags in the pollution transport. Particularly in a dynamic setup agricultural nitrate pollution is a stock variable, implying that ambient pollution in period t depends both

[46] Increasing the noncompliance fine to firm i will on the one hand encourage it to comply but on the other hand will encourage other firms to cheat and this cheating effect will dominate and optimal abatement effort will not yield at a Nash equilibrium (Herriges et al., 1994).

[47] Liquidity constraints may appear if the fines continue to increase, forming an upper bound on the fine (Herriges et al., 1994).

on farmers' present and past input decisions (Cochard, 2003). Hence $S(\mathbf{x}_t, \mathbf{x}_{t-1}, ..., \mathbf{x}_0, b)$ with $\mathbf{x}_t = (\mathbf{x}_1, \mathbf{x}_2, ..., \mathbf{x}_n)$ the vector of individual productive choices in period t. To take into account these stock effects a dynamic model is required and the resulting dynamic tax must vary across periods in order to implement the first-best. However, such a measure would be too demanding in terms of required information compared to static measures, turning the later more likely to be preferable.

6.4.3 Under asymmetric information

Similarly to the previous analysis the regulator imposes an ambient tax (t_i) on the deviations between the optimal and observed ambient concentration of the pollutant. Under this scheme the farmer i's total tax payment is:

$$T_i(e_i(\Theta_i)) = t_i \int_{\underline{\Theta}_i}^{\bar{\Theta}_i} \left[e_i(\mathbf{x}_i(\Theta_i), \Theta_i) - e_i^*(\mathbf{x}_i(\Theta_i), \Theta_i)\right] h(\Theta_i) d\Theta_i$$

where $\int_{\underline{\Theta}_i}^{\bar{\Theta}_i} [...] \, h(\Theta_i) d\Theta_i$ is the deviation between the optimal ambient concentration and the observed ambient emission level.

It is notable that after following the standard procedure the ambient tax rate is given by (6.7), implying that the optimal ambient tax rate is equal to the Pigouvian tax rate defined previously under complete information (Xepapadeas, 1997). The particular instrument does not take into account the private characteristics of each farmer and can lead to under-implementation of inputs and thus under-emissions as compared to socially optimal regulatory schemes. Only the farmer with the worst characteristics emits at the same level as with the input tax, while the rest emit less.

6.4.4 Mixed Incentive Schemes

Ambient-based schemes have substantial appeal compared to input tax schemes because there is no need to devise firm-specific policies, and would optimally coordinate point and NP control without the need to develop and implement separate point and nonpoint instruments (Shortle et al., 1998). However, even under complete information they appear to have many and notable drawbacks that render their final implementation difficult or even impossible. These disadvantages, as outlined in the economic literature, can be summarized in the following table:

Ambient Measures Drawbacks
1) Collective nature of ambient taxes: a) Weakness of Nash equilibrium concept b) The problem of collusion
2) The non-budget balancing feature
3) Time lags in pollution transport
4) Entry / exit incentives
5) Risk-averse polluters
6) The difficulty in metering ambient pollution
7) The farmers' need to know their environmental types
8) The coordination on a Nash equilibrium
9) Divergences between regulator's and farmers' beliefs about the pollution process

Source: Cochard F., (2003)

Given the difficulties associated with the regulation of NPS pollution problems through pure ambient-based instruments, a combination of policies could be considered an attractive regulatory alternative. Such policies are known as mixed incentive schemes, based on individual and collective efforts (Cochard, 2003).[48] Mixed instruments are drawn on at least two basic instruments and are often superior to pure tax policies (Schmutzler and Goulder, 1997) since the regulator can gather the advantages of the several basic instruments (Cochard, 2003) in order to secure the socially optimal emission level.

An efficient regulatory scheme for NPS pollution consisting of a *mix of ambient and Pigouvian taxes* has been designed by Xepapadeas (1995). Under this scheme the farmers are given the opportunity to pay a Pigouvian tax based on their individual emissions by revealing their individual decisions. In exchange they are charged a reduced ambient tax rate or they may even be excluded by the ambient tax payment, depending on the amount of revealed information. If farmers choose not to reveal individual emissions then they are charged an ambient tax.

[48] Such schemes have been suggested by Xepeadeas (1995), Franckx (2002) and Kritikos (2004), as well as Millock and Zilberman (2004).

Under certainty the total tax payment under Xepapadeas' mixed mechanism is:

$$T_i(e, m_i) = H_i(m_i)(e(\mathbf{x}_i) - e^*) + t_i F_i(m_i)$$

where $H_i(m_i)$ is the ambient tax rate charged per unit of deviation from the optimal ambient pollution level and $F_i(m_i)$ represents the observed part of individual emissions. Both elements depend on the monitoring effort (m_i) undertaken to determine the physical characteristics of the farmer i that permit the quantification of individual emissions.[49] Therefore the after-regulation profits of farmer i is given by:

$$\pi(\mathbf{x}_i, m_i) = \pi(\mathbf{x}_i) - H_i(m_i)(e(\mathbf{x}_i) - e^*) - t_i F_i(m_i)$$

It is logical that without monitoring effort there is no observability of individual emissions $(F_i(0) = 0)$ and each polluter i is liable only for the ambient tax, where the optimal tax rate is equal to the marginal damage $\left(\frac{\partial D(e)}{\partial e}\right)$. On the other hand, there is a monitoring level defined as $\bar{m}_i$ that guarantees perfect observability $(F_i(m_i \geq \bar{m}_i) = e_i)$. In this case farmer i is liable only for a Pigouvian tax, where the optimal tax rate charged per unit of his own emissions is given by (6.1). Finally, in the intermediate situation $(0 < m_i < \bar{m}_i)$ between full and no observability farmers pay both a Pigouvian tax on the observable part of their emissions and a reduced ambient tax.[50]

Even though under certainty neither the regulator nor the farmer have an incentive to increase monitoring effort, implying that it is socially optimal to have unmonitored individual emissions $(m_i = 0)$ and ambient taxes are the only instrument to be implemented to regulate NPS pollution, the choice of no monitoring is not always optimal from the polluter's point of view under uncertainty (Xepapadeas, 1995).

The significance of the proposed mixed incentive scheme is made more precise in the presence of stochastic ambient concentration. Under uncertainty there is a probability that farmers, who have in-

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[49] Alternatively, m_i can represent the information provided by the farmer himself or the amount of installed equipment (Xepapadeas, 1997). For instance, in an agriculture NPS pollution problem farmers can reveal information about their emissions by revealing the rates of inputs used and the application method (Xepapadeas, 1995).

[50] It is logical that the more emissions are observable the less the ambient tax rate is.

ternalized the social costs by adjusting their input and thus emission decisions to the social optimal level, find themselves liable to an ambient tax due to random effects causing observed ambient pollutant to exceed the social optimum level. In such a case any increase in observability of individual emissions ($m_i > 0$) in an NPS pollution problem might be desirable (Xepapadeas, 1999) and in fact it can be regarded as some type of partial insurance for individual polluters against the possibility of paying an ambient tax due to stochastic movements of ambient levels (Xepapadeas, 1995). Therefore risk-averse individual polluters might prefer to have an effluent charge imposed on their observed individuals emissions, after having internalized social costs, in exchange for a reduction in the level of the ambient charge (Millock et al., 2002).[51]

Under an uncertainty context the expected tax payment is defined as:

$$E\left\{T_i(e, m_i)\right\} = H_i(m_i)\phi_i(\tilde{e}(\mathbf{x}_i) - e^*) + t_i F_i(m_i)$$

with $\tilde{e} = \bar{e} + \varepsilon$ the observed ambient concentration in the presence of stochastic factors (ε) and $\tilde{e} = \sum_{i=1}^{n} \bar{e}_i$ the expected total emissions. After some manipulations the farmer i's expected profit function after the implementation of the mixed incentive scheme becomes:[52]

$$E\left\{\pi(\mathbf{x}_i, m_i)\right\} = \pi(\mathbf{x}_i) - H_i(m_i)\left(\phi_i(\bar{e}(\mathbf{x}_i) - e^*) + \frac{\sigma^2}{2}\frac{\partial\phi_i(\cdot)}{\partial\bar{e}}\frac{\partial\bar{e}}{\partial x_{ij}}\right) - t_i F_i(m_i)$$

Based on Xepapadeas (1995) the optimal menu of ambient and emission tax rates is determined as:

$$H_i(m_i) = \frac{\left(\partial\pi\left(\cdot\right)/\partial x_{ij}\right)}{\left(\phi_i'(0)\right) + \frac{\sigma^2}{2}\phi_i'''(0)} \quad \text{and} \quad t_i = \frac{\left(H_i'(m_i)\right)' \frac{\sigma^2}{2}\left(\phi_i''(0)\right)}{\left(F_i'(m_i)\right)}$$

Therefore under uncertainty the combined use of the appropriate chosen Pigouvian and ambient taxes can induce risk-averse polluters to reveal socially optimal information about their own emissions

[51] The larger the variance of expected ambient emission is, the larger the revealed part of individual emissions.

[52] For further details regarding the farmer's and regulator's maximization problem and the characteristics of the optimal tax scheme see Xepapadeas (1995).

without the regulator's need to incur monitoring costs. According to Xepapadeas (1999) this fundamental complementarity between the two instruments can be used to introduce a mixed scheme that could be more implementable than pure ambient taxes.

Even though the proposed mixed scheme mitigates the unpopularity problem of ambient taxes, it does not solve the informational problems since complete information is needed from the regulator's side in order to set the correct emission tax rates (Cochard, 2003).

Another mixed incentive scheme is proposed by Franckx (2002) that relies on a system of ambient level inspections. The regulator uses ambient pollution levels to guide monitoring efforts (Franckx, 2002)[53] and farmers are aware that if ambient pollution exceeds the social optimum level then a specific number of inspections will be carried out (Cochard, 2003). Under this scheme if an inspected farmer i is found not in compliance with the social optimum then a fine (F) is imposed and furthermore the regulator has the legal authority to oblige the farmer to incur the cost of the compliant technology ($\mathbf{w}^a\mathbf{x}_i^a$). On the other hand, the rest non-inspected farmers are not liable for any tax no matter their compliance status (Cochard, 2003). Therefore the total tax payment is given by:

$$T_i(e) = \begin{cases} (F + \mathbf{w}^a\mathbf{x}_i^a) & \text{if inspected and } e_i^* < e_i \\ 0 & \text{if inspected and } e_i^* > e_i \\ 0 & \text{if not inspected} \end{cases}$$

Farmers have to take into account the effect on the monitoring probability (p_i) when they define their productive choices and thus how much to pollute. Thus inspections are a crucial element in any enforcement problem and make sense if they can somehow change farmer's behavior. According to Franckx (2002) the only role the fine plays is that when it is increased, the equilibrium inspection probability can go down. It is notable that the regulator can use infinite fines to induce all producers to comply with certainty, without inspecting any farmer in equilibrium, but this will violate certain budget constraints. Moreover, in a long-run relationship if the regulator always inspects the polluter in the first round of the game, then he develops a *reputation* for inspecting the polluters with a positive probability in the future, and this can induce compliance with

[53] Alternatively the regulator could guide monitoring effort according to his beliefs about which producer is the noncompliant.

a positive probability (Franckx, 2002).

However, random monitoring cannot always be feasible or can be prohibitively costly to establish (Millock and Zilberman, 2004). For instance, in the case of water contamination the pollutant dissipates quickly and it may be very costly to trace the pollution back to its source. Therefore when public random monitoring system is ineffective, the penalty scheme discussed above needs to be modified.

Millock and Zilberman (2004) have developed a mixed mechanism that induces *self-reporting of accidents* in the absence of public monitoring. In our context the excess implementation of agricultural inputs and thus excessive emissions flow could be perceived as an accident. Under this policy scheme each farmer i is made aware that if the regulator detects the damage from an accident then there is a positive probability that he is liable for a fixed penalty. This penalty is imposed on all farmers in the population and it is set equal to the full social cost of an accident that has not been reported, in order to induce the optimal level of abatement. Therefore, under the background threat of a probabilistic collective penalty the regulator offers farmers the chance to self-report the accident and undertake abatement actions in exchange of a reduced individual fine compared to the probabilistic collective penalty.[54] One of the fundamental advantages of self-reporting is that reduces the cost of risk-bearing since farmers pay a certain penalty instead of a random collective fine.

Finally, since individual and collective penalties when used in isolation do not provide polluters with efficient incentives to reduce emissions to the socially optimal level, a two-part penalty system of individual and collective fines is proposed by Kritikos (2004). The combined use of individual and collective penalties aim to induce the enforcement of a regulatory law, targeting a reduction of individual and total emissions under the context of incomplete information.

According to Kritikos' mixed incentive scheme if the ambient pollution level exceeds the targeted level (i.e. social optimum) then a collective penalty,[55] defined as a linear function of the deviation $\Phi_i^K = \min(\pi_i, (e - e^*)\,\Phi^K)$, is imposed on every farmer. At the same time the regulator can spot-check whether individual limits $(\bar{e}_i)$ are respected via unannounced monitoring of arbitrarily chosen farmers

[54] Actually the role of the collective penalty is to increase the incentives to self-report.

[55] To deal with the liability problem arising under the individual penalty, the combined mechanism limits the collective penalty to each farmer's profits $\pi_i(\mathbf{x}_i)$.

and individual penalties given by a linear function $\Phi_i^I = (e_i - \bar{e}_i)\,\Phi^I$ are imposed only on monitored farmers whose actual emissions exceed the individual limit $(\bar{e}_i)$.[56]

Based on Kritikos (2004) the two-part penalty mechanism is given by the following scheme:

$$T_i(e) = \alpha\Phi_i^I + \beta\Phi_i^K \quad \text{with} \quad \begin{array}{l} \alpha = \begin{cases} 1 & \text{if inspected and } \bar{e}_i < e_i \\ 0 & \text{otherwise} \end{cases} \\ \beta = \begin{cases} 0 & \text{when } e \leq e^* \\ 0 & \text{when } e > e^* \text{ and } \bar{e}_i \geq e_i \text{ if inspected} \\ 1 & \text{otherwise} \end{cases} \end{array}$$

If an inspected farmer is found in noncompliance then he is liable for both an individual and collective penalty simultaneously. In such a case the total tax payment is $T_i(e) = \left(\Phi^K + p\Phi^I\right)$. On the other hand if an inspected farmer is found in compliance then the collective penalty is not imposed. Finally, not-inspected producers are liable for the collective penalty and their total tax payment is given by $T_i(e) = (1-p)\Phi^K$.

The attractiveness of the particular policy scheme lies on the fact that it handles the problems inherent with implementation of the individual and collective penalty when used as single instruments. Indeed, under the combined mechanism there are neither multiple Nash equilibria nor liability problems, avoidance behavior does not pay and the mechanism is shaped in a way that no penalties are imposed in an equilibrium situation. However, this instrument requires knowledge of the farmers types otherwise the first-best cannot be achieved (Cochard, 2003).

[56] Where Φ^K and Φ^I are fixed fines imposed in each case that can be set equal to the emission tax t.

7

Voluntary Approaches in Environmental Policy

Since the early 1970s environmental policies focused heavily on command-and-control regulation to ensure adequate protection of environmental quality. However, from nearly the beginning these instruments have been widely criticized for being costly and inflexible (Lyon and Maxwell, 2002), as well as complex and characterized by interdependencies (Pesaro, 2001). Due to their substantially inability to reverse the environmental degradation process (Pesaro, 2001) search has turned towards less costly and more efficient means of achieving environmental protection goals (Dawson and Segerson, 2002). Thus regulators relied on market-based incentives, such as emission taxes or tradable permit systems, that became increasing common by the late 1980s (Lyon and Maxwell, 2002). Nevertheless, a new instrument, namely voluntary environmental approaches, has been recently added to the regulator's tool box (Lyon and Maxwell, 2003) that goes beyond even market-based environmental regulation (Lyon and Maxwell, 2002).

The expression Voluntary Approaches (VAs) usually means a series of "*commitments from polluting agents or industrial sectors to improve their environmental performance*" (Brau et al., 2001). VAs are complements to the current regulatory system and not a substitute (Lyon and Maxwell, 2002), primarily used to alleviate the economical impact of new environmental laws on heavily affected sectors (Šaeur et al., 2001), as well as extend the scope and efficacy of individual air, water, waste and toxic laws (Mazurek, 1998). They are based on a new style of interaction between public and economic actors, where all social forces and all activity fields participate in the prevention and maximum possible reduction of environmental impacts, leading from polluter pays principle to a precautionary and shared responsibility principle (Pesaro, 2001). Voluntary activities appear to be a "*softer*" form of regulation and, in effect, can be broader and more encompassing than mandatory requirements and regulations (IEA, 1997).

Such instruments are expected to lead to socially less expensive solutions to the given environmental problems, since they have the potential to reduce both environmental compliance costs and the associated administrative and transaction costs (Segerson and Miceli, 1998). They can further increase environmental effectiveness and social welfare, as well as contribute to innovation processes and information dissemination (Šaeur et al., 2001). However, despite their growing popularity[1] since the beginning of 1990s program innovation, lack of data, and weak metering and evaluation methods make it difficult to determine the extend to which voluntary programs have actually reduced pollution or abatement and administrative costs (Mazurek, 1998).

Due to the increasing importance of Voluntary Approaches as an environmental policy instrument and the particular importance they acquire in the context of the EU policies, the latest available developments in the area are presented in some detail for the better understanding of the properties and the various aspects of the instrument.

7.1 Voluntary Approaches: some Introductory Issues

This subsection describes all the major features of voluntary approaches, as presented in the recent environmental economics literature, with the purpose of enhancing our understanding regarding this environmental policy instrument. In particular, a description of the various differentiation criteria of VAs, the factors that motivate their establishment, as well as the economic agents who appear to most likely initiate or participate in VAs is provided. Moreover, the benefits and drawbacks, as well as the existing implementation difficulties are described. Finally, the effectiveness of VAs is assessed and some rules about their effective use are presented.[2]

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[1] Even though the use of VAs become more common since the beginning of 1990s, there are some precedents of voluntary approaches in the OECD member countries as far back as the 1960s and early 1970s. For instance, the first environmental agreements between industries and authorities in Japan and France were reported in 1964 and 1971 respectively (Šaeur et al., 2001).

[2] The review of voluntary approaches is not restricted to environmental contracts designed for the agricultural sector. Therefore the term "farmers" will be many times replaced by the term "agents", "polluters" or "producers".

7.1.1 Typology of Voluntary Approaches

Voluntary approaches can be differentiated according to various perspectives, but the prevailing taxonomy is based mainly on the degree of public intervention, meaning the degree of public authority's impact on a certain hierarchical level of public administration (Šauer et al., 2001). Based on this criterion VAs typically fall into one of Lévêque's three basic categories: *unilateral commitments*, *public voluntary schemes* and *negotiated agreements* - the distinctive characteristics of which are given below:

1. *Unilateral commitments.*

Unilateral commitments or agreements are environmental improvement programs undertaken by a single producer or a group of producers and further communicated to their stakeholders (Brau et al., 2001). Such actions are often met in economic literature as: business-led corporate environmental programs, self-regulation and corporate environmentalism (Alberini and Segerson, 2002). The main characteristic of such schemes is that the initiative rest solely with the polluters themselves. Even though unilateral programs may be developed after consultation with government bodies (Lyon and Maxwell, 2002), the regulator do not play any active role in their design (Alberini and Segerson, 2002). Industries, producers and trade associations prepare their complete environmental improvement programs, define the environmental goals and state measures leading to their achievement (Šaeur et al., 2001). Moreover, they may authorize another third party to monitor and conflict-resolution, in order to increase credibility and the environmental effectiveness of their commitment. The ultimate aim of such agreements is not only to encourage polluters to voluntary adopt better environmental management codes but also to improve public perception and / or regulatory goodwill so that to reduce costs associated with permitting and reporting indirectly, as well as minimize the threat of more stringent regulation (Mazurek, 1998).

It is notable that unilateral agreements usually belong to an industry trade association (Lyon and Maxwell, 2003), where participation is a condition of trade association membership and the ultimate sanction for a participant that fails to implement the established voluntary practice codes or make adequate progress towards program goals is the threat of dismissal from the trade organization (Mazurek, 1998).[3]

Notable examples of unilateral agreements are Arco's voluntary introduction of reformulated gasoline, the German Industry and Trade Association's plan to reduce carbon dioxin emissions (Lyon and Maxwell, 2002) and the Chech Association of the Petrol Industry and Commerce initiative concerning fuel quality characteristics (Šaeur et al., 2001). Similar commitments are the 3M's 3P programme, Dow Corporation's WRAP and CMA's Responsible Care programme (Dawson and Segerson, 2002). Responsible Care (1988) is the most prominent unilateral program in US to date (Mazurek, 1998). The scheme was prepared in response to the decreasing level of public trust on chemical industry and to the looming danger of stricter regulation (Šauer et al., 2001). It aims at regaining society's trust and limit regulatory intervention to a level that is acceptable to the industry (Lyon and Maxwell, 2002). Thereupon, the initiator Chemical Manufacturers Association (CMA) provided its members with general guidance documents that explain how to adopt six management practice codes that range from pollution prevention to product stewardship and allowed participants to use a registered Responsible Care trademark in order to obtain public recognition (Mazurek, 1998).

2. *Public voluntary schemes.*

Public voluntary schemes or agreements are environmental programs explicitly developed by some public body (i.e. US EPA) or by a quasi-public but non-governmental body (i.e. International Organization for Standardization / ISO) and to which polluting agents and sector associations can only agree with (Brau et al., 2001). The initiator of such schemes is the regulator who unilaterally determines both the rewards and obligations from participation, as well as the eligibility criteria (Alberini and Segerson, 2002). Participating agents just agree to certain non-mandatory rules that affect their activities, technology or management. In particular, these rules include the characteristics of the given program such as the requirements for individual participation, measures to be undertaken, ways of moni-

[3] According to Mazurek (1998) in practice noncompliant participants are mostly provided information and technical assistance, and in some case sanctions means letters of inquiry at a first stage and terminate with dismissal.

toring commitment[4] and means of evaluating the results (Šaeur et al., 2001). Supplementary funds for science and research, technical aid, as well as rights to use an ecological logo or certification symbol are also provided by the scheme in order to secure a broader and more efficient implementation of voluntary environmental actions.

Examples of public voluntary programs are the environmental management systems certification standards, EMAS and ISO 14000 respectively (Šaeur et al., 2001). In this category fall the Conservation Reserve Program and its successor the Environmental Quality Incentives Program (Dawson and Segerson, 2002), as well as the US Green Lights and Energy Star programs aiming to reduce energy-related CO_2 emissions or minimize specific process emissions (IEA, 1997). It is noticeable that from the 42 US national voluntary initiatives the 31 are purely public voluntary programs (Mazurek, 1998). 33 / 50 Program (1991) is the US major public voluntary program, designed to induce manufacturers to progressively cut the emissions of 17 key toxic chemicals by providing some favorable publicity and some limited technical assistance, as well as by signaling the increased threat of federal regulation (Lyon and Maxwell, 2002). Finally, among the US public voluntary schemes the "AgStar" and "Ruminant Livestock Efficiency" programs are designed exclusively for the agricultural sector, aiming to encourage farmers to adopt best management practices to reduce agricultural methane emissions (Mazurek, 1998).

3. *Negotiated agreements.*

While under the previous categories of VAs the prime mover behind a new program was either the polluter or the regulator, respectively, under negotiated agreements these two actors are both active participants (Lyon and Maxwell, 2002). According to Pesaro (2001) negotiation is not a bargaining form to solve conflicts but a peculiar interaction model of a new way of policy-making, where economic actors are no longer only "part of the problem" but also "part of the solution". In literature negotiated agreements can be also termed: environmental covenants, voluntary environmental agreements, vol-

[4] Progress is monitored primarily via annual self-reports that in some cases are verified by a third, independent party. However, reporting requirements vary from facility to facility and may be annual, biannual or quarterly (Mazurek, 1998).

untary partnership, bilateral or cooperative agreements and formal voluntary approaches.

Negotiated agreements refer to contractual arrangements between a regulatory authority and an individual polluter, or a sector, or association. The terms of the agreement are jointly set and they are the result of negotiations between the involved parties. Such formal approaches include a specific goal, clearly defined tasks, a time schedule and other conditions necessary for the fulfillment of the expected results (Šaeur et al., 2001), as well as reciprocal commitments and shared responsibility on the part of all participating parties (IEA, 1997). In particular, polluters are obliged to improve their environmental performance in the time and ways outlined by a number of constraints and clear rules, while the regulator's obligation may involve a commitment either not to enforce a particular action against the polluter or to exempt the polluter from certain regulations (Alberini and Segerson, 2002). Moreover, the regulator's commitment could also be positive since it may involve considerable up-front specific financing and technical assistance, a law for using ecological logos, or grant of a particular permit or approval for other activities (Alberini and Segerson, 2002) in order to ease some of the additional administrative and organizational burden and to encourage participation (IEA, 1997).

A primary goal of negotiated strategies is to improve the effectiveness and efficiency of laws by reducing regulatory burden and providing relief to regulated industry (Mazurek, 1998). Their use can be mainly justified in cases where the environmental target can be achieved through technology innovation, especially when market imperfections exist or when environmental innovation has positive spillovers (Xepapadeas, 1997). Such bilateral agreements tend to be heterogeneous in nature (Lyon and Maxwell, 2003) and may also take on the status of legal binding contracts if legislation gives the authority to executive branches of government to sign them (Lyon and Maxwell, 2002) or the existing law allows the regulator to sign such agreements (Šaeur et al., 2001). However, legal obligations may be also included and operate only if the agreed goal is not fulfilled or the bargaining process does not always end up with the signing of an agreement.

It is worth mentioning that such voluntary approaches are also called *private environmental agreements* if concluded between polluters and those harmed or their representatives (Šaeur et al., 2001). One theoretical justification is the Coase 's idea of direct negotiation between polluters and victims, that however requires favorable conditions such as clearly defined ownership rights toward environmental goals, access to information, zero transaction costs etc.

Characteristic examples of such negotiated agreements are the French agreement on the treatment of End-of-Life Vehicles to reduce car waste destined for a land fill site, the New Zealand cement industry's agreement with the government as part of the government's plan to reduce carbon dioxide emissions, and the German energy sector's agreement with government to reduce CO_2 emission through a 20% reduction in energy consumption in order to deter the passage of a waste heat ordinance and the implementation of a carbon/energy tax (Dawson and Segerson, 2002). In the same context lies the Swedish agreement of producer responsibility for packaging and the Dutch policy on implementing target emissions levels in the chemical industry (Lyon and Maxwell, 2002). Finally, even though Project XL and Common Sense Initiative (CSI) involve negotiation aiming to reduce administrative costs associated with reporting, monitoring and permitting, they also resemble public voluntary programs (Mazurek, 1998).[5]

7.1.2 Criteria for Differentiating Voluntary Approaches

The discussion above suggested that Voluntary Approaches can be identified as the "*commitments undertaken by producers and sector associations, which are the result of negotiations with public authorities or are explicitly recognized by the authorities or producers themselves*" (Mazurek, 1998). Based on Šauer et al., (2001) further criteria can be identified that can serve to differentiate voluntary approaches within the scope of the three basic categories of the previous section. These differentiation criteria are:

a) *The degree of detail of the agreement.*[6]

[5] Project XL and CSI were designed in response to complaints from the regulated community (firms) regarding the growing details and complexity of federal pollution control laws, to reform environmental regulation (Mazurek, 1998).

[6] This criterion can be also referred to as "degree of structure" (IEA, 1997).

VAs can be rather generic or very detailed in their definition of the actions, objectives and content, the number of sectors involved etc. (IEA, 1997). Usually, the regulator can distinguish between *target-based* and *implementation agreements*, according to their relationship to the targets of environmental policy. In particular, target-based agreements are based on exactly specified and quantified goals (i.e. German energy sector's negotiated agreement), while implementation agreements aim to determine the means that develop a consensus with previously established environmental policy goals. Moreover, voluntary approaches can either have established less precise environmental targets (i.e. Project XL and CSI), be directed toward technological progress or merely obligate participating polluters to provide information regarding their effects on the environment (Šaeur et al., 2001).

b) *Level of legal obligation toward fulfilling the agreement.*

Whether VAs include legal obligations or not depends on existing laws that may or not allow the regulator to sign such agreements. It is notable that in Netherlands the majority of undertaken negotiated contracts have civil-law characteristics, implying that if a participating agent fails to fulfill the agreement's goals then he is responsible before a civil court (Šaeur et al., 2001). On the other hand, in US only the Project XL contains legally binding features, since it "promises" relief from existing laws and regulations in exchange of environmental performance superior to status quo standards (Mazurek, 1998). However, in several countries voluntary or negotiated agreements have been established without comprising any legal obligation. Characteristic examples are the negotiated agreements employed in Norway to deal with non verifiable packaging emissions, which are wastes not covered by a formal tax base and thus taxes or direct regulations can not be enforced by a third party such as a legal court (Nyborg, 2000). Such agreements cannot be sustained by legal enforcement but only by mutual compliance by involved parties (Nyborg, 2000).

c) *Sanction types in case of failure to fulfill the agreement.*

The sanction types that operate if signatory agents fail to fulfill the agreement's provisions can be financial, moral, different means of regulation or even cost associated with the resignation from a bilateral agreements (Šaeur et al., 2001). It is notable that the most EPA voluntary initiatives are non-binding and impose no sanctions - compliance actions and fines - for program withdrawal (Mazurek, 1998).

d) *Agreement initiator.*

Based on the previous analysis the agreement initiator can either be a public body, an individual polluter or a group of polluters, a sector representative, or even a non-governmental organization (Šaeur et al., 2001). A voluntary approach can be designed unilaterally by one of the actors listed above or be the result of their cooperation. Nevertheless, the final type of VA employed and thereupon the type of the initiator depends on the nature of the environmental problem and the degree to which legislative laws and regulatory policies address the problem (Mazurek, 1998).[7]

e) *Characteristics of the subjects damaging the environment and participating on the agreement.*

Obviously, voluntary approaches target various types of agents, whose economic activity pollutes the ambient environment, starting from extraction activities (i.e. mining, forestry) up to manufacturing (i.e. chemicals, electronics and computers), or agriculture. "Liable" to a VA can be individual activities, industries or sectors, as well as a "group of subjects" such as the US Energy Star programs that include agreements with construction, electronics, office equipment and energy firms (Mazurek, 1998). It is underlined that unilateral programs target individual industries (Mazurek, 1998).

f) *The level of openness toward third parties.*

Negotiated agreements involve the widest array of participants up front in order to minimize potential legal challenge later on. For instance Project XL requires industry participants to recruit residents living near participating facilities or have a direct interest in the outcome, to participate in a 6-month negotiation process (Mazurek, 1998). On the other hand, unilateral agreements, such as

[7] The most common case of voluntary approaches in Europe are negotiated agreements (Šauer et al, 2001).

Responsible Care, excluded environmental and labor organizations from program development in order to preserve Project's autonomy (Mazurek, 1998).

g) *The level at which the agreement is concluded.*

Voluntary approaches can be further separated relatively to their application level into agent-specific, industry-wide, national, federal or regional approaches. In particular, industry-wide approaches involve a collective environmental target defined for the whole industry, in contrast to agent-specific approaches that specify individual actions and solutions for specific needs. Furthermore, every geographic region can develop its own regional VA "model" that is strongly characterized by the territorial peculiarities. Such negotiated agreements are more likely than industry-wide applications, since the design and implementation capability appears to be greater when actors and problems occur on a restricted area. It is notable that Japan has 30.000 negotiated agreements regulating industrial activities on local level (Šaeur et al., 2001). Voluntary programs may also target states and localities, such as the US State and Local Outreach Program (Mazurek, 1998), as well as set national target. In US, 42 national voluntary initiatives have been developed since 1988 (Mazurek, 1998). It is possible that a single VA can comprise all these features. For example the Italian "Part for Energy and Environment" agreement, developed under the Kioto Protocol commitments, was signed in national, regional and local level, as well as by individual economic actors on particular goals (Pesaro, 2001).

h) *The no-surprise feature of the agreement.*

Under "non-surprise VAs" the regulator offers assurances to participants that he will not change the terms of the agreement (i.e. target), in response to changing environmental protection needs. Such agreements guarantee the polluters that no additional costs or restrictions will imposed to them in the future, but may not allow the use of new information leading to inadequate environmental protection and inefficiencies (Langpap and Wu, 2004). On the other hand an "agreement with surprises" handles such problems but may discourage participation and conservation effort. Examples of such non-surprise policies are the Habitat Conservation Plans, the Safe

Harbor Agreements and Candidate Conservation Agreements with Assurances (Langpap and Wu, 2004). Furthermore, in US the Department of Interior has developed a "No Surprise" policy under which it signs agreements with companies or individual landowners committing not to change the rules applying to a particular piece of property for a fixed period of time (Lyon and Maxwell, 2002).

i) *The nature of enforcement instruments.*

The regulator can alternatively employ “carrot” or "stick" instruments to induce participation and achievement of the established environmental goals (Segerson and Miceli, 1998). *Carrot-based* VAs create a surplus over aggegate costs of environmental protection via the provision of positive incentives such as: financial inducements in the nature of total-cost or sharing subsidies (i.e. US Conservation Reserve Program), information subsidies (i.e. US Green Lights), technical assistance and / or public recognition through awards, press announcements or a law for the use of product logos that bears the program’s name (i.e. US Energy Star) (Mazurek, 1998). It is notable that US agricultural water quality policy has mostly relied on carrot-based programs (Wu and Segerson, 2003).

However, the provided incentives may not be always positive. Agents can be obligated to alter their production or abatement practices via *stick-based approaches* that rely on the explicit or implicit threat of a harsher outcome if a VA is not reached, through the implementation of existing mandatory restrictions (Segerson and Miceli, 1998) or the establishment of a new regulation. Such instruments are merely utilized for the control of industrial NPS problems (Wu and Segerson, 2003), with notable examples the Superfund Act and Clean Air Act Amendments aiming to induce firms to internalize the costs of their current toxic pollution (Khanna and Damon, 2002). The difference between an agreement and a direct regulation lays on the fact that regulation is enforced without the agents consent, while the agreement requires mutual acceptance of the terms (Nyborg, 2000). However, under stick-based approaches this difference can be fairly trivial, since the regulator actually threatens harsh policy measures to induce acceptance of the VA (Nyborg, 2000). Hence, it is evident that the use of the term “voluntary” is not that successful since this term embodies the free will of agents to commit themselves to an environmental initiative.

Furthermore, the regulator can implement a *mixed-based approach* that uses the carrot approach in combination with the stick to induce polluting agents to reduce their emissions to the desirable level (Segerson and Miceli, 1998). Nevertheless, there is the category of VAs that involves neither carrot, stick or a combination of these measures and is known as *pure voluntary agreements*. Actually they are the private environmental agreements undertaken directly between polluters and those harmed (Šaeur et al, 2001). However, when VAs are used as an independent policy instrument environmental effectiveness is not achieved relatively easy, justifying the perception that an efficient VA should combine both voluntary and mandatory tools.

7.1.3 Motivation behind Voluntary Approaches

Voluntary environmental initiatives have been attributed to a variety of motives. Lyon and Maxwell (2002), as well as Alberini and Segerson (2002) attempt to provide some insights regarding the great appeal of VAs and identify the basic motives behind voluntary actions. These basic incentives include: marketing in relation to "green" consumers and investors, cost-cutting, personal satisfaction and most importantly influence on the regulatory strategy. In detail each of these motivations entail the following:

1. *Personal stewardship.*

Personal satisfaction or utility gained from undertaking environmental friendly activities voluntarily, can be considered as a factor that stimulates voluntary actions (Alberini and Segerson, 2002). Even though, such a motive is mostly likely when pollution stems from individual rather than collective behavior, personal stewardship is important for the effective "massive" reduction of pollution activities.

2. *Response to government-created incentives.*

Voluntary environmental actions can be induced via positive or negative regulatory incentives, described in the previous section. In particular, the regulator must guarantee that under a VA based on "carrot instruments" the polluters' profit level or net benefit is at least as high as under the pre-policy level. On the other hand, under a VA based on "negative inducements" the default/ non participation policy must be made the more costly policy, so that polluters are induced either to participate in public voluntary programs or negotiate voluntary reductions in pollution (Alberini and Segerson, 2002).

3. *Response to market-based incentives.*

Voluntary initiatives can be also motivated by the actions of the so called "green" consumers and investors. Getting more specific:

a) *Consumers with "green" preferences.*

It is known that high income consumers - at least in developed nations - are willing to pay a premium for environmentally friendly products (Lyon and Maxwell, 2002) and use marketplace to induce producers to adopt environmental friendly behavior (Alberini and Segerson, 2002). The actions of such "green" consumers can stimulate shifts in demand and supply of environmentally friendly products (i.e. organic products, reformulated gasoline and biodegradable plastic bags), turning environmental activities more profitable and stimulating potential changes in corporate behavior (Lyon and Maxwell, 2002). Thus producers that want to improve their public image with respect to environmental issues and increase "green" consumers' goodwill (Khanna and Damon, 1999), are willing to go above and beyond the levels of care required by the environmental regulations (Lyon and Maxwell, 2002). In particular, they proceed in environmental friendly changes either in their production processes or in product characteristics, which allow product differentiation and fill a green market niche (Alberini and Segerson, 2002), leaving potential room for an increase in sales (Khanna and Damon, 1999) and profits via a higher price as long as there is sufficient demand.

b) *"Green" investors.*

Even though producers of intermediate goods does not seem to be affected by green reputation in output market, a voluntary environmental action can be utilized to improve access or terms received in input markets (Alberini and Segerson, 2002). According to Lyon and Maxwell (2002) there are "socially responsible" mutual funds which avoid investing in producers deemed environmentally irresponsible (i.e. tobacco, nuclear power). "Green" investors participate in such funds and reduce the supply of capital to the excluded producers, raising their capital costs. Moreover, stock prices respond negatively to unfavorable news about corporate pollution (Lyon and Maxwell, 2002), since stockholders may equate poor environmental performance with large penalties and/or liability for the cost of cleanup at contaminated sites (Alberini and Segerson, 2002). A notable example is the 33/50 program where the publication of TRI figures damaged the stock values of the heaviest polluters, forcing them to substantially reduce their releases (Alberini and Segerson, 2002). Therefore, a pollution reduction agreement can guarantee long-term benefits by increasing "green" investors' confidence (Khanna and Damon, 1999) and resulting in reduced capital costs (Alberini and Segerson, 2002).

4. *Improving productivity.*

Incentives to generate environmental improvements voluntarily are also present in the absence of regulatory and output and/or input market incentives. It is logical that producers proceed in voluntary pollution reduction if such an action directly cuts costs and increases profitability. Such a "painless" pollution prevention can be achieved by improving the efficiency of manufacturing processes (Lyon and Maxwell, 2002), indicating either a reduced or improved use of a given input that is directly related to pollution generation (Alberini and Segerson, 2002). Notable example of voluntary environmental improvements in the absence of incentives is the 3M Corporation's "Pollution Prevention Pays" Program where line workers identified opportunities for waste reduction leading to cost savings (Lyon and Maxwell, 2002).

5. *Optimizing corporate regulatory strategy.*

Probably, the most important motivation behind voluntary environmental actions is the economic agents' pursuance to influence and hence shape regulatory decisions. In particular, producers may take strategic actions trying either to preempt or mitigate the effects of future regulation, reduce the extend of monitoring or alternatively raise rivals costs (Lyon and Maxwell, 2002). In particular:

a) *Preempting tougher regulation.*

Voluntary environmental improvements, such as "Responsible Care" program and "Big Three" automakers' Vehicle Recycling Partnership, can be perceived as an attempt to avoid the transaction costs and / or compliance costs associated with the traditional mandatory legislative / regulatory process (Lyon and Maxwell, 2002). In particular, producers through self-regulation attempt to take proactive steps with just enough stringency to conciliate society (i.e. environmentalists) in order to deter the demand for regulation. Moreover, since many public voluntary agreements rely on the background threat of legislation producers tend to accept to offer a greater level of pollution reduction so that the threat is removed. Finally, negotiated VAs involve direct negotiation between industry and the regulatory body that actually bypasses the legislative process (Lyon and Maxwell, 2002).

b) *Weakening Forthcoming Regulations.*

However, if it is impossible to preempt regulation (i.e. Clean Air Act Amendments) then voluntary actions of producers can be employed to influence the regulations subsequently set by the government (Lyon and Maxwell, 2002). In such a case, high quality producers that anticipate environmental regulations can grab the opportunity for comparative advantage over market competitors and hence profit if they choose their environmental friendliness level before the regulator sets standards.

c) *Reducing regulatory monitoring.*

Producers can use VAs to gain public recognition for responsible environmental management, which may indirectly reduce monitoring rate or lax the scrutiny from regulator. In particular, if the regulator can observe the producers' actions then the later has an incentive to proceed in voluntary pollution reduction via irreversible investments in order to convince the former that it is less likely to violate the standards in the future. In return the regulator may pursue a laxer monitoring policy and focus his monitoring and enforcement efforts on other producers (Lyon and Maxwell, 2002).

d) *Encouraging Anticompetitive Regulation.*

Finally, voluntary pollution abatement can be employed by large producers to stifle competition and raise their industry-wide rents.

In an uncertainty context the regulator cannot know whether the costs of the new regulation are too high to lead small producers off the market, offering large firms an incentive to try to convince him via their voluntary actions that the industry-wide compliance costs are low so that a strong regulation should be imposed to provide substantial benefits at low costs (Lyon and Maxwell, 2002).

7.1.4 Characteristics of agents undertaking voluntary initiatives

The decision whether to initiate or participate in a voluntary program or not is affected by various factors such as: agent-specific characteristics and external pressures. The basic factors that appear to have affected economic agents' willingness and ability to initiate and participate into voluntary initiatives are listed below:

1. *Producer-specific characteristics: size,*[8] *R&D expenditures, financial and environmental performance, costumer interfacing.*

Voluntary initiatives appear to be skewed towards large producers (Mazurek, 1998). This can be attributed to the higher exposure to liability due to higher public profiles, the presence of economies of scale due to lower marginal abatement costs, the better access to capital markets and / or the higher ability to influence the regulator through their overcompliance. In particular, large firms may feel more pressure to act from environmental groups, politicians, regulators and concerned citizens due to higher public profiles, while compliance with regulations can be relatively cheaper for large firms since the fixed costs associated with environmental compliance are large enough to generate economies of scale (Lyon and Maxwell, 2002).

However, even though participation is expected to be higher in R&D intensive industries there are no strong evidence to support this thesis and the same holds for the impact of profitability and recent growth of the company on the initiation and participation incentive (Alberini and Segerson, 2002).

Poor environmental performance is positively related to the willingness to participate (Alberini and Segerson, 2002), since such an

[8] As a proxy of the agent size variables such as sales' figures; the number of employees; the value of assets can be used(Lyon and Maxwell, 2002).

environmental record may either imply lower costs of "performing well" today encouraging producers to undertake new voluntary actions or it may attract the attention of media and pressure groups pushing producers towards VAs (Lyon and Maxwell, 2002). It is notable that producers with good environmental record may not be willing to incorporate voluntary actions due to the fear of bad publicity if they fail to maintain their outstanding performance (Lyon and Maxwell, 2002). Indeed, revelation of information regarding poor environmental performance affects negatively the market value of the producer, inducing indirectly agents to corporate in voluntary actions. In particular, when revealed information indicates higher than expected levels of emissions investors view this as a negative economic signal linked either with inefficient production, more intensive regulatory monitoring or with a higher probability of future environmental litigation (Lyon and Maxwell, 2002). Hence agents are willing to reduce their returns currently by undertaking abatement investment if they expect to be rewarded for superior environmental performance in subsequent years via increased market value.

Furthermore, producers with higher advertising to sales ratios and producers of final-good products are more likely to voluntarily cut emissions (Lyon and Maxwell, 2002), since the product characteristics or production practices are more visible or recognized to consumers (Alberini and Segerson, 2002).

2. *External Pressures: community, environmental and industry group, regulator.*

Moreover, the magnitude of expected future regulatory financial incentives, the allocation of the bargaining power and the nature of bargaining process appear to play significant role in the decision to initiate or participate in a VA (Alberini and Segerson, 2002). In particular, the likelihood and extent of corporate voluntary actions are increasing in the perceived level of future government regulation (Lyon and Maxwell, 2002). Furthermore, the implementation of voluntary actions is affected by the strength of community, environmental, and industry group pressures (Lyon and Maxwell, 2002), even in the absence of regulatory programs and enforcement (Alberini and Segerson, 2002). On the one hand, green consumers can raise the benefits from friendly corporate environmental actions through increased sales, while the environmental organizations and citizens not belonging in consumer groups can raise the costs through the pressure on regulator for future regulations (Lyon and Maxwell, 2002). Finally, industry groups may be another source of motivation in order to undertake corporate voluntary actions, since association members may pressure each other to coordinate actions needed to forestall regulation threat.

3. *Industry characteristics: degree of competition.*

Arguments have been put forward that the adoption of voluntary approaches is affected by the extend of competition (Alberini and Segerson, 2002). However there is no evidence to support the perception that corporate environmentalism is more likely under less concentrated industries.

7.1.5 Assessment of Voluntary Approaches: Benefits and Drawbacks

The remarkable turn towards the instrument of voluntary approaches has been strongly connected with the associated advantages over mandatory tools. However, the evaluation of the VAs effectiveness has revealed some notable disadvantages justifying the observed implementation difficulties. Therefore, its important to define both the benefits and drawbacks of voluntary approaches in order to complete the instrument's "profile".

Benefits of voluntary actions include:[9]

1. *Adaptability, flexibility and cost effectiveness.*

The main characteristic of VAs is that instead of dictating the use of a particular means individual agents are allowed to choose the means by which the determined environmental target is to be fulfilled (i.e. abatement strategies). Thus producers are left free to find cost effective solutions to reach the target, which are tailored to circumstances and their specific production characteristics, implying a potential for greater flexibility and administration / transaction cost savings compared to other traditional tools. Moreover, since VAs lead to higher "solution variability" agents are "permitted" to correspond rapidly and adjust their strategies to the timely changes

[9] The most "categories" of benefits and drawbacks are from (IEA, 1997). For details see IEA (1997), Table 2.

of technical and economic parameters (Šauer et al., 2001), providing great stability in long-term requirements (IEA, 1997). Hence, VAs allow the regulator to appear simultaneously environmental-friendly and business-friendly (Nyborg, 2000).

However, it is pointed out that the perception that negotiated agreements are more cost effective than administrative approaches (Šauer et al., 2001) is strongly dependent on the assumption that mandatory approaches are inflexible, which in practice may not always hold (Alberini and Segerson, 2002).

2. *Promotes understanding and trust in the sector*, as well as *continuous dialogue with the regulator.*

VAs can lead to a collective understanding of environmental problems and a mutual responsibility from polluters' side (Šauer et al., 2001), since they require positive actions and not passive reactions to instruments implemented by regulators (Pesaro, 2001). They can promote and increase cooperation and trust among industry, non-government organizations and the public (Mazurek, 1998). In particular, VAs use cooperative strategies to improve outcomes (Mazurek, 1998), promoting a high degree of functional representativeness and interaction among actors involved in the policy processes, reducing confrontation and shifting from a centralized and authoritative environmental policy into a participatory and decentralized policy (Pesaro, 2001). Moreover, by including a third, independent party in the goal establishment step (i.e. reporting and monitoring) the trustworthiness towards the sector and / or individual agents can be improved.

3. *Encourages innovation, information exchange on best practices and potentially more efficient and quicker implementation.*

VAs provide a forum for information sharing among various parties (Lyon and Maxwell, 2002), leading to information dissemination regarding the alternative techniques of pollution reduction. Moreover, their flexibility may encourage creativity that may also lead to technical and organizational innovations, reducing environmental compliance costs, administrative and transactions costs associated with the preparation, conclusion and inspection of the concluded agreement (Šauer et al., 2001). Additionally, under the best conditions VAs may imply the faster achievement of established goals in a way that other approaches cannot (Pesaro, 2001).

4. *Devolves responsibility to local level, and integration of envi-*

ronmental improvements into business planning cycle.

Under VAs producers and sectors are encouraged to proceed in proactive approaches, as well as to alter the input usage that may increase productivity and savings on materials, energy, wages (Khanna and Damon, 2002).

5. *Provides "green image" to participating economic entities* and *creates "soft effects".*

Finally, voluntary actions are viewed as a way to promote favorable public opinion and influence consumers' choice regarding a range of products, leading to the substitution of older, less environmental friendly products with new products that process desirable characteristics in environmental terms.

Despite these advantages of VAs, there are some drawbacks that reduce their effectiveness. These drawbacks include:

1. *Disturb competition.*

VAs can be used to cause disturbances in the conditions of economic competition and thus restrain trade, offering private benefits to participants but not to society (Mazurek, 1998). In particular, VAs may be employed to make it impossible for some producers to enter the market or prohibit a third party from entering the system or even drive gradually a product out of the sector (Šauer et al., 2001). It is notable that in 1977 the European Committee evaluated 20 instances of that kind of disturbances (Šauer et al., 2001) and that unilateral agreements are considered to have the greatest potential to restrain trade within the industry by changing relative costs or by establishing entry barriers (Mazurek, 1998).

2. *Room for the activities of the "free riders".*

The primary disadvantage of VAs arises when the agreements in usage are not associated with any individual sanctions but depend upon a collective responsibility if the aggregate target is not met (Šauer et al., 2001). In particular, when there is interaction among agents the nature of VAs benefits is collective and characterized by no rivalry and no excludability (Brau et al., 2001), then free riding is likely through non-compliance and short-term thinking to take advantage of not participation (IEA, 1997). Under such industry-wide VAs some individual polluters can relay on others to meet the target, without incurring the costs of pollution reduction and thus getting the benefits of the others effort - if they believe that the participation of the others will be sufficient to forestall the imposition of the costly policy (Alberini and Segerson, 2002). Thus each producer gains a greater benefit by not reducing emissions because of associated cost savings, by either deciding not to fulfill the established goal from the beginning (ex-ante) or at the very end (ex-post) if they do not anticipate a long-term cooperation with the VA partners (Šauer et al., 2001). The free riding behavior can lead to a failure of an agreement before the potential benefits are realized.[10]

3. *Uncertainty about legality and anti-trust legislation.*

The existing legislative framework and uncertainties about the instrument's legality limit the regulator's ability to use voluntary efforts due to the required attention and resources on meeting legal requirements, as well as judicially imposed deadlines (Mazurek, 1998). In particular, concerns about the legality of Project XL led to lower participation rates than expected leading to lower environmental benefits. Moreover, anti-trust law defines the limits within which self-regulatory codes can be developed and enforced, constraining attempts for self-regulation (Mazurek, 1998). In particular, it may constraint the type of desicion-making tools (i.e. product or process standards) or enforcement mechanisms if the voluntary initiative is perceived as an attempt to fix prices and thus restrain trade. Finally, the applicability and effectiveness of VAs is further restrained by the actors' incapability and unwillingness to understand the conditions and constraints of such an instrument (i.e. Part for Energy and Environment), as well as to create new and wider policy networks and ways of interaction (Pesaro, 2001).

4. *Insufficiencies in monitoring, inadequate clarity and accountability.*

The lack of clearly defined decision making, participatory, monitoring and reporting procedures has made voluntary approaches vulnerable to the charge that they lack transparency (Mazurek, 1998) and has decreased their trustworthiness in the eye of the public, making difficult the fulfillment of an agreement. However, many VAs do not include monitoring and reporting requirements, damaging the credibility of the VAs and making ex post evaluation of effectiveness difficult (Lyon and Maxwell, 2002). Voluntary actions such as Project XL and Responsible Care have failed to develop independent, third

[10] However, according to Dawson and Segerson (2002) even if all polluters are identical, it is possible to have an equilibrium in which a subset of polluters in the industry participates and the remaining free-ride

party verification methods to monitor companies (Mazurek, 1998).

5. *No guarantee of parties obeying agreement, could be open to abuse.*

6. *No incentive to go further than the agreed objectives, may appear not demanding enough.*

7. *Technological innovation may not be encouraged unless stated or included in the agreement.*

8. *Number of participants could be restricted due to transaction costs.*

When private polluters are very heterogeneous, fragmented and loosely organized then VAs may not be well-targeted, limiting the instruments environmental effectiveness. In practice the regulator lacks information about the likely characteristics of polluters (i.e. amount of pollution and the cost of pollution abatement). This lack of information restricts the instrument's ability to deal with adverse selection problems (Alberini and Segerson, 2002). However, it should be noted that it is inappropriate to evaluate a VA only on the participation base, since high participation rates does not guarantee that the VA will achieve the desired environmental target, since agents proceed in cosmetic abatement and thus the aggregate abatement may fall short of the goal (Alberini and Segerson, 2002). Thus when a VA is designed both the terms of participation and the abatement obligations must be clearly set (Alberini and Segerson, 2002).

10. *Can be quite time-consuming to negotiate the agreement, as well as costly and bureaucratic.*

It is underlined that a lengthy preparation and negotiation process of a VA may lead to a delay in the solution of urgent ecological problems and may even produce serious, irreversible environmental changes (Šauer et al., 2001). Moreover, Vas that relies on carrot approaches hinge on the ability to generate necessary funds and can be socially costly both due to its impact on industry size and the excess burden of taxes needed to create the appropriate funds (Alberini and Segerson, 2002).

How to enhance effectiveness?

Despite the increasing trend towards the instrument of voluntary approaches, few are known about their actual effectiveness since their assessment is hampered by program novelty, lack of data and weak metering and evaluation methods (Mazurek, 1998). The only thing that can be concluded with accuracy is that implementation problems have led to lower than expected environmental results (Mazurek, 1998) and that they do not always involve welfare-enhancing voluntary abatement (Lyon and Maxwell, 2002).

Nevertheless, effectiveness of voluntary approaches - mostly negotiated VAs - can be improved if the instrument embodies some particular features. In particular:

- The VA should be written and have the form of a contract, which is enforceable, either through private or public law to bind the involved parties to their commitments (Pesaro, 2001) that are doable. Moreover, the contract must be published and be made available to all parties capable of fulfilling all agreement conditions (Šauer et al., 2001).
- The wording of the context of a reference, the premises, the implementation deadlines and the starting conditions the objectives and the proposed means of achieving them must be clear and plausible (Pesaro, 2001). Moreover, the environmental goals should be defined on the basis of real and realistic capabilities of the parties, as well as according to national environmental policy plans. There might be inconsistencies if the environmental targets are established by the industrial sector itself.
- The VA should take into account all subjects influenced by an agreement implementation, and should satisfy them all, in three dimensions: interests, process and the personal dimension (Šauer et al., 2001).
- A trustful and independent third party in the goal establishment step needs to be included, that controls the activity phases (i.e. the competitive conditions between agents to avoid a distortion of competition) and checks the achievements of targets (Pesaro, 2001), so that the VA gains the trust and support of the general public (Šaeur et al., 2001).
- Open access to the VA should be imposed in order exclude any anticompetitive, discriminatory use of the VA (Brau et al., 2001).
- A clear mechanism of sanctions must be established for the case that participating parties fail to meet established goals so that to motivate the willingness to attain the environmental goals. Moreover, the threat of sanctions must be credible in order to induce an efficient abatement level and can either imply that the regulator will indeed impose the more costly policy if the target is not met or he will just

impose the existing regulation - the later threat is considered to be the more credible threat than the former (Alberini and Segerson, 2002).

- Transparency during both the negotiation and implementation process is needed with respect to social actors and institutions in order to establish a climate of trust and understanding (Pesaro, 2001).
- Moreover, the VA should respect known and available data and conditions. Moreover, it should provide accessibility to information concerning all the environmental performance at all the phases in order to facilitate an evaluation of measurable and observable goals.
- The free riding agents should be at least partly excluded by the benefits of the industry-wide VA (i.e. taxes are imposed only on non-signatory agents) in order to guarantee that VA's benefits go mainly to the participatory producers (Brau et al., 2001).
- A minimum participation should be introduced for the VA to be operational (Brau et al., 2001). Moreover, a minimum abatement level needs also to be introduced to avoid cosmetic emission abatement, as well as to offset the negative effects on production and profits of a noncompliant behavior.
- Finally, the negotiation process should be politically and ideologically neutral so that VAs represent appropriate tools for efficient implementation of goals in environmental pollution improvements (Šauer et al., 2003). It is considered that VAs function well in an ethical environment since participating subjects perceive this cooperative strategy as a source of long term benefits.

7.2 Voluntary Approaches in Agriculture

Public concerns about the adverse impacts of agricultural activities on environmental quality (i.e. water, soil, biodiversity) have led to the design of various Voluntary Approaches focusing exclusively on environmental improvements on agricultural activities.

The US federal farm "Environmental Quality Incentives" and "Conservation Security" programs are notable examples of such agricultural VAs, aiming to encourage the adoption of improved nutrient management practices, such as drip irrigation, integrated pest management and site–specific farming, by offering farmers green payments (Isik, 2004). In the same context the "Groundwater Management Area" approach, developed in Oregon, aims to reduce nitrogen applications through Best Management Practices (BMPs) relaying on the background threat of mandatory actions if the nitrate contamination at all monitoring wells were not reduced to the established standard until a defined time period (Fleming and Adams, 1997).[11]

Furthermore, the US "Conservation Reverse Program" involves a contract between the USDA and individual farmers in order to withdraw erodible farming from crop production for 10 years and the further establishment of a long-term vegetation cover (i.e. grass, trees) to stabilize the soil, through the provision of rental payments per acre per year (Owen et al., 1998). Finally, the US Natural Resource Conservation Service (NRCS) is a voluntary program that provides technical assistance to farmers by professional conservationist so that they can better set up and maintain a sound conservation program, consistent with the soil needs (Owen et al., 1998).

These examples of agricultural VAs have many similarities with the EU agri-environmental measures, which involve an annual aid per hectare to farmers who, for at least five years, use agricultural production methods designed to protect the environment and maintain countryside

Having described VAs in the previous sections we continue in this section, in order to obtain better insight into the structure and the mechanisms of the VAs, by presenting the approaches of modeling voluntary agreements from the agriculture's point of view, both in static and dynamic context.

7.2.1 Static Context: Individual and Multiperson Voluntary Approaches

It is reasonable to consider that both the regulator and the individual farmer decide to initiate or participate in a voluntary action if and only if the payoff under the VA is greater or at least equal to the payoff in the unregulated or mandatory state.

From the regulator's perspective, a VA can increase the social net benefits ($NSB_v(\mathbf{x}_i)$) since it may solve environmental problems effectively rather than ineffectively (Šauer et al., 2001). On the other hand, the individual farmer i's net benefits under the VA ($\pi_i^v(\mathbf{x}_i)$) can be increased either through an increase in sales due to the improved

[11] In some US states farmers are offered reduced property taxes in order to adopt soil-conserving BMPs (Helfand and House, 1995).

image with respect to environmental issues or through a reduction in expenses via savings on materials, energy, lower risks etc. (Šauer et al., 2001) or the preemption of mandatory approaches that impose unwanted net costs making farmers worse off compared to the absence of the policy (Alberini and Segerson, 2002).

Hence the necessary conditions for each side, in order to proceed in such an environmental voluntary approach, are:

$$\text{Regulator: } NSB_v(\mathbf{x}_i) \geq NSB_s(\mathbf{x}_i) \tag{7.1}$$

$$\text{Farmer: } \pi_i^v(\mathbf{x}_i) \geq \pi_i^s(\mathbf{x}_i) \tag{7.2}$$

where s represents either the unregulated (UN) or mandatory state (L). Usually, the mandatory state is based on the background threat of regulation that can either be: (i) a pure ambient tax $T_i(e) = t_i(e - e^*)$ or a reduction in governmental subsidies $T_i(e) = B_i - t_i(e - e^*)$ (Wu and Segerson, 2003) imposed with certainty if the aggregate environmental target is not met ($e > \bar{e}$),[12] or (ii) a prespecified mandatory abatement vector $\mathbf{x}_{iL}^{\alpha}$ imposed legislatively under a probability p (Segerson and Micely, 1998). The potential mandatory regulation has many similarities with the codes of good farming of the Nitrate Directive (91/676/EEC) that are mandatory for all the farmers located on areas characterized as vulnerable to nitrate pollution (EC, 1991) and may cover issues such as: construction of manure storage facilities, reduced tillage, establishment of buffer strips near water resources etc. (Wu and Segerson, 2003).

Negotiated agreements

Based on Alberini and Segerson (2002) the conditions (7.1) and (7.2) provide the upper and lower bound on the abatement vector x_{iv}^{α} that is about to emerge under the bargaining process of a bilateral VA. The range of these bounds, however, depends on the magnitude of the background threat and the social costs of the financial incentives. Along with the allocation of bargaining power these factors determine whether the negotiated VA induces the first best abatement vector $x_{iv}^{\alpha *}$ that maximizes the net social benefit under the VA and thus leads to efficient environmental protection.

[12] The aggregate target $\bar{e}$ can be the first-best level e^*.

According to the *bilateral VA model* proposed by Segerson and Micely (1998) the regulator and farmer i (or a sector representative) negotiate in order to define the voluntarily implemented pollution abatement vector $\mathbf{x}_{iv}^{\alpha}$ under the background threat of a legislatively imposed mandatory abatement vector $\mathbf{x}_{iL}^{\alpha}$.[13] Under such a "stick-based approach" the payoff of involved agents is:

$$\begin{array}{ll} \text{Acceptance:} & NSB_v(\mathbf{x}_{iv}^{\alpha}) \text{ and } C_v(\mathbf{x}_{iv}^{\alpha}) \\ \text{Non-Acceptance:} & pNSB_L(\mathbf{x}_{iL}^{\alpha*}) \text{ and } pC_L(\mathbf{x}_{iL}^{\alpha*}) \end{array}$$

where $\mathbf{x}_{iL}^{\alpha*}$ is the abatement vector that maximizes the net social benefit under the legislation and p is the legislation possibility with $0 \leq p \leq 1$. It is considered that the imposition of legislation is not necessarily certain if the agreement is not reached.[14]

Under this context the condition (7.1) determines the minimum $(\mathbf{x}_{iv}^{\alpha})^{\min}$ and maximum $(\mathbf{x}_{iv}^{\alpha})^{o}$ acceptable abatement levels that the regulator can accept under a VA, while the condition (7.2) the maximum value of the abatement vector $(\mathbf{x}_{iv}^{\alpha})^{\max}$ the farmer is willing to accept respectively. Therefore a bilateral voluntary agreement is expected to be the equilibrium outcome of a bargaining process if and only if the minimum abatement vector the regulator is willing to accept is less than or equal to the maximum abatement vector the farmer is willing to accept. Consequently, a necessary and sufficient condition for the attainment of a negotiated environmental agreement is:

$$(\mathbf{x}_{iv}^{\alpha})^{\min} \leq (\mathbf{x}_{iv}^{\alpha})^{\max} \tag{7.3}$$

indicating the existence of a vector $\mathbf{x}_{iv}^{\alpha}$ that is mutually beneficial and thus acceptable to both parties. However, the condition (7.3) does not guarantee that the first best abatement vector $\mathbf{x}_{iv}^{\alpha*}$ is mutually acceptable under the agreement and the actual bargaining outcome depends on the allocation of bargaining power or the nature of the bargaining process (Alberini and Segerson, 2002).

[13] The legislation is more costly both in terms of total and marginal compliance and transaction costs compared to the VA, since the later implies reduced reliance on formal legal procedures and reduced conflict, as well as increased flexibility. The farmer i derives no direct benefit from the VA, he just incurs reduced costs as compared to the regulatory alternative.

[14] The legislation probability is assumed to be exogenous and known.

Depending on the allocation of the bargaining power the abatement vector that is mutually beneficial to involved parties can be even less than legislatively imposed abatement vector $\mathbf{x}_{iL}^{\alpha*}$. In particular, when the regulator has all the bargaining power two different types of equilibria are possible: (i) the first-best abatement vector $\mathbf{x}_{iv}^{\alpha*}$ if the inequality $(\mathbf{x}_{iv}^{\alpha})^{\min} < \mathbf{x}_{iL}^{\alpha*} < \mathbf{x}_{iv}^{\alpha*} < (\mathbf{x}_{iv}^{\alpha})^{\max}$ holds, leading to supercompliance since $\mathbf{x}_{iL}^{\alpha*} < \mathbf{x}_{iv}^{\alpha*}$, or (ii) an abatement vector $\mathbf{x}_{iv}^{\alpha}$ that is less than the first-best vector and it can even be lower than $\mathbf{x}_{iL}^{\alpha*}$ if the inequality $(\mathbf{x}_{iv}^{\alpha})^{\min} < (\mathbf{x}_{iv}^{\alpha})^{\max} < \mathbf{x}_{iv}^{\alpha*}$ holds. On the other hand, if the the bargaining power lies with the farmer then a reduction in efficiency is expected,[15] since the bargaining outcome leads to the farmer's cost minimizing abatement vector $(\mathbf{x}_{iv}^{\alpha})^{\min}$, that is less than the first-best and legislative abatement vector. Finally, if the bargaining power is shared between the negotiated parties then the concluded abatement vector is less than the first-best vector but can be even greater than the legislatively imposed $\mathbf{x}_{iL}^{\alpha*}$.

The bargaining outcome is also affected by the magnitude of the legislative threat (p). It is logical that high values of p lead to abatement vectors higher than $\mathbf{x}_{iL}^{\alpha*}$ and likely equal to $\mathbf{x}_{iv}^{\alpha*}$. However, the magnitude of the background threat in the case of agricultural sector is considered to be low, since there is limited political will regarding the imposition of mandatory controls in agriculture (Segerson and Micely, 1998). In such a case, the voluntarily implemented abatement vector and thus the environmental effectiveness of the agreement can be enhanced through a cost-sharing subsidy (κ) used in combination with the stick-approach. Hence, under the mixed-based approach the involved agents' payoffs can alternatively be:

$$\begin{array}{ll} \text{Acceptance:} & NSB_v(\mathbf{x}_{iv}^{\alpha}) - \varrho\kappa \text{ or } C_v(\mathbf{x}_{iv}^{\alpha}) - \kappa \\ \text{Non-Acceptance:} & pNSB_L(\mathbf{x}_{iL}^{\alpha*}) \text{ and } pNSB_L(\mathbf{x}_{iL}^{\alpha*}) \end{array}$$

where ϱ is the social cost of the subsidy since the necessary funds are raised via distortionary taxes.

In this context the bargaining outcome defines the equilibrium combination $(\mathbf{x}_{iv}^{\alpha}, \kappa)$, which depend on the allocation of the bargaining power, as well as both on the magnitude of parameters p and

[15] This makes sense if the individual farmer i is the representative of the agricultural sector.

ϱ. In particular, when the bargaining power lies with the regulator three equilibrium combinations are possible: $(\mathbf{x}_{iv}^{\alpha *}, 0)$, $((\mathbf{x}_{iv}^{\alpha})^{\max}, 0)$, or $(\mathbf{x}_{iv}^{\alpha}, \kappa)$ with $\mathbf{x}_{iv}^{\alpha} < \mathbf{x}_{iv}^{\alpha *}$ due to the provision of a socially costly subsidy. On the other hand, if the bargaining power lies with the farmer the equilibrium abatement value can never be the first-best $\mathbf{x}_{iv}^{\alpha *}$ and the provided subsidy value κ is higher compared to the one paid if the regulator had the bargaining power. Since the probability p is low a high subsidy value is required to induce a higher abatement vector, depending however on the social cost of funds. Therefore if the social cost ϱ is low the resulting equilibrium abatement vector is higher compared to the vector that would have been accepted without a subsidy $((\mathbf{x}_{iv}^{\alpha})^{\max})$.

Sector Representative

In the previous analysis the negotiated abatement vector was the result of bilateral negotiations between the regulator and an individual farmer i. Nevertheless, the farmer i can also be the representative of a sector or a group of farmers. In such a case the abatement level proposed to the regulator by farmer i must be collectively acceptable or rejected by the negotiating group of farmers. This involves that farmers must reach to a prior agreement regarding the abatement vector x_{iv}^{α} they are willing to accept under the VA, before communicating their final proposal to the regulator (Manzini and Mariotti, 2003). Although farmers are all on the "one side" of the bargaining process they do not always share the same preferences and the regulation may have different impact on each of them. Thus the regulator must ensure the proposed abatement vector is acceptable to all the farmers. Under heterogeneity there is a farmer that entirely drives negotiations even when all other farmers are present and the resulting equilibrium abatement vector $\mathbf{x}_{iv}^{\alpha}$ to some extend depends on his characteristics. Particularly, the farmer with the most aggressive attitude towards environmental control (i.e. the lowest admissible abatement level) determines the outcome of the negotiations, inducing the lowest abatement vector (i.e. $(\mathbf{x}_{iv}^{\alpha})^{\min}$). In other words the agreed abatement vector is the same that would result in bilateral negotiations between the regulator and the most aggressive farmer. It is possible that the "toughest" farmer is a low profit farmer that exploits his weakness to achieve a better deal in negotiations and all other farmers effectively free-ride on him to avoid at a minimum cost the probabilistic legislative intervention. Even though an increase in the probability p could increase the magnitude of the negotiated abatement vector due to the increased cost of rejection for both parties the net effect is ambiguous: the farmers are willing to accept higher abatement vectors, while the regulator is willing to accept a lower vector (Manzini and Mariotti, 2003).[16]

Public voluntary programs

However, the most agricultural VAs do not involve a negotiation process. They are voluntary actions designed exclusively by the regulator and to which individual farmers or their representative can only agree with.

Wu and Segerson (2003) have designed a public voluntary scheme, where the regulator uses the background threat of an ambient tax to induce the individual farmer i to abate pollution so that the prespecified environmental goal e_s is met. Contrary to the previous model the particular VA scheme does not dictate a specified pollution abatement vector $\mathbf{x}^{\alpha}_{iv}$ and if the reliance on the VA appears insufficient to control NPS pollution the mandatory treat is imposed with certainty. It is evident that the choices of the farmer determine whether or not the ambient tax is imposed. Therefore, under such a stick-based public VA the farmer i faces the following costs, when choosing $\mathbf{x}^{\alpha}_{iv}$:[17]

$$C_i(\mathbf{x}^{\alpha}_i) = \begin{cases} C_i(\mathbf{x}^{\alpha}_{iv}, \Theta_i) & \text{if} \quad e(\mathbf{x}^{\alpha}_{iv}, \Theta_i) \leq e_s \\ C_i(\mathbf{x}^{\alpha *}_{it}, \Theta_i) + t(e - e_s) & \text{if} \quad e(\mathbf{x}^{\alpha}_{iv}, \Theta_i) > e_s \end{cases}$$

where the magnitude of the ambient tax is chosen this way to induce the implementation of the abatement vector $\mathbf{x}^{\alpha *}_{it}$ that guarantees the achievement of the target $(e(\mathbf{x}^{\alpha}_{iv}, \Theta_i) = e_s)$, requiring however that the regulator knows the physical characteristics of the farmer (Θ_i).

According to Wu and Segerson (2003) the most US agricultural policies use carrot instruments to induce the voluntary use of environmental friendly practices. In such a case the public VA pays

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[16] If a voluntary agreement is not reached, there is a probability of a legislative intervention and the cost of rejection for each party is the risk of triggering legislative intervention.

[17] In this formulation the expected ambient pollution and thus the cost of abatement depend on the physical characteristics of the farmer (i.e. land quality) (Wu and Segerson, 2003).

the farmer i a pre-specified subsidy κ if the target is met voluntarily, while if $e > e_s$ the background threat involves a reduction of the subsidy. Therefore, under a mixed-based public VA the farmer i faces the following benefits, when choosing $\mathbf{x}_{iv}^{\alpha}$:

$$\kappa = \begin{cases} \kappa & \text{if} \quad e \leq e_s \\ \kappa - t(e(\mathbf{x}_{is}^{\alpha *}, \Theta_i) - e_s) & \text{if} \quad e > e_s \end{cases}$$

As previously the subsidy reduction rate t is set in such a way as to induce the abatement vector $\mathbf{x}_{is}^{\alpha *}$ that guarantees the achievement of the quality target. According to Wu and Segerson (2003) the particular individual voluntary approach can induce the cost-minimizing abatement vector[18] without the need for farm-specific characteristics and regardless the cutoff level of pollution $\bar{e}$.[19]

It is worth mentioning that green payments may be required in order to encourage the voluntary adoption of improved nutrient management practices, since uncertainty about the impact of adoption of different farming systems (and technologies) and the irreversibility of the investment impose important barriers to adoption, even when the investment appears to be profitable (Isik, 2004).[20] However, the increasing reliance on subsidy programs may create expectations for future such programs and the uncertainty about their final implementation may delay the voluntary adoption of site-specific technologies. Therefore the effectiveness of cost-share subsidy policies is enhanced if the regulator enacts such a program immediately, threatens to removed it soon and promises never to restore it again (Isik, 2004).

Multiperson Voluntary Approaches

It has been pointed out that a free-riding problem is likely to emerge if a voluntary program is signed by a sector representative or a group of farmers, which however does not always deter a subgroup of farmers to sign the voluntary agreement and form *a coalition VA*.

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[18] The cost-minimizing vector solves the problem $\min C_i(\mathbf{x}_{iv}^{\alpha}, \Theta_i)$ subject to $e(\mathbf{x}_{iv}^{\alpha}, \Theta_i) \leq e_s t$. For details see Wu and Segerson (2003) page 5.

[19] Wu and Segerson (2003) consider that the target level e_s and the cutoff level $\bar{e}$ differ. In particular, $\bar{e}$ is the level of pollution used for determining total tax payments

[20] Agricultural abatement involves also reversible small scale measures such as change in fertilizer use and other farming practices (i.e. changes in tillage, buffer strips) (Huhtala and Laukkanen, 2004).

It is reasonable to consider that there is an equilibrium number of farmers (K^*) that signs the VA if and only if the payoff for signatories ($\pi_v(K^*)$) is higher or at least equal to the payoff when no VA is signed ($\pi_{NV}(0)$).[21] Thus, based on Brau et al. (2001) the necessary condition for the existence of such an equilibrium coalition K^* is given by:

$$\Pi^v(K^*) \equiv \pi_{iv}(K^*) - \pi_{iNV}(0) \geq 0 \qquad (7.4)$$

where the payoffs are considered to be concave in the number of signatory farmers K, which belong to the interval $[1, n]$.

In the same time, the following stability conditions must hold:

$$\pi_{iv}(K^*) \geq \pi_{iNV}(K^* - 1) \qquad \text{(I)}$$
$$\pi_{iv}(K^* + 1) \leq \pi_{iNV}(K^*) \qquad \text{(II)}$$

implying that exists is no farmer i that has an incentive either to leave (not sign) or join (sign) the coalition VA.

It is notable that if the inequality $\pi_{iv}(K) < \pi_{iNV}(K-1)$ holds for all the range of values $[1, n]$ then there is no equilibrium coalition and $K^* = 0$. On the other hand, if the inequality $\pi_{iv}(K) > \pi_{iNV}(K-1)$ holds for all $K > K^{min}$, with $1 \leq K^{min} < n$, then the equilibrium number of signatory farmers is $K^* = n$, implying that all the farmers that belong in the specific agricultural industry sign the VA and form the so-called "grand coalition".

According to Brau et al. (2001) whether or not the grand coalition is achieved depends also on the profit-maximizing number of signatory farmers $\left(\tilde{K}\right)$ and the value $\hat{K}$ for which the condition (I) is satisfied for all $K \geq \hat{K}$. In particular, if $\hat{K} < n$ then the equilibrium coalition is the grand coalition, implying that the regulator must introduce a minimum participation constraint that requires at least a number of farmers equal to $\hat{K}$ greater than K^{min} to sign the VA. On the other hand, if $\tilde{K} < n$ farmers have an incentive to form a “VA club” (or profit maximizing coalition) that excludes some farmers from the possibility to sign the VA, degrading the environmental effectiveness of the agreement. In such a case the regulator can prevent

[21] According to Brau et al. (2001) $\pi_{NV}(0)$ is the "business-as-usual" payoff, reflecting profits in the unregulated case. Finally, symmetry is considered among farmers.

farmers from adopting exclusive membership rules by establishing an open membership rule.

However, the fact that the benefits of an industry-wide VA (i.e. preemption of a costly regulation, technological cooperation) can be reaped by the non-signatory farmers also affects the farmer's decision to sign the VA. If the signatory and non-signatory farmers are benefited the same amount by the VA, then it is logical that no farmer accepts to sign the VA and the equilibrium coalition is $K^* = 0$. Consequently, the regulator must design the VA in such a way that spillovers to non-signatories are minimized and thus the most benefits are reaped by signatory farmers through a policy mix which penalizes only farmers that do not sign the VA. Finally, the regulator must impose a minimum binding abatement constraint to guarantee the environmental effectiveness of VAs,[22] since if farmers are left free to choose the abatement vector they may proceed in cosmetic abatement in order to maximize their profits (Brau et al., 2001).[23]

Regarding the public VA proposed by Wu and Segerson (2003) it is argued that the cost-minimizing abatement vector does not always emerge in the equilibrium when the VA is offered to the agricultural industry or a group of farmers, since there is a trade-off between the target and the cutoff level.[24] In particular, if the cutoff level is set below the target level ($e_s > \bar{e}$) the regulator guarantees that the environmental target is met voluntarily but not at minimum total costs because of free riding incentive. On the one hand, by lowering the cutoff level the ambient tax payments increase and thus the net gains in terms of avoided tax payments increase farmers' incentive to meet the target voluntarily. On the other hand, a lower $\bar{e}$ level increases

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[22] The Danish CO_2 Agreement Scheme has introduced a minimum environmental tax (Brau et al, 2001).

[23] However under heterogeneous farmers, requiring all the firms to participate is infeasible since high-cost firms are unwilling to participate and the regulator needs to balance carefully the VA terms to induce at least the most firms to participate (Lyon and Maxwell, 1999).

[24] Under multiple farmers the cost-minimizing abatement vector solves the problem: $\min C_1(\mathbf{x}_{1v}^{\alpha}, \Theta_1) + C_2(\mathbf{x}_{2v}^{\alpha}, \Theta_2) + ... + C_n(\mathbf{x}_{nv}^{\alpha}, \Theta_n)$ subject to $e(\mathbf{x}_{1v}^{\alpha}, \mathbf{x}_{2v}^{\alpha}, ..., \mathbf{x}_{nv}^{\alpha}; \Theta_1, \Theta_2, ..., \Theta_n) \leq e_s$. Moreover, as before, if the target is not met then all farmers are subject to an ambient tax, implying that costs are: $C_i(\mathbf{x}_{it}^{\alpha *}, \Theta_i) + t(e(\mathbf{x}_{1v}^{\alpha}, \mathbf{x}_{2v}^{\alpha}, ..., \mathbf{x}_{nv}^{\alpha}; \Theta_1, \Theta_2, ..., \Theta_n) - e_s)$. For details see Wu and Segerson (2003) page 14.

respectively the potential for free-riding since some farmers abate more, allowing other farmers to abate less, increasing the possibility that the emerging Nash equilibrium is not the cost-minimizing.[25] Therefore, given the trade-off between e_s and $\bar{e}$ the regulator needs to balance these two effects when designing the agricultural VA.

7.2.2 Dynamic Context: Individual and Multiperson Voluntary Approaches

Even though the dynamic analysis of voluntary approaches takes place in infinite time, in practice their time horizon involves a limited number of years (Cavaliere, 2000). For instance, the EU agri-environmental measures last for a minimum five year period (EC, 2004). Nevertheless, if it is assumed that such voluntary actions can be renovated, the use of an infinite time horizon is justifiable.

Individual Voluntary Approaches

In a dynamic context the methodology of negotiating a voluntary approach provided by Segerson and Micely (1998) can be extended with a second period representing the entire future time horizon in order to incorporate surprise and non-surprise features.

According to the modified negotiated VA model developed by Langpap and Wu (2004) the regulator and farmer i negotiate in period 1 about the voluntary combination of abatement vectors $(\mathbf{x}_{1v}^{\alpha}, \mathbf{x}_{2v}^{\alpha})$ to be implemented in period 1 and 2 respectively. Involved agents are aware that if the agreement is not reached then a mandatory combination of abatement vectors $(\mathbf{x}_{1L}^{\alpha *}, \mathbf{x}_{2L}^{\alpha *})$ is to be legislatively imposed in period 1 and 2 respectively under an exogenous probability p. During the negotiation in period 1 the involved agents take into consideration the current status of pollution S_1 which is known, while form expectations about the status of pollution S_2 in period 2 given the uncertainty about the future environmental conditions. This implies that the voluntary abatement vector $\mathbf{x}_{2v}^{\alpha}$ is determined before the state S_2 is observed, on the contrary to the legislatively imposed abatement vector $\mathbf{x}_{2L}^{\alpha *}$ that is set on a period-by-period basis to incorporate the observed changes .

It is logical to consider that if during period 2 the nitrate pollution

[25] The regulator can deal the free riding problem by setting $e_s = \bar{e}$, which however ensures that the zero abatement vector can also be a Nash equilibrium since farmers incur the same costs whether the target is met voluntarily or under the ambient tax.

in a nearby lake increases more than expected, due to unpredictable environmental factors, then the regulator should require the farmer i to increase the land set-aside or the size of the buffer strip. This is feasible only when the concluded VA does not contain a "no-surprise" provision that binds the regulator on the agreed abatement vector $\mathbf{x}_{2v}^{\alpha}$ regardless of the new available information. In particular, under a "surprise VA" there is a probability q that the regulator revises the agreed vector and actually imposes the welfare maximizing mandatory vector $\mathbf{x}_{2L}^{\alpha *}$ that can be higher or even lower than $\mathbf{x}_{2v}^{\alpha}$.

Therefore, under uncertainty about the future state of the world a surprise VA is mutually accepted by involved agents if and only if the following inequalities hold simultaneously:[26]

$$\begin{array}{ll}\text{Farmer:} & C_v^1(\mathbf{x}_{1v}^{\alpha}) + q\mathcal{E}C_L^2(\mathbf{x}_{2L}^{\alpha *}) + (1-q)\mathcal{E}C_v^2(\mathbf{x}_{2v}^{\alpha}) \\ & \leq p\left[C_L^1(\mathbf{x}_{1L}^{\alpha *}) + \mathcal{E}C_L^2(\mathbf{x}_{2L}^{\alpha *})\right] \\ \text{Regulator:} & NSB_v^1(\mathbf{x}_{1v}^{\alpha}) + q\mathcal{E}NSB_L^2(\mathbf{x}_{2L}^{\alpha *}) + (1-q)\mathcal{E}NSB_v^2(\mathbf{x}_{2v}^{\alpha}) \\ & \geq p\left[NSB_L^1(\mathbf{x}_{1L}^{\alpha *}) + \mathcal{E}NSB_L^2(\mathbf{x}_{2L}^{\alpha *})\right]\end{array}$$

implying that the farmer enters into the agreement if his cost under the VA is no larger than his expected costs under the mandatory regulation, while the regulator enters the agreement if the expected net social benefit under the VA is higher than the expected net social benefit under the regulation.

It is reasonable that the regulator is more willing to enter into a surprise VA since he retains the flexibility to use the new information and thereby increase the social net benefit. On the other hand, the effect of surprise provisions on the farmer's willingness to sign the VA are ambiguous, since it is affected by his expectations about the surprise abatement vector $\mathbf{x}_{2L}^{\alpha *}$ compared to the initially agreed vector $\mathbf{x}_{2v}^{\alpha}$. In particular, if $\mathbf{x}_{2L}^{\alpha *} > \mathbf{x}_{2v}^{\alpha}$ then the expected costs are higher and the farmer i is more willing to enter into a VA that provides assurances that no surprises are possible in period 2. Although the provision of assurances encourages participation in the VA it does not allow to use the new available information, a fact that might undermine the efficiency of the agreement since it it is likely that the future abatement vector $\mathbf{x}_{2v}^{\alpha}$ is lower than $\mathbf{x}_{2L}^{\alpha *}$. Therefore,

[26] The conditions under which a no-surprise VA is concluded are obtained by setting $q = 0$. In both cases the terms $\mathcal{E}C_L^2(\cdot)$ and $\mathcal{E}NSB_v^2(\cdot)$ represent agents' expectations about their payoff in period 2 since none knows in period 1 the future state of the world.

the regulator faces the dilemma of whether to provide assurances or not when negotiating a voluntary action with an individual farmer.

Individual public VA

In an infinite time horizon the regulator can use a trigger strategy in order to induce voluntary abatement of nitrate pollution in each time period (Wu and Segerson, 2003). In particular, during the first period no tax is imposed and the individual farmer is left free to decide whether or not to meet voluntarily the standard e_s. However, if at the end of the first period the ambient target e_s is not met then an ambient tax is imposes for all the remaining periods, implying that the regulator gives no second chance to the farmer to meet the target at no additional cost.

Under such a knowledge the farmer decides whether or not to voluntary abate pollution during the first period by comparing the present value of the infinite stream of total costs under the VA and the ambient tax. Hence the individual farmer i decides to meet the ambient target voluntarily if and only if the following inequality holds:

$$\begin{aligned} C_i(\mathbf{x}_{iv}^{\alpha}, \Theta_i)\left[\delta + \delta^2 + \delta^3 ...\right] &> \left[C_i(\mathbf{x}_{it}^{\alpha *}, \Theta_i) + t(e - e_s)\right]\left[1 + \delta + \delta^2 ...\right] \\ C_i(\mathbf{x}_{iv}^{\alpha}, \Theta_i)\frac{1}{1-\delta} &> \left[C_i(\mathbf{x}_{it}^{\alpha *}, \Theta_i) + t(e - e_s)\right]\frac{\delta}{1-\delta} \end{aligned}$$

Therefore the target is met voluntarily if and only if the discount rate (r) is sufficiently low.[27] Moreover, provided that r is low enough the cost-minimizing abatement vector is the unique optimal response if and only if the cutoff level $\bar{e}$ is set sufficiently low so that the present value of the stream of tax payments exceeds the cost-savings of the first period's noncompliance (Wu and Segerson, 2003).

Multiperson / industry wide VAs

The decision to ex-post free-ride (non-compliance) is as significant as the ex-ante free-riding decision (non-participation), since a signed VA may be led to a failure before the potential benefits are realized if signatory farmers decide not to comply with the agreement's

[27] Where $\delta = 1/(1+r)$ is the discount factor.

provisions. Nevertheless, under the proper design of the industry-wide VA the regulator can induce the majority or even the entity of the agricultural sector to both participate in and comply with the agreement.

In this context Xepapadeas and Passa (2005) have analyzed the long-run structure of a public VA offered by the regulator to a large number of homogeneous farmers in order to choose the cost saving methods by which the voluntary emission level (e_v) and the targeted ambient pollution path $\bar{S}$ are both attained. Farmers are aware that the regulator has full observability of participating farmers and thus in the event of ex-ante free riding, leading to a deviation $\Delta S(t)$ from the ambient target, a legislatively imposed regulation (e_L) may be triggered under a probability p. Nevertheless, participating farmers know that it is prohibitively costly for the regulator to simultaneous control the compliance status of all signatory farmers. Therefore, if they decide ex-post not to fulfill the VA's provisions then there is a probability q that they will be subjected to an individual legislation and a non-compliance fine after a random inspection, otherwise they can keep emitting at the profit-maximizing emission level e_o.[28]

Both the legislation and audit probability are endogenous dependent on the state variables of the problem. Nevertheless, the probability of inspection can be also exogenously fixed. Hence, in their general form the probabilities are given by:

$$p(t) = p(\Delta S(t), x(t), \boldsymbol{\omega}_v(t)) \text{ and } q(t) = q(z(t), \Delta S(t), \boldsymbol{\omega}_c(t)) \text{ or } \bar{q}$$

where $\omega_v(t)$ and $\boldsymbol{\omega}_c(t)$ is a vector of other parameters affecting the probabilities, x and z is the proportion of participating and complying farmers.[29] Finally, to be more realistic the auditing probability is dependent on the regulator's budget $B_t = B(K, S, z, F)$ and is given by $q = q(z, \Delta S, B, \boldsymbol{\omega}_c)$.

Individual polluting farmers' decisions about whether or not to participate in and comply with the VA are based on the evolutionary

[28] According to the authors a similar type of regulatory framework can be considered for the EU Nitrate Directive (91/676/EEC) that aims to reduce water pollution caused by nitrates generated from agricultural sources.

[29] It holds that $\frac{\partial p(\cdot)}{\partial \Delta S}, \frac{\partial q(\cdot)}{\partial \Delta S} > 0$ and $\frac{\partial p(\cdot)}{\partial x}, \frac{\partial q(\cdot)}{\partial z} < 0$ with x and $z \in [0,1]$. It is notable that if $p(\Delta S, 1 \,|\Delta S > 0) = 0$ holds then the deviation is due to non-compliance. Moreover, if nobody complies the value of $q(z = 0, \boldsymbol{\omega}_c)$ will be large but not unity since not every farmer is audited.

processes of comparing expected profits associated with the different decisions, modelled by replicator dynamics. In particular, when it comes to choosing a strategy regarding participating in the VA programme, or complying with the VA's provisions farmers adopt a more passive decision making by comparing the profits of a strategy to participate in and comply ($\Pi_v(e_v)$) with the corresponding expected profits of a non-participating ($p\Pi_L(e_L) + (1-p)\Pi_N(e_o)$), or non-complying strategy ($q\Pi_C(e_L, F) + (1-q)\Pi_N(e_o)$). Successful strategies, in the sense of those attaining higher expected profits, are imitated by other farmers with a probability proportional to the profit difference and increase their respective share in the total population of farmers.

Such movements in the composition of the population of farmers regarding participation in and compliance with the VA are described by the replicator dynamics equations (7.5) and (7.6), given below, that operate in fast and slow time scales. It is considered that decisions to participate or not are finalized relative fast due to legislative procedures, while the decisions to comply or not are unconstrained and thus expected to evolve much more slowly, implying that the equilibrium composition of participating farmers (x^*) is reached faster than the equilibrium composition regarding compliance (z^*). The term ε is a small positive parameter used to distinguish these different time scales.

According to Xepapadeas and Passa (2005) the described evolutionary approach with fast-slow selection dynamics is used to determine jointly the steady-state equilibrium fraction of signatory and complying farmers, as well as the corresponding steady-state equilibrium pollution stock (S^*) under different assumptions about the legislation and inspection probabilities and provides useful recommendations about the proper policy design. The associated dynamic system in the general form is given by:[30]

$$\varepsilon\dot{x} = x(1-x)\left[p(x,\Delta S)\Delta\Pi_L^N - \Delta\Pi_v^N\right] \tag{7.5}$$

$$\dot{z} = z(1-z)\left[q\left(z,\Delta S,\boldsymbol{\omega}_c\right)\Delta\Pi_C^N - \Delta\Pi_v^N\right] \tag{7.6}$$

$$\dot{S} = n\{x[ze_v + (1-z)\mathcal{E}e_L(q\left(z,\Delta S,\boldsymbol{\omega}_c\right))] + (1-x)e_o\} - bS \tag{7.7}$$

[30] When $\varepsilon \to 0$ algebraic equation (7.5 defines the slow manifold. On this manifold the slow system (7.6) and (7.7) evolves and approaches steady states.

where the term $\Delta\Pi_L^N$ represents the profit losses under the non-participating strategy when legislation is imposed, $\Delta\Pi_v^N$ the profit losses under the participating, complying strategy and $\Delta\Pi_C^N$ the profit losses under the non-complying, participating strategy when both legislation and fine are imposed.[31] Finally, (7.7) is the pollution stock dynamic equation and $\mathcal{E}e_L(q(\cdot)) = qe_L + (1-q)e_o$ are the expected emissions of a non-complying but participating farmer.

The main finding is that the structure of the legislation and auditing probability, the levels of legislative emissions e_L and non-compliance fines F are the main factors characterizing the evolutionary equilibrium outcome (Xepapadeas and Passa, 2005). The more complex the structure of the legislation and audit probability is, the more likely is that the evolutionary equilibrium implies partial participation and partial compliance, and dependents largely on the initial conditions. In particular, under no binding budget constraint regarding monitoring costs, commitment to legislation and auditing probabilities that are both set higher than the associated critical values $\left(p(\overline{\Delta S}) > \hat{p}(\overline{\Delta S}) \text{ and } \bar{q} > \hat{\bar{q}}\right)$, the regulator can induce full participation ($x^* = 1$) and compliance ($z^* = 1$) with the public VA. The same outcome can be achieved under certain initial conditions and properly chosen legislative mandate and non-compliance fines when the auditing probability depends on the nitrate pollution stock S and the complying proportion z.

However, if these conditions are not fulfilled or the available budget is limited then partial participation, partial compliance with multiple equilibria and irreversibilities and even fluctuation in the nitrate pollution stock are possible evolutionary outcomes.

In the previous public VA model compliance could be enhanced either through a higher number of inspections or a higher non-compliance fine. Nevertheless, high compliance rates can be feasible under low inspection probabilities and small fines, if imposed at all (Friesen, 2003). According to Friesen (2003) noncompliance can be deterred through targeting schemes that divide regulated farmers into two groups: *target group* (G2) and *non-target group* (G1). Such schemes can be either based on farmers' past compliance record or on random targeting. Under the past compliance targeting scheme, the

[31] Where $\Delta\Pi_L^N = (\Pi_N(e_o) - \Pi_L(e_L))$, $\Delta\Pi_v^N = (\Pi_N(e_o) - \Pi_v(e_v))$ and $\Delta\Pi_C^N = (\Pi_N(e_o) - \Pi_C(e_L, F))$.

farmer i is moved into the target group if noncompliance is revealed during an inspection, otherwise he is moved into the non-target group as a reward for compliance. Inspections in the G2 group are more frequent than in group G1 ($q_2 > q_1$) and the regulator can increase the compliance incentive in G2 by affecting the fines and transition probabilities between the groups. In particular, by allowing complying farmers to escape group G2 more quickly than noncompliant farmers and spending more time in G1, the regulator can induce compliance in G2, at least for some of the time, since such a behavior increases the escape probability into the G1 group. Moreover, by imposing no penalty in the G1 group ($F_1 = 0$) and the maximum fine is imposed in G2 group ($F_2 = F^{\max}$) the compliance incentive is maximized.

Finally, under the optimal targeting scheme the farmer i is randomly moved into the target group. In such a case the farmer can escape from G2 and move back to G1 on the basis of observed compliance behavior during an inspection. By randomly selecting farmers for the G2 group, more frequent inspections in the target group are needed while inspections in the non-target group have no additional deterrent effect in the target group, implying that $q_2 > 0$ and $q_1 = 0$.[32] It is likely that even though the optimal targeting scheme involves less enforcement costs, the past compliance targeting scheme has wider range of applicability, since it can be used for "large" partial compliance rate goals (Friesen, 2003).

[32] As previously, the compliance incentive is maximized if $F_1 = 0$ and $F_2 = F^{\max}$, as well as if a compliant farmer is moved immediately into the G1 group.

8

Conclusions

Undoubtedly EU agriculture is very important from an environmental point of view. Agricultural activities can be associated with several environmental services such as, the maintenance of many cultural pastoral and arable landscapes, the decline of greenhouse emissions, gains to biodiversity, sustainability and resource management.

However, EU agriculture can be also associated with a series of adverse environmental effects such as:[1]

- *Loss of biodiversity* due to the drainage of wetlands, irrigation of arid lands and the ploughing up of unproved grasslands.
- *Loss of landscape diversity and quality* as well as *deterioration of important habitants and decline in number of species* mainly due to the increasing scale of production, homogenization of landscapes, and intensively managed and irrigated farmlands.
- *Threats to high natural value farming systems* and *traditional forms of agriculture in marginal areas* due to economic trends, farm enlargement and intensification.
- *Soil quality pollution* (i.e. salinization, erosion, acidification) due to the bad soil management because of the selection of erosion prone crops, use of heavy machinery, intensive irrigation and intensive monoculture practices.
- *Air pollution* (i.e. ammonia, greenhouse gases).
- *Water pollution* (i.e. eutrophication, salinization).

In recognition of the above problems Pillar II reflects the environmental concerns in the EU and develops a regulatory framework with the purpose of improving environmental conditions related to the

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[1] In this context, potentially one of the most notable pollution problems, facing the EU agriculture which has been widely discussed, in the environmental economics literature is *nitrate leaching* (NO_3), the nitrate removal from the soil by the action of water, which ends up either through runoff or deep percolation into water bodies.

agricultural sector. The regulatory framework forming EU agricultural policies contains instruments related, sometimes very closely, to instruments that have been proposed and developed in the environmental economics literature for regulation of environmental problems regarding agriculture.

In this context our study provides a survey of environmental policies which can be used to control agricultural pollution, and tries to explore the existing links of these policies with EU policies and, furthermore, to provide some insights regarding the potential incorporation of more policy instruments from the existing set developed by environmental economics, into actual EU policy schemes.

The major characteristic of agricultural pollution is its NPS character, which on the one hand constraints the use of available policy instruments and on the other makes necessary the development of new instruments. In the environmental economics literature pollution problems are classified as point-source (PS) or non-point-source (NPS) problems. Agricultural pollution is a typical NPS problem due to associated uncertainty about polluters and the degree of each agent's contribution to total pollution, a fact that eliminates emission-based instruments from the set of available NPS pollution instruments. The origins of this uncertainty can be attributed to: (i) stochastic influences that affect fate and transport of pollutants due to variability in natural processes and / or technological uncertainty, (ii) the great number of static sources of pollution emissions (the fundamental relationship between polluted area and pollution source is not known with certainty), and / or (iii) the regulator's inability to infer individual emissions from ambient pollution levels or inputs used, due to monitoring and measurement inefficiencies resulting from budgetary restrictions or legal restrictions.

These characteristics of NPS agricultural pollution limit the range of potential policy instruments and also the efficiency of many remaining options. Standard instruments, such as Pigouvian taxes, tradeable permits and emission standards, appear to be inadequate to handle efficiently NPS pollution problems. Hence NPS control has focused on other elements of NPS pollution problems that may be observable such as: *polluters' decisions* (input-based schemes) or the *consequences of their actions* (ambient-based schemes). Moreover, agricultural NPS problems, such as nitrate pollution, can be handled via Voluntary Approaches (VAs), liabilities for damages, markets and moral suasion.

Since the interdependence between agriculture and environment is becoming more apparent, the formal modeling of agricultural pollution is becoming necessary and its mostly done in the context of optimization models, which have been regarded as providing and adequate description of the mechanisms underlying the agents actions (farmers, regulators), even if actors are not strictly speaking maximizing.[2] These maximization problems faced by farmers and regulators, can be defined under various contexts: static, dynamic, asymmetric information, as well as under certainty or uncertainty.

In particular, under certainty the ambient pollution is a strictly increasing function in productive inputs and strictly decreasing in abatement inputs, while under uncertainty the static model is augmented by stochastic elements (i.e. environmental variables, site characteristics) that may affect both emission generation and emission abatement. On the other hand, in a dynamic context the deterministic model is described by the evolution of the pollution stock, which is determined by collective emissions generated in each time period and the amount of pollution removed through natural processes. Uncertainty reflecting for example random natural decay rate, can be introduced into the model through a Brownian motion. Finally, under asymmetric information ambient pollution is a stochastic function of input choices and the type of the discharger (i.e. ability, soil composition, costs of abatement) containing also a random variable reflecting observation errors of individual emissions.

As mentioned above it is considered that involved agents (farmers and regulator) adopt an explicit maximizing behavior, seeking to achieve the allocation of resources that maximizes their respective payoff functions. In a static context and a certainty framework, farmers choose the input vector that maximizes their private net benefits, while the regulator seeks to achieve the ex-ante efficient allocation of resources that maximizes social net benefit resulting from agricultural operations. On the other hand, in a dynamic context the farmers seek to maximize the present value of their payoff function

[2] This is the well known "as if" argument in economic theory, where agents are regarded as if maximizing some objective function, since the outcome of their observed behavior can be explained in terms of maximizing behavior. Thus models based on maximizing behavior can be used to explain the underlying mechanisms that generate observed data.

under different behavioral rules regarding the evolution of pollution stock: myopic, open-loop and feedback. The regulator seeks to maximize the present value of social welfare by choosing the optimal path of the input vector for each farmer, subject to transition equations reflecting the dynamic constraints of the problem. Under uncertainty the farmers' and regulator's static and dynamic problem is expressed in expected value terms.

After comparing the optimality conditions resulting from the regulator's and farmers' problems in the various contexts, it is evident that unregulated farmers over-pollute, since they do not take into account the external effects of their production choices on ambient environment. Hence, in the absence of policy interventions unregulated market fails to deliver the efficient allocation of resources, a standard and well known result in economic theory, justifying the introduction of environmental policies.

A number of approaches have been developed over the last two decades for dealing with NPS pollution problems. These approaches lead to the policy schemes discussed extensively in this survey, namely:

- Effluent-based schemes, where emissions, when it is possible to be observed or inferred with some accuracy, is the basis for the design of alternative instruments.
- Input-based Schemes, where inputs which contribute to agricultural NPS pollution is the basis for the design of alternative instruments.
- Output-based Schemes, where the output of production processes which contribute to agricultural NPS pollution is the basis for the design of alternative instruments.
- Ambient-based Schemes, where in the absence of information about the farmers individual contribution to agricultural NPS pollution, the ambient pollution levels measured at receptor points is the basis for the design of alternative instruments.
- Voluntary Approaches. This a new instruments, which however is used in EU agricultural policies, and where the basis for regulation is a voluntary agreement between farmers and regulators with the ultimate purpose of reducing agricultural NPS pollution.

> The above instruments are extensively analyzed in the corresponding chapters, and their advantages and disadvantages, the cases where they have been used to control agricultural pollution, and their links with existing EU policies are presented.

It should be noted at this point that the empirical applications related to the above policy rules require estimation of the associated production or cost functions, so that policy experiments will be possible. Policy experiments will produce the changes in the input vectors and the changes in output due to a policy change. In our case policy changes relate mainly to the degree of decoupling (full/partial) and the land -set-aside policies. Through policy experiments it will be possible to obtain information regarding changes in inputs use at the farm and aggregate level, which can then be used as a basis to infer individual emission levels. This is the transformation of a NPS pollution problem to a PS pollution problem. In the context developed in this survey, the approach is based on estimating econometrically the required functions. Similar information can be obtained by a programming approach (LP, PMP) which is the main type of modelling for empirical policy analysis in GENEDEC. For example the $AROPAj$ model, models a producer's maximization programme, which maximizes gross margins subject to a set of technical constraints and policy constraints. The $AROPAj$ model uses as inputs, prices and technical and CAP-related parameters and produces as output: optimal land use, produced quantities (e.g. livestock numbers) optimal feeding, net emissions. In $AROPAj$ parameter estimates can be improved through a calibration procedure that involves re-estimation. Thus obtaining environmental impacts, such as for example, changes in organic manure (N), or total N pollutants, due to decoupling, expected emissions can be approximated and then used to infer individual emissions. Thus one can argue that the transformation of a NPS pollution problem to a PS pollution problem can be handled either through standard econometric technics, which estimate the parameters of smooth functions involved in optimization models, or through models based on linear programming.

It seems that environmental economics provide a substantial body of environmental policy instruments to deal with agricultural NPS pollution control, furthermore, since the area continues to receive lots of attention from researchers, new advances are taking place and new results are derived. On the other hand the EU actually tries to reduce agricultural pollution through the development and implementation of policy schemes. It seems that there is a substantial area for cross fertilization where theoretical models can be motivated by actual situations arising in the EU agricultural sector in its relation to the environment, so that new theoretical solutions and policy schemes emerge, while policy makers can use theoretical results to introduce new policy schemes, or to improve of fine tune existing, with the ultimate purpose or substantially improving the European environment. It is hoped that the present survey provides some insights that could help towards the attainment of this goal.

9

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