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EFFECTS OF AGRICULTURAL POLICIES AND PRACTICES ON THE ENVIRONMENT: REVIEW OF EMPIRICAL WORK IN OECD COUNTRIES

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EFFECTS OF AGRICULTURAL POLICIES AND PRACTICES ON THE ENVIRONMENT: REVIEW OF EMPIRICAL WORK IN OECD COUNTRIES

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PUBLISHER'S NOTE
The views expressed in this document are those of the author, and do not necessarily reflect those of the Organisation or its Member countries

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FOREWORD

This report was prepared by Dr Floor Brouwer, LEI, The Netherlands, for discussion in the Joint Working Party on Agriculture and the Environment.

This document is made available to the public as a consultant's report. The opinions expressed and the arguments employed in this document are the sole responsibility of the author and do not necessarily reflect those of the Committee for Agriculture and the Environment Policy Committee or the governments of OECD Member countries.

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EFFECTS OF AGRICULTURAL POLICIES AND PRACTICES ON THE ENVIRONMENT: REVIEW OF EMPIRICAL WORK IN OECD COUNTRIES

by Floor Brouwer, LEI, The Netherlands

Introduction

Market interventions and subsidies have been key instruments in agricultural policy for most OECD countries. Major instruments applied are market-price support confered through border measures (*e.g.*, tariffs, export subsidies, quantitative and qualitative restrictions) and administrative pricing regimes, other production-linked supports such as deficiency payments, and subsidies for intermediate inputs (Krissoff *et al.*, 1996). Direct and indirect interactions between agricultural support measures and the environment are of major societal interest. Understanding these interactions can be gained by studying the experience of existing policy measures actually implemented in OECD countries.

Numerous studies have looked at the effects of agricultural policies on the environment, from both a quantitative and qualitative perspective. Many are site- or commodity-specific. This report review the approaches used and methodologies applied in these studies, based on an examination of a selection of the literature (particularly from North America and Europe). Emphasis is given to an overview of empirical work, examining main environmental issues addressed, approaches used and findings achieved. Such an effort would gain our understanding regarding linkages between agricultural policy measures, agricultural practices and environmental quality. It is intended to provide guidance on quantitative work that could be undertaken in the OECD to analyse the environmental effects of agricultural policies. The main purpose of such work would be to understand and quantify the effects of different policies on the environment, in order to provide advice on what constitutes a "good policy" — *i.e.*, best for the environment, agriculture and trade.

Assessing the environmental effects of policy reform[1]

Agriculture interacts dynamically with the available factor endowments in response to technological change. The general criterion is that costs of (variable or non-factor) inputs need to offset the marginal returns of inputs. The trend in the developed world has generally been to replace brute labour with machinery, agrochemicals (fertilisers and pesticides) and information. Substitution can follow different pathways, depending on which factor inputs are most binding. A comparison of technological change in agriculture in the USA and Japan during the century to 1980, for example, indicates that both countries witnessed the substitution of relatively abundant technology for scarce factors (Kawagoe *et al.*, 1986). The labour-saving effect was substantial in Japan, with a large increase in the use of both machinery and

1. The driving-force–state–response (DSR) framework has been developed to link driving forces (market signals and government policies) to state conditions (environmental conditions that arise from these driving forces) and responses (changes in farm behaviour and in government policies).

fertilisers. Technological change in the USA also resulted in a major increase in the share of machinery, but its absolute effect on the use of fertilisers was small because the increase in their factor share started from a small base.

Tangermann and Buckwell (1999) emphasise the methodological difficulties of determining the effects of agricultural policies or their reform. They criticise empirical approaches that compare conditions before and after a policy change. The problem essentially is to filter the policy component from the many other factors that also shape outcomes. Consider, for example, the impact of the 1992 MacSharry reforms on the arable crops sector. Arable crops account for a considerable share of the budget of the Common Agricultural Policy (CAP) — 16.1 billion euros out of a total expenditure of 40 billion ECU — and of overall support.[2] One of the objectives of this reform was to reduce surplus production and to restore balance in the market for cereals. The share of cereals used as animal feed was expected to increase, following a reduction in intervention prices.

The use of cereals in compound feed produced in the EU did increase during the first half of the 1990s: between 1991/92 and the end of the 1990s total consumption of cereals in the EU increased by around 20 million tonnes (a rise of approximately 15%). The majority of the increase was used for feed. But the reform of the arable crops regime was responsible for only a part of this increase. Other relevant factors included changes in global market conditions for raw materials (Brouwer and Hoogeveen, 2000) and in currency exchange rates, both within the EU and between the euro and the US dollar. High prices for soybean products in late 1997, for example, increased the demand for alternative sources of protein. In the wheat market, reduced stockpiles in the USA, combined with production declines in the former Soviet Union and increases in the demand for cereals by China, raised the price of wheat traded on world markets, thus reducing the need for market intervention in the form of export subsidies. The impact of the MacSharry reform might also have been much bigger than what is observed in the statistics. Beef production fell in the EU during the 1990s, which again reduced the demand for feed and might have also influenced the market of compound feed.

The situation facing farmers differed also within EU Member States. In the UK, for example, the reduction in intervention prices was not translated into a correspondingly large reduction in farm-gate prices. This was largely due to the declining trend in world supplies and increasing prices, combined with a weakening in the pound Sterling relative to the U.S. dollar (Winter, 2000).

Two other considerations need to be taken into account when assessing the environmental effects of changes in farming practices. The first is *the time-delay between changes in farming practices and their impacts on the physical and biological environment.* Changes in government policies (support measures, economic instruments, command-and-control measures) and market signals primarily affect farming practice (*e.g.* fertilisation, application of pesticides, transition of production practices towards organic farming). Beneficial effects may be observed only long after mitigation policies are introduced. Lake eutrophication, for example, may continue long after farmers reduce their fertiliser use. Figure 1 shows this phenomenon for a lake in Finland. Following the adoption of agri-environmental measures (including conservation tillage and filter strips of vegetation to prevent nutrient loads from surface runoff) in the mid-1990s, phosphorus loading from agriculture fell by about 20%; however, phosphorus concentrations in the lake continued to rise and eutrophication has not stopped yet. Model estimates indicate that the external phosphorus load must be reduced by some 40% to stop eutrophication.

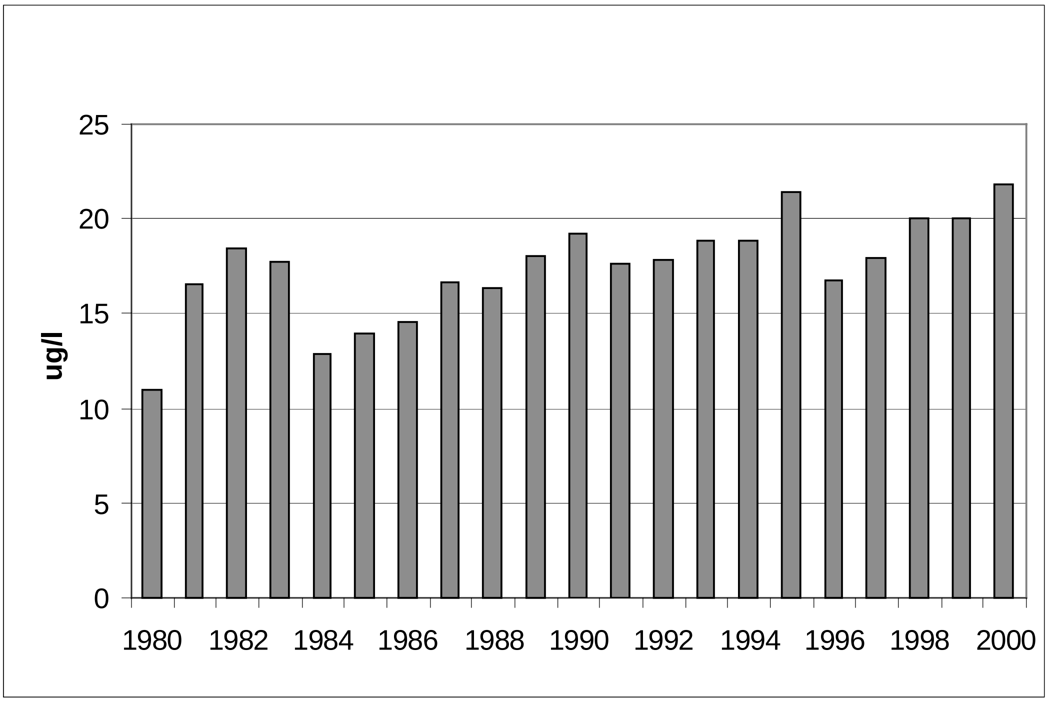
The second consideration is *the site-specific and transboundary nature of changes in farming practices that impact the physical and natural environment.* Hydrogeographic and climatic conditions and soil

2. The Producer Support Estimate, or PSE, for wheat produced in the EU is estimated to have averaged around 48% over the period 1998-2000 (OECD, 2001*a*).

characteristics affect the nature of ultimate impacts of changes in farming practices on the achievement of water-quality standards. Normally the delay between the implementation of a measure to control nitrate pollution and an improvement in groundwater nitrate concentrations can be measured in decades. However, the response time can sometimes be less than 10 years if geological conditions are favourable. As shown in Figure 2, nitrate concentrations as measured in two wells on a small island in Denmark increased rapidly during the 1980s following a change in leek culture (conditions became favourable to grow them year-round). Leeks require large amounts of nitrogen, and nitrate levels increased from around 35 mg per litre in 1982 to more than 150 mg per litre by the end of the 1980s. The subsequent reduction in nitrate concentrations (from 150 to 50 mg per litre) followed from the introduction of a water-protection zone. Part of the land was purchased to restrict farming practices, and some farmers shifted into extensive (grass-based) livestock production. But the island's topography and its hydrogeographic conditions have been the main reason for the short response time: annual precipitation (around 500 mm) is below potential evapotranspiration (550 mm): any surplus water percolates into the groundwater. The shallow depth of the groundwater on the island and the rapid recharge rate are the main reason for the rapid response time.

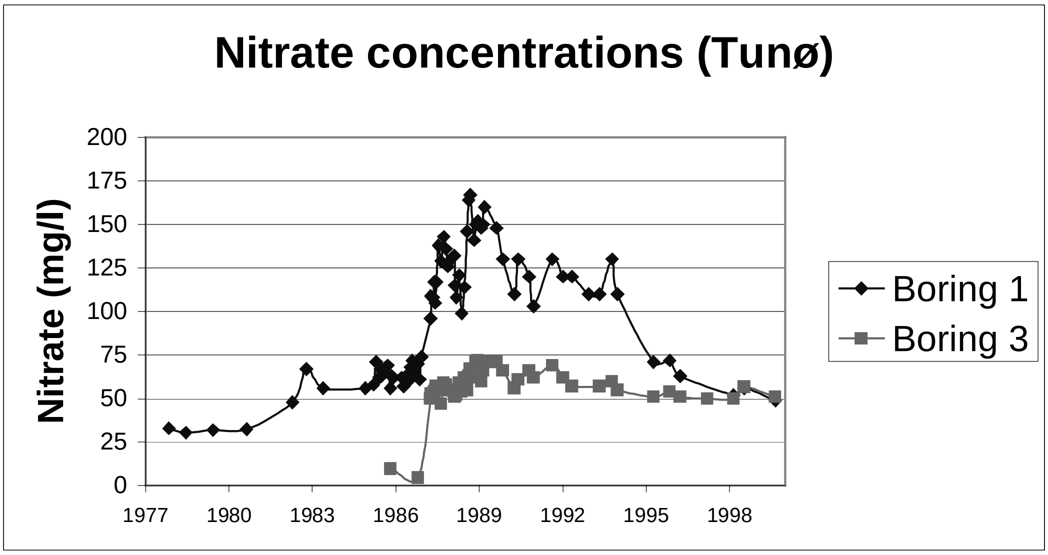
Figure 1. Phosphorus concentration in lake Pyhäjärvi in 1980-2000

(mean of the open-water period)



Source: Ventelä *et al.* (2000).

Figure 2. Nitrate concentrations in two wells on the Danish island of Tunø



Source: Aarhus County.

Table 1. Criteria for the assessment of policy impacts

	Effects	
Immediate	***Intermediate***	***Ultimate***
	Measures aimed at control of fertiliser practice	
• Reduced application of nutrients, including buffer strips • Increase facilities to store livestock manure • Limit period to apply livestock manure	• Reduction of nutrient surplus • Decrease of nitrate content in the soil	• Reduction of nitrate and phosphorous concentrations in water • Improvements of biodiversity
	Measures aimed at control of pesticides	
• Reduce application of pesticides • Change in technologies and practices used to control pests	• Decrease of pesticides content in the percolation water	• Reduction of pesticide concentrations in water • Improvements in biodiversity
	Measures aimed at control of water demand	
• Reduction in the amount of water abstracted	• Recovery of groundwater tables	• Recovery of wetlands

This report distinguishes between immediate, intermediate and ultimate (long-term) responses to changes in agricultural policies, distinguishing between measures that control the use of fertilisers, pesticides and demand for water (Table 1). Immediate impacts relate mainly to changes in farming practices, whereas ultimate impacts relate to improvements in ecosystems and water-quality. Empirical work to assess the environmental effects of agricultural policies tends to focus on immediate effects. Yet, as already noted, the response time between a change in farming practice and an improvement in environmental quality may range from a couple of years (*e.g.* pesticides in surface water) to several decades (*e.g.* nitrates in groundwater).

The difficulty of assessing the impact of agri-environmental measures on environmental pollution and environmental quality is specifically acknowledged in the EU's Regulation 2078/92. This scheme, which strives to ensure that agricultural production methods are compatible with requirements for environmental protection and proper management of the countryside, has had immediate impacts on farming practices. But the available evidence regarding changes in water quality and ecosystems is still limited (Buller *et al.*, 2000).

Reducing harmful effects by reforming agricultural policy

This section reviews investigations into the impacts of agricultural policy reform, organised by environmental issue: emissions of greenhouse gases (GHGs), nitrate pollution of water and soils, water pollution from the use of pesticides, and abstraction of water for agricultural use. An investigation of policy measures reducing harmful environmental effects will be followed by an investigation of some empirical work regarding agricultural policy measures that support a variety of flora and fauna, as well as scenic landscapes. The review of measures that enhance beneficial effects and incorporate positive externalities of agriculture will largely be based on a review of literature from the EU.

Greenhouse-gas emissions

Climate change, linked to the enhanced greenhouse effect, is a global concern. Three gases — nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) — together account for around 90% of this effect. These gases are also the ones most closely associated with agriculture. Measured in terms of carbon-dioxide equivalent emissions ("CO_2 equivalents"), agriculture in OECD countries contributes about 7% to global anthropogenic GHG emissions. It contributes less than 5% to total emissions of carbon dioxide, but approximately 40% to emissions of N_2O and CH_4 (OECD, 1999). Methane is emitted mainly through the digestive processes of ruminant animals and from the anaerobic decomposition of livestock manure; smaller amounts are produced during the cultivation of paddy-rice. Emissions of nitrous oxide from agricultural practices arise primarily from the application of (in)organic fertilisers to farmland.

The OECD (1999*a*) identifies a broad set of agricultural practices that can contribute to the reduction of greenhouse-gas emissions. Those factors pertaining to animal husbandry and manure management include:

- *The digestibility of feed*. Methane is one of the products in case of incomplete digestion and some efficiency gains might be achieved here. However, ruminants require a minimum level of fibre. Otherwise, an acidosis might stop the bacterial digestion in the rumen.

- *The efficiency with which feed is converted to meat or milk.* The higher milk production per cow, the greater is the proportion of nutrients that go into producing meat or milk, and the smaller is the proportion required for metabolic maintenance. However, dairy cattle has achieved a high degree of efficiency in a large part of the developed world, with a negative feed input — milk output ratio for the first months of the lactation period. For physiological reasons (size and capacity of the digestive apparatus) they cannot take up enough dry matter for the genetically programmed production they release in the first months.

- *Using low-emission manure management practices*. Producers may not adequately air manure systems with daily use of triturator and airing pump system. Consequently, when applying manure the upper layer of the soil could be damaged by the applied pH resulting from anaerobic fermentation, with high greenhouse gas emissions. Although the majority of farmers still spread manure in a relatively dry form, which practice produces limited emissions of GHGs, more and more are applying it as a slurry. The latter practice yields higher emission rates, favoured by the anaerobic conditions under which soil microbes break down the manure.

- *Reducing livestock density*. The introduction of the milk quota system in the EU, for example, led to a reduction in dairy cattle. As production per cow increases over time, fewer cows are needed to fill the quota and so EU dairy herds decreased over the years. Furthermore, the quota has been reduced several times since it was introduced. In addition, agri-environmental programmes offer compensation to farmers who have introduced measures for entering extensive grassland farming. Both measures contribute to extensification of production systems and reduce local (but not necessarily global) greenhouse-gas emissions.

Rice cultivation accounts for a much smaller share of methane emissions than was believed in the past. GHG emissions associated with rice cultivation can be managed by:

- *Avoiding the shallow flooding of rice fields.* Water depths in excess of one meter are estimated to reduce methane emissions by 40%. This, however, may in reality be hardly practised in paddy fields. In Japan, for example, water depths in rice paddy fields normally are 20-30 cm.

- *Increasing field draining and reduce flooding periods.* Intermittent flooding might be one of the most important ways to limit methane emissions associated with rice production.

- *Reducing rice-straw residue*. Ploughing straw residue into the soil following harvest is thought to reduce methane emissions. However, the critical equilibrium of microorganisms in the rice paddy soil might well interfere here, because of the additional aeration since rice straw rots relatively slowly.

- *Reduce the use of organic fertiliser in flooded rice fields.*

- *Increasing the use of low-methane-emission rice varieties.*

- *Restricting the burning of crop residue.* In addition to CH_4, CO_2 and N_2O are also emitted through burning.

Crop practices that control N_2O emissions generally relate to the application of nitrogen. For example, farmers plant nitrogen-fixing crops or increase the uptake of nitrogen by crops. And any improvement in the efficiency of nitrogen fertilisers would not only diminish emissions of nitrous oxide but also reduce nitrate leaching. Management practices that protect humic soils are important to avoid oxidation of organic matter.

Storey and McKenzie-Hedger (1997) examine key interactions between agricultural policies and emissions of GHGs. A decline in livestock numbers has been one reason for reduced methane emissions in several OECD countries. This was due, at least in part, to reforms of agricultural policy in the EU (*e.g.* milk quota regime) and in New Zealand. Programmes to control crop production, such as the set-aside and other land-conversion schemes in the EU, Canada and the USA, may have helped to moderate emissions of N_2O.

Empirical knowledge on options to reduce greenhouse gas emissions from agricultural sources remains partial, although there is some evidence on possible interactions between those emissions and other environmental quality indicators. An increase of milk production per animal and/or a reduction of the number of cattle may reduce methane emissions, which however could have harmful effects in terms of land management practices and water quality as well. The potential for reducing greenhouse-gas emissions was estimated using the Canadian Economic and Emissions Model for Agriculture (CEEMA), containing the Canadian Regional Agricultural Model (CRAM) and a Greenhouse Gas Emissions Model (GHGEM). The assessment for Canada concludes that, considered in isolation, reducing greenhouse-gas emissions would be linked to shifts in production practices. However, most changes to farming practices (Table 2) that mitigate GHGs in agriculture yield ancillary benefits, such as reduced nitrate leaching, odour, wind and water erosion.

Table 2. Changes in farm-management practices that can contribute to mitigating greenhouse gases

Practice	Affected gas		
	CO_2	N_2O	CH_4
Improving the management of grasslands and grazing	•	•	•
Changing cultivation techniques	•	•	
Improving the management of soil nutrients		•	
Improving the management and feeding of livestock	•	•	•
Using crop residues in industrial products	•		
Improving the management of water in agriculture	•	•	•
Improving the management of manure	•	•	•
Sequestering carbon through shelterbelts and trees in farmyards	•	•	
Producing biofuels	•	•	

Source: Agriculture and Agri-Food Canada (2000).

Input use in agriculture

Nutrients, pesticides, energy and water are the main intermediate inputs used in agriculture. These inputs have different effects on year-to-year variability in yields. Fertilising, for example, increases the probability of high yields, but rainfall needs to be adequate to achieve them. On the other hand, fertilising may also increase the probability of low yields when rainfall is inadequate. Increasing fertiliser use therefore increases both the mean yield and the variability of yields. In contrast, insecticides cut production losses in the event of an infestation, but do not affect yields if no infestation occurs. Fertilisers may therefore the considered as a risk-*increasing* input and pesticides (or at least insecticides) as risk-*reducing* (Karagiannis and Xepapadeas, 1999). Irrigation provides insurance against dry weather conditions.

Horowitz and Lichtenberg (1994), in exploring the conditions under which pesticides alter profit risk, observe that if pesticides are indeed risk-reducing, then that would imply that risk-averse farmers would use them more readily than risk-neutral farmers. In fact, several econometric analyses suggest that pesticides are risk-*increasing*, as pesticide use increases the variability of yields. Altogether, the existing literature on risk-increasing or -reducing responses to changes in market support, management practices and input use seems to be controversial. Farm responses also take into account biophysical conditions, the available technologies and market conditions.

Abler and Shortle (1992) examined the interaction between farm support programmes and environmental policies for agriculture. Based on a three-region (USA, EU, rest-of-the-world) partial-equilibrium model they argue that a bilateral elimination of farm programmes — without restrictions placed on the use of agrochemicals — would induce a shift in production and chemical use from the EC to the USA. The authors propose to have the elimination of farm support programmes to be accompanied by bilateral restrictions on use of agrochemicals. A restriction on the use of agricultural chemicals would be less attractive in the USA, if accompanied by a bilateral elimination of farm programmes. The analysis has a high level of aggregation, but provides evidence on the linkages between international trade, environmental policies and farm support programmes.

Nitrate pollution of water and soils

Nitrate pollution from agricultural sources is a high-priority issue for several OECD countries. The leaching of nitrates is very much related to the intensity of agriculture, as affected by climate, geography and on-farm water management (Brouwer *et al.*, 2000). Intensification of agricultural production tends to increase the use of chemical fertilisers. Intensification and specialisation of agriculture involves the development of capital-intensive and geographically specialised farming, which is mainly observed in regions where agriculture is most productive. The development and adoption of new technologies are factors of importance to the spatial location of agricultural holdings.

Using panel data for 22 countries (including Australia, Canada, the EU, Japan, New Zealand, South Korea, and the USA) Lewandrowski *et al.* (1997) found that agricultural assistance (measured in PSE (Producer Subsidy Equivalent) has a positive and significant effect on fertiliser use per hectare farmed. Statistical evidence showing an effect of price support on land use was weak, however, to the extent it was observed at all. It reduces the demand for agricultural land in developed countries, but has a neutral effect in low-income countries. Their analysis also indicates that when farm support is reduced the environmental effects are not necessarily the reverse of those that occur when support is increased. That is essentially because farmland becomes comparatively less expensive as a factor of production and more of it gets used. They also argue that a global reduction in support levels, such as might be negotiated under a multilateral agreement, would allow international prices of commodities to rise, which would spur production in countries previously exposed to world market prices. Finally, the authors underscore the need for environmental protection measures that adequately integrate external effects into farming practices.

The above analysis indicates the importance of considering fixed assets such as land and quotas when assessing the environmental impacts of reforming farm-support programmes. Land prices interact with agricultural policies and the use of inputs. A comparison of organic and conventional dairy production in the Netherlands, for example, indicates that existing direct-income support and compensatory payments are more beneficial for organic milk production than conventional milk production. The price advantage enjoyed by organic milk production would allow its market share to increase. An increase in organic milk production would further increase pressure on land availability, resulting in a potential increase in the prices of land and of rough feed. In turn, this would reduce the relative advantage of organic production over conventional production. The authors also argue that a reduction in the amount of milk quota would contribute to the further extensification of conventional milk production, because it would cause a drop in land prices and the increase in milk-quota costs. Producers of organic milk might benefit from such a change in market intervention because they are less dependent on milk quotas — relative to the available land resource — than are conventional producers and would be affected less.

Plantinga (1996) also examines the interaction between agricultural policies, land use and environmental quality in the context of milk price-support measures. He argues that higher prices provide incentives for farmers to keep lower-quality land in production. These lands tend to be susceptible to soil erosion. The study demonstrates that a reduction in support prices would reduce such incentives, thereby reducing soil erosion and enhancing water quality. Depending on how the land is managed, this may reduce soil erosion and its associated degradation on water-quality. The available options to change land-management practices in agriculture, or to shift farmland into non-agriculture uses, therefore are important considerations when undertaking policy reform. McRae *et al.* (2000) also emphasise the linkages between agricultural support measures and the utilisation of land — *e.g.*, support that is directly related to crop production, which provides incentives to farm marginal land or wetlands. The arguments presented here are seemingly based on the presumption that marginal land from an economic perspective is equal to marginal land from an environmental perspective. However, the picture across OECD countries remains diverse. An appropriate land management is required to maintain the biodiversity in a range of low intensity farming systems with highly diverse habitat types, including among others, semi-natural grasslands, important areas for breeding and migratory birds, and areas with many 'natural' features like hedgerows, small woodlands and ponds. Marginalisation might occur on land where agriculture is unable to compete effectively with more productive regions. Since market and price support measures are not oriented towards these farming systems, targeted payments would support attempts to counteract marginalisation (Dax and Hellegers, 2000). Payments under the Less Favoured Area scheme have been introduced in the EU during the 1970s to prevent land abandonment, preserve the farming population in those areas and conserve the countryside. By 1999, this scheme was combined with the agri-environment regulation to create a new instrument with rural development measures supporting integrated rural development across the EU.

In regard of the above, some part of the terraced rice paddy fields in hilly and mountainous regions in Japan are marginal areas of land from an economic perspective, but not marginal areas from an environmental perspective. Paddy fields in these areas have several environmental benefits, such as their flood prevention function, as well as prevention of landslides and soil erosion, and the maintenance of biodiversity (OECD, 1997). Similarly, maintenance of agricultural production in economically marginal areas in temperate and Mediterranean climates could sustain landscapes, avoid erosion, maintain natural environments and prevent the occurrence of natural disasters (*e.g.* fires and avalanches).

In the EU, pollution-control measures targeted at nitrates are linked to the reform of the arable crops regime. A reduction of the intervention prices of cereals lowers the cost of reducing the protein content of compound feed, enabling intensive livestock production units (mainly pigs and poultry) to apply nutritional management measures to reduce nitrogen output without compromising profitability throughout the production chain. However, world-market prices of soybean meal have been shown to influence the effectiveness of a reduction in cereal prices on nitrogen emissions (Brouwer *et al.*, 1999). The CAP reform implemented during 1992 and 1995 had already significantly reduced the cost of lowering the protein content of compound feed. But while it facilitated the control of nitrate pollution, it did not achieve it. A reduction of intervention prices — adopted in the context of Agenda 2000 — are expected to further provide a wider application of low-protein feeds. The protein levels adopted in diets remains driven largely by the highly volatile world-market price of protein sources like soybeans. The EU has a very limited influence on such prices because of its small share in the global consumption of soybean meal. The ability for nitrogen pollution control through reforming the arable crops regime in the EU therefore remains highly dependent on world market conditions.

Several assessments are available in the EU regarding the impacts of policy measures for nutrient surpluses. The impacts of agri-environmental policy measures to control nitrogen pollution are examined by Kleinhanss *et al.* (1997). Their analysis used a neo-classical, regionalised simulation model and a nutrient balance model to simulate the impact of alternative policy measures on agricultural production and nutrient surpluses. Their results indicated that the 1992 reform of the CAP affected income and agricultural production more than it did nitrogen surpluses. A significant reduction in fertiliser consumption (-14% on average) would be achieved with a reduction of intervention prices, but nitrogen surpluses would only be reduced by 1% to 6% because the supply and use of livestock manure was not significantly changed. (In contrast with crops, there was little change in the number of livestock and hence manure production.) Economic instruments (*e.g.* a tax on fertiliser consumption) show a high response in regions with excess amounts of nutrients that need to be disposed of. In conclusion, they expect "in the long term, the CAP reform may induce a better distribution of animal manure because of the density restrictions for beef premiums, and lower prices of home-grown feed will reduce the regional concentrations of livestock. The models do not allow the assessment of these structural changes" (p. 144).

In addition to performing an assessment at the *sectoral* level, Kleinhanss (2000) also quantified the economic and environmental impacts at *farm* level of agricultural policy reform. Using a linear-programming model to assess the impact of the Agenda 2000 reforms (proposed in 1999) on farms in Germany he foresees an overall decline in nitrogen surplus, but mainly on crop-producing farms that already produce little surplus. Because the analysis does not take into account agri-environmental measures intoduced under Agenda 2000, however, it may underestimate the reform's beneficial effects on the environment.

Detailed investigations at the national level have also been undertaken. For example, a sector model for Denmark was developed to assess the impact of policy changes on production, prices and emissions of nutrients (Jensen, 1996). Helming (1998) used a partial-equilibrium, comparative-static, mathematical programming model to assess the effects of taxes on nitrogen input use and surpluses on farms in the Netherlands. His results showed that:

- A tax on nitrogen used in feed concentrates reduces the nutrient, but increases the (shadow) price for animal manure.
- A tax on nitrogen used in mineral fertilisers decreases their use, but increases the amount of nitrogen used in feed concentrates. The demand for fertilisers is very inelastic under a system of relatively low fertiliser taxes, because of the limited incentives to change fertiliser practices under the given market conditions and high prices for feed concentrates.
- A tax on nitrogen used in feed concentrates and mineral fertilisers increases the price for animal manure.
- A tax on nitrogen surplus transfers money from the producers to the users of animal manure.

In sum, a tax on nitrogen surplus costs farmers less per unit of nitrogen surplus reduced and is therefore more cost-effective than other approaches to reduce nitrogen surpluses.

Elasticities of demand for fertilisers are low because of the restricted number of substitution possibilities. The price-elasticity of the demand for chemical fertiliser expresses the extent to which demand responds to price changes. If demand is inelastic, and price elasticity very small, high levels of tax would be required to significantly reduce its use. The short-run price-elasticity of chemical fertiliser demand ranges from -0.2 to -0.3, meaning that a 10% increase in the price of fertiliser would reduce its consumption in the short run by between 2% and 3%. A stronger response (corresponding to a long-run price elasticity from -0.5 to -0.6) can be expected in the long run, as opportunities to change inputs (such as capital equipment and labour) and outputs can more easily be exploited. A tax on fertilisers would also increase the opportunity costs of livestock manure and would improve the efficiency of using manure for crop production. Several estimates of price elasticities are based on linear-programming models, and restrictions on production could limit substitution among crops and underestimating elasticities.

The impact of policy instruments (Box 1) to control non-point sources of water pollution in the USA is examined for the USA by Ribaudo *et al.* (1999). Because of the large number of farming systems, and local differences in farming practices and hydrogeographic conditions, the authors advocate addressing the problem with a mixed set of policy instruments that are regionally targeted. Also, they show that there is considerable uncertainty about the *magnitude* of the impact on water quality of changes in farming practices, but not the *direction*.

Box 1. Soil and water-conservation programmes in the United States

The Conservation Reserve Program (CRP), the Environmental Quality Incentives Program (EQIP) and the Water Quality Incentives Program (WQIP) are voluntary-compensatory schemes to control soil erosion and other environmental purposes. WQIP was consolidated into EQIP by the 1996 Farm Act. EQIP includes payments for the adoption of farm management practices that reduce environmental and resource problems, such as conservation tillage. It (and the earlier WQIP) is designed to offset private losses incurred in adopting the practice, any increased risk during its first years of implementation and any other constraints on the short-term. In addition to support for the implementation of land-management practices, cost-sharing grants are available for the construction of structures (*e.g.* for storing animal waste). Large structures used for confined animal-feeding operations are not eligible for cost-sharing grants, however.

Pesticide contamination of water

Problems related to the use of pesticides tend to cause concern in a large number of OECD countries, among others in the EU and the USA. Aerial spraying of pesticides and the contamination of surface water is an issue in several countries, as are potential threats to non-target species and to human health (especially from residues in drinking water).

Falconer and Oskam (2000) assessed the impacts on pesticide use in the EU of the 1992 reform of the arable crops regime. Their results suggest that the combined effects of a reduction in the intervention prices of cereals and the introduction of set-aside requirements reduced the use of pesticides for growing cereals by almost 10%, and the use of pesticides overall by 3%. The effects of the Agenda 2000 measures on pesticides use are estimated to be very modest. One reason for the limited reduction of pesticide use under Agenda 2000 is that price decreases and set-aside schemes work in opposite directions. In addition to the CAP, a range of other factors have also contributed to the reduction of pesticide use in the EU, including technological development with chemical substitution, and a smaller amount of active ingredients per hectare now suffices to treat plants compared to what was used in the past. Other factors driving usage of pesticides are climatic and weather conditions that affect the occurrence of pests, changes in environmental legislation and autonomous trends in agricultural areas to go down. Pesticide use has also been reduced through acreage restrictions in the USA. There, the CRP reduced crop acreage and, hence, pesticide use on land that might otherwise have remained in production. However, agricultural support programmes can provide incentives to increase pesticide use on the land that is not set aside (U.S. Department of Agriculture, 1997).

Pesticide use in aggregate tends to be unresponsive to small price changes, though demand for individual products can be very responsive to even small price changes. Rayment *et al.* (1998) indicate that most empirical studies offer evidence on price elasticities regarding demand for pesticides of between -0.2 and -0.5. Hoevenagel *et al.* (1999) reviewed the literature on price elasticities, and offer a range of price elasticities of demand for pesticides, with the highest applying to specific categories of pesticides, such as herbicides (between -0.7 and -0.9) and insecticides (between -0.3 and -0.8). The price elasticity of herbicides relates to the options available to control pests and diseases without treatment. Mechanical weed control, for example, can often substitute for herbicides. The price elasticity of demand for pesticides differs for specific crops. For cereals, for example, it could be as high as -1.1 (Hoevenagel *et al.*, 1999).

The U.S. Department of Agriculture (1997) has recommended several criteria for the successful implementation of voluntary water-quality programmes. The criteria are general and not specific to pesticides:

- Get farmers to recognise their contribution to severe local pollution problems.
- Recommend practices that offer economic benefits.
- Target watersheds where agriculture is the primary source of water quality impairment.
- Design cost-share programmes to be flexible; these are more efficient than ones that offer fixed cost-shares or that specify what practices must be followed.
- Conduct local research on the performance of recommended practices (this can improve adoption).
- Interact with the main stakeholders.
- Ensure that monitoring and project evaluation are adequate and contribute to improving the programmes.

The report also stresses that improvements in water quality from non-point-source pollution often take years to respond to changes in farm-management practices.

Water use for agricultural purposes

Irrigation accounts for a major share of water use in many OECD countries. Concern over excessive withdrawals from groundwater aquifers is widespread because many aquifers are being depleted at rates far exceeding their rates of natural replenishment, and because in some areas groundwater pumping (combined with drainage) has changed the character of wetland ecosystems. Surface-fed irrigation presents a different set of problems. In many OECD countries public bodies manage large irrigation works, especially those involving surface-water reservoirs, and the price of water supplied to farmers rarely reflects total costs of delivery, much less external costs (OECD, 2001*b*). The U.S. Department of Agriculture (1997), for example, indicates that prices for water from surface-water-based projects tend to be based on the costs of impounding and delivering the water to end-users. Frequently, a significant share of the capital costs of dams and related infrastructure used in irrigation projects are absorbed by taxpayers or charged to other users, such as electricity rate-payers. Moreover, where there is competition among different classes of users, prices often do not reflect relative scarcity. In the EU, the Water Framework Directive aims to promote sustainable water use based on a long-term protection of available water resources and to ensure the progressive reduction of pollution of groundwater and prevent its further pollution. This Directive encourages full cost recovery including environmental, social and economic conditions as well as geographical and climatic conditions.

The OECD (1999*b*) summarises various complexities related to the reform of water subsidies that are of particular relevance to OECD countries. First, water subsidies are usually incorporated into the value of irrigated farmland and an increase in the price of irrigation water may consequently reduce the value of that land. Also, most of the on-farm capital used for irrigation is sunk during construction: it cannot be easily transferred elsewhere. The result is that a farmer who bought her land prior to a major water-price increase may end up incurring large capital losses if she sells it following a price rise.

Varela-Ortega *et al.* (2001) examine links between agricultural policy and agricultural demand for water. During the period before Spain entered the European Community (in 1986), irrigation was promoted through national programmes; it continued to be promoted following accession. Nevertheless, the country achieved a 20% reduction in irrigation water use between the late 1980s and 1992, mainly owing to national rules that limited the extraction of water. During the early 1990s, for example, agri-environmental programmes were established in the region of Castile-la Mancha (Tablas de Daimiel); these aimed to restore wetlands by reducing water extractions from shallow aquifers.

Massaruto (2000) discusses the different effects of the CAP on irrigation in EU Member States. In Spain, the area sown to cereals declined after the 1992 reform of the CAP, but the same policies may have provided incentives to increase the area of irrigated maize in France. Rainelli and Vermersch (1999), compared yields of maize on irrigated land in France with yields on non-irrigated land. They found that the output from commodities and investment in equipment tend to balance with yield increases from irrigation. A side effect of this would be increased nutrient losses. Reducing the amount of water and fertiliser applied may reduce nitrate leaching while maintaining yield. In addition, they argue that yield variability tends to be reduced on irrigated land, which is an argument in favour of irrigation to counteract risks of crop losses due to droughts. Some empirical results relating to cereal farms in France (between 1987 and 1995) emerge from their analysis:

- Productivity gains from irrigation tend to increase over time. The maize premia in France are based on average yields for a region, regardless of whether irrigation has been used. However, the reference yield distinguishes between irrigated and non-irrigated land. Direct payments that are based on acreage for irrigated crops imply a production right, which generates a rent that is captured in land transactions in a way similar to the production rights associated with milk quota. (During the first half of the 1990s some regions of France with a high share of irrigation experienced increases in land prices that exceed national averages.)

- Use of agrochemicals per kilogramme of harvested product is lower on irrigated land than on non-irrigated land.

- Labour input per kilogramme of harvested product is rather similar for irrigated and non-irrigated land. Labour efficiency tends to increase over time, with the rise steeper for irrigated crops. This suggests the presence of economies of scale.

LaFrance (1992) explores the interaction between commodity prices and soil degradation. He argues that subsidisation of water leads to an increasing exploitation of soils and a decline of soil fertility in the long run. Commodity price support does not necessarily improve or degrade soils, although empirical evidence in the USA suggests that price support tends to further degrade the physical structure of soils. LaFrance favours conservation payments rather than price support to promote land conservation. This issue is considered in the following section.

Enhancing beneficial effects by reforming agricultural policy

Loss of wildlife habitat and landscapes are issues of significant concern in many OECD countries and several investigations have been made to examine the impact of action to arrest these trends. [The estimated value of paddy fields in Japan, for example, has been assessed according to substitutive cost method, at a level of 4,600 billion Yen. The main functions of such fields are to prevent flooding and mitigate the damage caused by floods, and to preserve rural landscapes and recreational amenities through visits by urban inhabitants (http://www.maff.go.jp/soshiki/kambou/Environment/env8.html). A large number of valuation methods have been applied, and almost 60 contingent valuation methods are available to estimate landscape values in Japan.

In the USA, farm-programme payments depended on crop base acreage until the mid-1980s. Such programmes provided a strong incentive to transform wetlands into cropland. The CRP, introduced in 1985, took land out of production and was initially targeted at highly erodible or environmentally sensitive cropland. In the 1990s it was broadened to include environmental objectives, such as the preservation of wildlife habitat. The programme compensates farmers for the conservation of certain wetlands or the restoration of converted wetlands. Heimlich *et al*. (1998) reviewed wetland policies related to agriculture in the USA and simulated the economic effects of converting wetlands to farmland, after equilibrium adjustment to the short-run wetland conversion (with baseline prices and production and conversion costs). The long term conversion of wetland relative to the level with baseline conditions is input to the U.S. Agriculture Sector Mathematical Programming Model (USMP). The demand and supply responses in the model are rather inelastic because farm incomes decline with an expansion of production and a fall in crop prices. The model specifies a range of crop rotations and production practices for highly erodible and less-erodible soils. Farmers may respond to measures to control soil erosion by switching to crops that are cause less erosion. Scenarios on low wetland conversion and high wetland conversion both estimate an increase in cropland acreage.

Because current and expected commodity prices provide sufficient incentive to drain and plough up wetlands, the U.S. Government offers payments to discourage such conversion. These payments obviously need to be sufficiently high to offset the income foregone by not converting wetland. The effectiveness of such programmes therefore largely depends on market conditions: high market prices discourage farmers from participating in wetland programmes unless compensatory payments adjust accordingly.

Feather *et al.* (1999) examine the use of non-market valuation models in designing conservation programmes such as the USA’s Conservation Reserve Program. Their analysis takes into account the benefits to society of outdoor recreation of targeted conservation programmes. More recently, Claassen *et al.* (2001) have evaluated agri-environmental programmes implemented in the USA over the past 20 years, programmes that have played a significant role in reducing soil erosion, protecting and restoring wetlands and creating wildlife habitat. They identify several considerations for improving their effectiveness:

- Payments should be directed to those areas where the environmental benefits are greatest relative to costs. Such environmental targeting has increased environmental benefits in the CRP, for example.

- Programmes should allow farmers to search for least-cost approaches to meeting environmental targets rather than imposing generic conditions that apply to all farmers. The wide range of biophysical and climatic conditions and farm management practices that prevail across the USA implies that measures would need to reflect that variation.

- Different programmes should be co-ordinated, so as to ensure that they do not duplicate or offset each other.

Wu *et al.* (2000) examine ways to balance environmental benefits with public expenditure. In addition to cost targeting (which aims to purchase a specific resource, such as a hectare of land, at minimum cost), benefit targeting (*e.g.*, purchasing land that provide the largest benefits per hectare) and benefit-cost targeting (obtaining the greatest benefits per dollar or euro spent) are adopted by the authors as well. Initially the CRP targeted benefits and focused primarily on the control of soil erosion. More recently, the programme has moved towards benefit-cost targeting.

The U.S. Department of Agriculture (1997) has looked at both the social benefits and costs of the CRP. The main components of the social benefits include increased net farm income, the net present value of future timber grown on land enrolled in the programme, improved surface water quality and enhanced small-game hunting (the largest component). Social costs relate mainly to the effect of the programme on the price of food, and costs of establishing vegetative cover. On balance, the USDA estimates that social benefits exceed social costs. The study also indicates that cost-effectiveness might be improved — and chances of overpayment lessened — by better targeting bids and considering the productivity of the land offered before accepting them.

Several investigations have been undertaken that attempt to assess the environmental effects of implementing the agri-environmental measures under the EU's Regulation 2078/92. Andersen *et al.* (1999), for example, investigated a selected number of agri-environmental indicators and made a distinction between changes in land use practices and changes in management practices. Both reflect the need to maintain land use management of agri-environmental programmes. Agricultural practices of land use are supported by the measures, as well as specific management practices (*e.g.* fertiliser and pesticide application and management of grassland). The evaluation presented of a set of agri-environmental programmes indicates that the environmental gains of the programmes are not always clear. However, it seems safe to conclude that they have shifted the attitudes and behaviours of farmers — a first and necessary step for the achievement of longer-term benefits for the environment and landscape.

A survey was undertaken in eight EU Member States regarding farmers' uptake to agri-environment programmes. In addition to the level of compensation - other factors include education of farmers and their knowledge and attitude on the generation of multiple output from farming, including food, biodiversity and landscape maintenance (Van Huylenbroeck and Whitby, 1999).

Transaction costs are the means used to achieve policy goals and their expenses need to be complemented with the achievements reached. Such cost-benefit assessments that balance marginal benefits with marginal costs are complex to untangle from other trends. Limited information is available on the transaction costs involved with targeted programmes relative to generic market and price support measures. Falconer and Whitby (1999) assess the costs involved with implementation and administration of agri-environmental programmes. The administrative costs relate to the information gathering (*e.g.* surveying and designation of land), contracting (*e.g.* promotion of scheme, negotiation with the farmers and administration of the contract) and policing (for monitoring, enforcement and evaluation). An investigation that was undertaken by these authors in about half of the EU Member States indicates that the administrative costs varies from low percentages (less than 5%) to 100% of the scheme expenditure. They also estimated the accumulated transaction costs of the implementation of agri-environmental policy in the EU during the period 1992 to 1996 to be some 1-2 billion euro, relative to the direct payments of almost 7 billion euro.

Direct payments are provided to Swiss farmers providing Required Environmental Services (RES). Such payments are subject to environmental conditions regarding stock-taking, provisions on fertiliser budgets, regular crop rotation, appropriate soil protection and targeted selection and use of plant chemicals (Hofer, 2001). Improvements are recorded in terms of the proportion of land receiving environmental compensation, proportion of land under organic farming, reduction of nutrient surpluses and in the amounts of pesticides used.

Concluding remarks

This paper offers a review of a selection of the published empirical work carried out in recent years to assess the environmental impacts of agricultural policy. The following conclusions are intended to suggest possible additional quantitative work in this area:

- In the context of empirical work, the environmental effects of reforming agricultural policies remain difficult to be isolated from the many other factors that interact as well. Market conditions (*e.g.* price fluctuations on the world market, fluctuating exchange rates between currencies, world supplies and demand for food) and other types of policy intervention (*e.g.* environmental protection measures) also need to be considered.

- Farmers do respond to changes in income and prices. It is widely agreed that market interventions and price-support measures have encouraged greater agrochemical use than would otherwise have been the case. Also, they may have promoted farming on marginally productive land. Reducing the overall levels of agricultural price support measures may therefore lead to environmental improvements, either by encouraging a more efficient use of inputs or a shift to more-extensive production systems. However, this is not always the case, especially in regions where agriculture has positive externalities to environmental and recreational goods and services, landscape and biodiversity, and agricultural policies can enhance them.

- There is limited evidence of environmentally effects of reducing price support measures on agrochemical use, because of the price elasticity of demand for agrochemicals tends to be low in the short run.

- Agricultural support tends to be capitalised into land values, with intensification and specialisation of production a result. This encourages the development of capital-intensive and geographically specialised farming, especially on farms with a high productivity potential or that are close to urban centres. A reduction in price support may lead to the cessation of farming on economically marginal land. The environmental perspective of farming on such marginal land is dependent, among others, on land management practices and vulnerability of soils.

- The agricultural land market plays a major role in the interactions between agricultural support programmes and the environment. Land prices reflect available opportunities, including agriculture, and tend to rise with increasing scarcity. Demand for non-environmental amenities (*e.g.*, nature parks) and incentives to leave more private land in its natural state (*e.g.*, buffer strips and small ponds) can contribute to that scarcity.

- Improved knowledge of the site-specific implications of policy measures on the environment have helped improve the design of programmes by better tailoring them to reflect local differences in economic structures and natural conditions.

- Farm-management practices need to be considered when identifying the interactions between agricultural policy and the environment. A wide range of management practices and different environmental profiles might emerge from reforms of agricultural policy. Farmers can be expected to respond to changes in farm-support programmes in many ways. Gaining a better understanding of the costs and benefits (to both the farmers themselves and to society at large) of different farm-management practices would improve the ability to predict the effects of policy changes on farmers' decisions and hence on the environment. Transparency of public policy however needs to be considered where flexibility of measures offers a range of options to farmers in meeting public objectives.

- Time and scale are crucial dimensions when considering the environmental effects of agricultural policies. A regional perspective is necessary when attempting to assess the environmental effects of agricultural policy, except perhaps in small, homogeneous countries. Environmental conditions vary widely, as do farm-management practices. An appreciation of the time it takes to see tangible improvements in the environment following a change in farming practices is also important.

REFERENCES

Abler, D.G. and J.S. Shortle (1992), “Environmental and farm commodity policy linkages in the US and the EC”, *European Review of Agricultural Economics*, 19, pp. 197-217.

Agriculture and Agri-Food Canada (2000), *Reducing Greenhouse Gas Emissions from Canadian Agriculture: Options Report*, Agriculture and Agrifood Canada, Ottawa.

Andersen, E., J. Primdahl, J. Oñate, J. Aguirre, C. Cummings, B. Peco, J. Schramek, B. Klingler and K. Knickel (1999), “Analysing the environmental effects of Regulation 2078/92”, in J. Schramek, D. Biehl, H. Buller and G. Wilson (eds.), *Implementation and Effectiveness of Agri-Environmental Schemes Established under Regulation 2078/92*, Volume II, pp. 137-284.

Brouwer, F.M., P. Hellegers, M. Hoogeveen and H. Luesink (1999), *Managing nitrogen pollution from intensive livestock production in the EU: Economic and environmental benefits of reducing nitrogen pollution by nutritional management in relation to the changing CAP regime and the Nitrates Directive*, Rapport 2.99.04, Agricultural Economics Research Institute (LEI), The Hague.

Brouwer, F., D. Baldock, C. Carpentier, J. Dwyer, D. Ervin, G. Fox, A. Meister and R. Stringer (2000), *Comparison of environmental and health-related standards influencing the relative competitiveness of EU agriculture vis-à-vis main competitors in the world market*, Report 5.00.07, Agricultural Economics Research Institute (LEI), The Hague.

Brouwer, F. and J. Helming (2000), “The effects of support measures on the profitability of organic farming relative to conventional farming: A case study for the Netherlands”, Document No. ENV/EPOC/GEEI(99)4/FINAL, OECD, Paris.

Brouwer, F. and M. Hoogeveen (2000), “The arable crops regime and nitrogen pollution control”, in F. Brouwer and P. Lowe (eds.), *CAP Regimes and the European Countryside: Prospects for Integration between Agricultural, Regional and Environmental Policies*, CABI Publishing, Wallingford, UK, pp. 103-116.

Buller, H., G.A. Wilson and A. Höll, eds. (2000), *Agri-Environmental Policy in the European Union*, Perspectives on Europe, Contemporary Interdisciplinary Research, Ashgate, Basingstoke, UK.

Claassen, R., L. Hansen, M. Peters, V. Breneman, M. Weinberg, A. Cattaneo, P. Feather, D. Gadsby, D. Hellerstein, J. Hopkins, P. Johnston, M. Morehart and M. Smith (2001), *Agri-Environmental Policy at the Crossroads: Guideposts on a Changing Landscape*, Agricultural Economic Report Number 794, Economic Research Service, U.S. Department of Agriculture, Washington, DC.

Dax, T. and P. Hellgers (2000), "Policies for Less Favoured Areas", in F. Brouwer and P. Lowe (eds.), *CAP Regimes and the European Countryside: Prospects for Integration between Agricultural, Regional and Environmental Policies*, CABI Publishing, Wallingford, UK, pp. 179-197.

Falconer, K. and A. Oskam (2000), “The arable crops regime and the use of pesticides”, in F. Brouwer and P. Lowe (eds.), *CAP Regimes and the European Countryside: Prospects for Integration between Agricultural, Regional and Environmental Policies*, CABI Publishing, Wallingford, UK, pp. 87-102.

Falconer, K. and M. Whitby (1999), "The invisible costs of scheme implementation and administration", in G. van Huylenbroeck and M. Whitby (eds.), *Countryside Stewardship: Farmers, Policies and Markets*, Elsevier Science Ltd., Amsterdam, pp. 67-88.

Feather, P., D. Hellerstein and L. Hansen (1999), *Economic valuation of environmental benefits and the targeting of conservation programs*, Agricultural Economics Report Number 778, Economic Research Service, U.S. Department of Agriculture, Washington, DC.

Heimlich, R.E., K.D. Wiebe, R. Claassen, D. Gadsby and R.M. House (1998), *Wetlands and Agriculture: Private Interests and Public Benefits*, Agricultural Economic Report Number 765, Economic Research Service, U.S. Department of Agriculture, Washington, DC.

Helming, J. (1998), “Effects of nitrogen input and nitrogen surplus taxes in Dutch agriculture”, *Cahiers d’économie et sociologie rurales*, 49, pp. 6-31.

Hoevenagel, R., E. van Noort and R. de Kok (1999), “Study on a European Union wide regulatory framework for levies on pesticides”, EIM Small Business Research and Consultancy, Zoetermeer.

Hofer, E. (2001),"The Swiss approach to environmental protection", In: Environmental Standards in Agriculture: Proceedings of a pan-European Conference, 5-7 October 2000, Madrid, Spain. Proceedings edited by Jan-Erik Petersen and Karen Shaw. London, Institute for European Environmental Policy, pp. 57-60.

Horowitz, J.K. and E. Lichtenberg (1994), “Risk-reducing and risk-increasing effects of pesticides”, *Journal of Agricultural Economics*, 45 (1), pp. 82-89.

Jensen, J. (1996), “An applied econometric sector model for Danish agriculture (ESMERALDA)”, Rapport no. 90, Statens Jordbrugs- og Fiskeriøkonomiske Institut, Valby (Copenhagen).

Karagiannis, G. and A. Xepapadeas (1999), “Agricultural policies, environmental impacts and water use under uncertainty”, Paper presented at the 3rd workshop SAGA (FAIR-CT97-3673), Milan, 17-18 December 1999.

Kawagoe, T., K. Otsuka and Y. Hayami (1986), “Induced bias of technological change in agriculture: The United States and Japan, 1880-1980”, *Journal of Political Economy*, 94 (3), pp. 523-544.

Kleinhanss, W., H. Becker and K.-H. Schleef (1997), “Impacts of agri-environmental policy measures on nitrogen emissions from agriculture”, in E. Romstad, J. Simonsen and A. Vatn (eds.), *Controlling Mineral Emissions in European Agriculture*, CABI Publishing, Wallingford, UK, pp. 137-155.

Kleinhanss, W. (2000), “Economic and environmental impacts of Agenda 2000”, in F. Brouwer and P. Lowe (eds.), *CAP Regimes and the European countryside: Prospects for integration between Agricultural, Regional and Environmental Policies*, CABI Publishing, Wallingford, UK, pp. 305-319.

Krissoff, B., N. Ballenger, J. Dunmore and D. Gray (1996), *Exploring Linkages Among Agriculture, Trade, and the Environment: Issues for the Next Century*, Agricultural Economic Report Number 738, Economic Research Service, U.S. Department of Agriculture, Washington, DC.

LaFrance, J.T. (1992), “Do increased commodity prices lead to more or less soil degradation?”, *Australian Journal of Agricultural Economics*, 36 (1), pp. 57-82.

Lewandrowski, J., J. Tobey and Z. Cook (1997), “The interface between agricultural assistance and the environment: Chemical fertilizer consumption and area expansion”, *Land Economics*, 73 (3), pp. 404-427.

Massarutto, A., ed. (2000), “Water pricing, the Common Agricultural Policy and irrigation water use”, Report for DG Environment, Contract B4-3040/99/110316/MAR/B2, European Commission, Brussels.

McRae,T., C.A.S. Smith and L.J. Gregorich, eds. (2000), *Environmental Sustainability of Canadian Agriculture: Report of the Agri-Environmental Indicator Project*, Agriculture and Agrifood Canada, Ottawa. < http://www.agr.ca/policy/environment/eb/public_html/ebe/aei.html >

OECD (1997), "Environmental benefits from agriculture: Issues and policies. The Helsinki Seminar", Paris.

OECD (1999*a*), “Agricultural practices to reduce greenhouse gas emissions: overview and results of survey instrument”, Document No. COM/ENV/EPOC/AGR/CA(98)149/REV1, Paris.

OECD (1999*b*), “Agricultural water pricing in OECD countries”, Document No. ENV/EPOC/GEEI(98)11/ REV1, Paris.

OECD (2001*a*), *Agricultural Policies in OECD Countries: Monitoring and Evaluation 2001*, Paris.

OECD (2001*b*), *Environmental Indicators for Agriculture — Volume 3: Methods and Results*, Paris.

Plantinga, A.J. (1996), “The effect of agricultural policies on land use and environmental quality”, *American Journal of Agricultural Economics*, 78, November 1996, pp. 1082-1091.

Rainelli, P. and D. Vermersch (1999), “Environmental impacts of agricultural support: Cereal irrigation in France”, in *Improving the Environment through Reducing Subsidies: Part III: Case Studies*, OECD, Paris, pp. 47-78.

Rayment, M., H. Bartram and J. Curtoys (1998), “Pesticide taxes: a discussion paper”, Royal Society for the Protection of Birds, Sandy, UK.

Ribaudo, M.O., R.D. Horan and M.E. Smith (1999), *Economics of Water Quality Protection from Non-Point Sources: Theory and Practice*, Report Number 782, Economic Research Service, U.S. Department of Agriculture, Washington, DC.

Storey, M. and M. McKenzie-Hedger (1997), “Agricultural policy reform and climate-change mitigation in Organisation for Economic Co-operation and Development countries”, in W.N. Adger, D. Pettenella and M. Whitby (eds.), *Climate-Change Mitigation and European Land-Use Policies*, CAB International, Wallingford, UK, pp. 23–34.

Tangermann, S. and A. Buckwell (1999), “The purpose and methodology of evaluation in regard to EU agricultural expenditure”, Working Paper BUDG-102 EN, Directorate-General for Research, European Parliament, Luxembourg.

U.S. Department of Agriculture (1997), *Agricultural Resources and Environmental Indicators, 1996-97*, Agricultural Handbook No. 712, Economic Research Service, U.S. Department of Agriculture, Washington, DC.

Varela-Ortega, C., J. Sumpsi and M. Blanco (2001), "Water availability in the Mediterranean region", in F. Brouwer and J. van der Straaten (eds.), *Nature and Agriculture in the European Union: New Perspectives on Policies that Shape the European Countryside*, Cheltenham, Edward Elgar (forthcoming).

Van Huylenbroeck, G. and M. Whitby (1999), "Conclusions and policy recommendations", in G. van Huylenbroeck and M. Whitby (eds.), *Countryside Stewardship: Farmers, Policies and Markets*, Elsevier Science Ltd., Amsterdam, pp. 177-191.

Ventelä, A-M., T. Kirkkala, J. Sarvala and H. Mattila (2001), "Stopping the eutrophication process of Lake Pyhäjärvi" (mimeograph) Pyhäjärvi Protection Fund, Pyhäjärvi, Finland.

Winter, M. (2000), "The arable crops regime and the countryside", in F. Brouwer and P. Lowe (eds.), *CAP Regimes and the European Countryside: Prospects for Integration between Agricultural, Regional and Environmental Policies*, CABI Publishing, Wallingford, UK, pp. 117-136.

Wu, J., R.M. Adams, D. Zilbermann and B. Babcock (2000), "Targeting resource conservation expenditures", *Choices*, 2nd quarter 2000, pp. 33-38.