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POLICY AND PRACTICE

Policy Challenges and Priorities for Internalizing the Externalities of Modern Agriculture

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ABSTRACT *Agriculture is inherently multifunctional. It jointly produces more than food, fibre or oil, having a profound impact on many elements of economies and ecosystems. A comprehensive framework is used to present new data on annual external costs in Germany (£1.2 billion; US$2 billion), in the UK (£2.3 billion; US$3.8 billion) and in the USA (£21 billion; US$34.7 billion). These costs are equivalent to £49–208/ha (US$81–343/ha) of arable and grassland. Agriculture also produces positive externalities, and though there is no comprehensive valuation framework, the public benefits in the UK appear to be in the range of £10–30 (US$16–49) per household, or some £20–60/ha (US$32–100/ha) of arable and pasture land. These external costs and benefits raise important policy questions. In particular, should farmers receive public support for the multiple public benefits they produce? Should those that pollute have to pay for restoring the environment and human health? Policy options available for encouraging behavioural changes are of three types: advisory and institutional measures; regulatory and legal measures; and economic instruments. Three of the most promising options for discouraging negative externalities and encouraging positive ones are: (1) environmental taxes; (2) subsidy and incentive reform; and (3) institutional and participatory mechanisms. The greatest challenge, however, will be to find ways to integrate such policy tools into effective packages that will increase the supply of desired environmental and social goods whilst ensuring farmers' livelihoods remain sustainable.*

Introduction

Modern agriculture in the late 20th century was highly successful at increasing food production, with per hectare cereal yields increasing by a multiple of 2–3

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in the USA and Europe over 50 years, and by 60–80% in Asia and Latin America since the 1970s (Conway, 1997; Pretty, 1998). Unlike other economic sectors, though, agriculture is inherently multifunctional. It jointly produces much more than just food, fibre or oil, having a profound impact on many elements of local, national and global economies and ecosystems (Food and Agriculture Organization of the United Nations (FAO), 1999).

These impacts can be negative or positive. For example, an agricultural system that depletes organic matter or erodes soil whilst producing food imposes costs that others must bear; but one that sequesters carbon in soils contributes both to the global good by mediating climate change and to the private good by enhancing soil health. Similarly, a system that protects on-farm beneficial wildlife for pest control contributes to stocks of biodiversity, whilst systems that eliminate wildlife do not. Only a few of these external effects have been properly measured or costed.

The environmental and human health effects of modern agriculture are wide-ranging and well documented (Conway & Pretty, 1991; Altieri, 1995; Pretty, 1995, 1998; Mason, 1996; European Environment Agency (EEA), 1998), and include: (1) pesticides contaminating water and harming wildlife and human health; (2) nitrate and phosphate from fertilizers, livestock wastes and silage effluents contaminating water, and so contributing to algal blooms, deoxygenation, fish deaths and nuisance to leisure users; (3) soil erosion disrupting watercourses, and run-off from eroded land causing flooding and damage to housing and natural resources; (4) harm to consumers exposed to harmful residues and micro-organisms in foods; and (5) contamination of the atmospheric environment by methane, nitrous oxide and ammonia derived from livestock, their manures and fertilizers.

The social impacts associated with modern agriculture are also significant. In every country of Europe, farms have become progressively both fewer in number and larger, and some areas now suffer from land abandonment. Changing farm size and abandonment have also brought a dramatic decline in the numbers of people employed in agriculture. The 1980s saw a 10% fall in the total agricultural labour force across the European Union (EU), accounting for more than 1.93 million jobs (Bollman & Bryden, 1997; Eurostat, 1997). Fewer farms, fewer jobs and larger-scale farming have also played a role in the rise of rural poverty and economic disadvantage (Pretty, 1998; Ministry of Agriculture, Fisheries and Food (MAFF), 1999).

However, agriculture's multifunctionality indicates that it can also deliver valued non-food functions, many of which cannot be produced by other economic sectors. Much of the 'natural' biodiversity in Europe is the result of centuries of farming, which has created and shaped both landscape and countryside. There are many other positive side-effects of agriculture, including: aesthetic value; recreation and amenity; water accumulation and supply; nutrient recycling and fixation; soil formation; wildlife, including agriculturally beneficial organisms; storm protection and flood control; and carbon sequestration by trees and soils. Positive social externalities include: provision of jobs; and contributions to the local economy, and to the social fabric of rural communities (Organization for Economic Co-operation and Development (OECD), 1997; Performance and Innovation Unit (PIU), 1999).

An important policy challenge for both industrialized and developing countries is to find ways to maintain and enhance food production, while seeking both to improve the positive functions and to eliminate the negative ones, so improving the overall sustainability of rural livelihoods and economies (Carney, 1998; Potter, 1998; Pretty, 1998; Dobbs & Dumke, 1999; MAFF, 1999).

Putting a Cost on Externalities

Most economic activities affect the environment, either through the use of natural resources as an input or by using the 'clean' environment as a sink for pollution. The costs of using the environment in this way are called 'externalities' because they are side-effects of the economic activity, they are external to markets and so their costs are not part of the prices paid by producers or consumers. When such externalities are not included in prices, they distort the market by encouraging activities that are costly to society even if the private benefits are substantial (Baumol & Oates, 1988; Pearce & Turner, 1990; EEA, 1998; Brouwer, 1999; Pretty *et al*., 2000).

An externality is any action that affects the welfare of or opportunities available to an individual or group without direct payment or compensation, and may be positive or negative. The types of externalities encountered in the agricultural sector have four features: (1) their costs are often neglected; (2) they often occur with a time lag; (3) they often damage groups whose interests are not represented; and (4) the identity of the producer of the externality is not always known.

We are concerned in this article with 'technological' (or physical) externalities, rather than with 'pecuniary' (or price-effect) externalities. Pecuniary externalities result, for example, when individuals or firms purchase or sell large enough quantities of a good or service to affect price levels. The change in price levels affects people who are not directly involved in the original transactions, but who now face higher or lower prices as a result of those original transactions. These pecuniary externalities help some groups and hurt others, but they do not necessarily constitute a 'failure' of the market economy (Davis & Kamien, 1972). An example of a pecuniary externality is the rising cost of housing for local people in rural villages that results from higher-income workers from metropolitan areas moving away from urban cores and bidding up the price of housing in those villages. This pecuniary externality is a legitimate public concern, and may merit a public policy response, but it is not the kind of environmental impact that we focus on in this article.

Technological externalities, however, do constitute a form of 'market failure'. Dumping raw sewage into a lake, without payment by the polluter to those who are adversely affected, is a classic example of a technological externality. The market 'fails' in this instance, because more pollution occurs than would be the case if the market or other institutions caused the polluter to bear the full costs of their actions (Davis & Kamien, 1972). It is technological externalities that are commonly simply termed 'externalities' in most of the current environmental literature (see Common, 1995; Knutson *et al*., 1998).

In practice, there are few agreed data on the economic cost of agricultural externalities. This is partly because the costs are highly dispersed and affect many sectors of economies. It is also necessary to know about the value of nature's goods and services, and what happens when these largely unmarketed goods are lost. Some suggest that the current system of economic calculations grossly underestimates the current and future value of natural capital (Abramovitz, 1997; Costanza *et al*., 1997; Daily, 1997; *Ecological Economics*, 1999).

It is relatively easy to assess abatement and treatment costs following pollution, but much more difficult to calculate agriculture's positive functions. How do we value, for example, skylarks singing on a summer's day, a landscape with hedgerows and trees, or a watershed producing clean water? Environmental economists have developed methods for assessing people's stated preferences for environmental goods through hypothetical markets, which permit an assessment of their willingness to pay for nature's goods and services or willingness to accept compensation for losses (Stewart *et al*., 1997; Hanley *et al*., 1998; Brouwer, 1999).

The Cost of Negative Externalities

Several studies have recently sought to cost the negative externalities of modern agriculture in Germany, Netherlands, the UK and the USA (Pimentel *et al*., 1992, 1995; Evans, 1995, 1996; Steiner *et al*., 1995; Davison *et al*., 1996; Fleischer & Waibel, 1998; Waibel & Fleischer, 1998; Bailey *et al*., 1999), or to illustrate the losses in ecosystem services with the modernization of agriculture in Sweden (Björklund *et al*., 1999). The data, however, are not wholly comparable in their original form, as different frameworks and methods of assessment have been used. Methodological concerns have also been raised about some studies (see Bowles & Webster, 1995; Crosson, 1995; Van Der Bijl & Bleumink, 1997; Pearce & Tinch, 1998).

Some studies have noted that several effects cannot be assessed in monetary terms, whilst others have appeared to be more arbitrary (e.g. the $2 billion cost of bird deaths in the USA arrived at by multiplying 67 million losses by $30 per bird: see Pimentel *et al.* (1992)) (all prices in US dollars). Davison *et al.*'s (1996) study of Dutch agriculture was even more arbitrary, as it added an estimate of the costs farmers would incur to reach stated policy objectives, and these were based on predicted yield reductions of 10–25% arising from neither cheap nor preferable technologies, which led to a large overestimate of environmental damage (see Van Der Bijl & Bleumink, 1997).

We use a framework developed for a recent study of UK agriculture (Pretty *et al*., 2000) to present new comparative data on negative externalities in the UK, the USA and Germany (Table 1). The framework uses seven cost categories to assess negative environmental and health costs. Two types of damage cost were estimated for the UK: (1) the treatment or prevention costs (those incurred to clean up the environment and restore human health to comply with legislation or to return these to an undamaged state); and (2) the administration and monitoring costs (those incurred by public authorities and agencies for monitoring environmental, food and health parameters). Only those externalities which gave rise to financial costs were estimated.

The framework includes only external costs, i.e. the costs passed on to the rest of society through the actions of farmers. Additional private costs borne by farmers themselves are not included, such as those resulting from increased pest or weed resistance from the overuse of pesticides, or from training in the use, storage and disposal of pesticides. However, there remain unmeasured distributional problems: for example, insect outbreaks arising from pesticide overuse can affect all farmers, even those not using pesticides.

Table 1. The annual external costs of modern agriculture in the UK, the USA and Germany (£ million, adjusted to 1996 prices)

Cost category	UK (£ million)	USA[a] (£ million)	Germany[a] (£ million)
1. *Damage to natural capital: water*			
a) Pesticides in sources of drinking water	120	592	58
b) Nitrate, phosphate and soil in sources of drinking water	71	814	+
c) Zoonoses (esp. *Cryptosporidum*) in sources of drinking water	23	+	+
d) Eutrophication, pollution incidents, fish deaths, monitoring costs	17	170	33
2. *Damage to natural capital: air*			
Emissions of methane, ammonia, nitrous oxide and carbon dioxide	1113	10 936[b]	1125[b]
3. *Damage to natural capital: soil*			
a) Off-site damage caused by erosion	14		+
Flooding, blocked ditches and lost water storage		2 287	
Damage to industry, navigation and fisheries		5 765	
b) Organic matter and carbon dioxide losses from soils	82	+	+
4. *Damage to natural capital: biodiversity and landscape*			
a) Biodiversity/wildlife losses	25	218	4
b) Hedgerows and drystone wall losses	99	+	+
c) Bee colony losses and damage to domestic pets	2	152	1
5. *Damage to human health: pesticides*	1	88	9
6. *Damage to human health: nitrate*	0	+	+
7. *Damage to human health: micro-organisms/disease agents*[c]			
a) Bacterial and viral outbreaks in food	169	+	+
b) BSE and new variant CJD	607	+	+
c) Overuse of antibiotics	+	+	+
Total annual external costs	£2342	£21 022	£1230
Total costs per hectare of arable and grassland	£208	£49	£71
Costs per hectare of arable only[c]	£228	£68	£166
Costs per kilogram of pesticide active ingredient[d]	£8.6	£2.2	£3.9

Sources: adapted from Pretty *et al.*, 2000; Pimentel *et al.*, 1992, 1995; Fleischer and Waibel, 1998; Ribaudo *et al.*, 1999; For full details of methodology, see Pretty *et al.* (2000).

[a]The Published data for the USA have been adjusted to remove some clear overestimates together with private costs borne by farmers themselves (see Pimentel *et al.*, 1992). The published data for the USA and Germany are supplemented with new estimates for the costs of gaseous emissions using data from the US EPA (1999) and the EEA (1999). For full details of methodology, see Pretty *et al.* (2000).

[b]The costs of emissions in the USA and Germany do not include data for ammonia (4% of cost in UK).

[c]Arable farming costs are assumed to be: 80% of categories 1a, 2 (nitrous oxide); 50% of 1b, 1d-e, 2 (methane and carbon dioxide), 3, 4, 5; and 25% of 7a (i.e. £1048 million in the UK, £12 753 million in the USA, £871 million in Germany).

[d]Pesticide costs are assumed to be 100% of category 1a and 5; and 50% of 1d–e and 4 (i.e. £193 million in the UK, £950 million in the USA, £104 million in Germany). In the late 1990s, pesticide use in US agriculture was 425 000 000 kg active ingredient (a.i.) (77% of total), in UK agriculture 22 500 000 kg a.i. (89% of total) and in German agriculture 26 500 000 kg a.i. (89% of total).

Note: + , Cost estimates not yet calculated (or available).

Even though the research to date still contains many gaps where costs have yet to be calculated, these studies suggest that the external costs of modern agriculture in 1996 amounted to £49–71/ha ($81–117/ha) of arable and grassland in Germany and the USA, but rising to £208/ha ($343/ha) in the UK. These differences, however, may not be significant, owing to large gaps and uncertainties in the data (without the cost of bovine spongiform encephalopathy (BSE), for example, the costs in the UK come down to £154/ha). It appears that costs per hectare of arable are higher than for livestock, and the external costs associated only with pesticides amount to £2.2–8.6/kg ($3.7–14.2/kg) of active ingredient used. These are substantial burdens on non-agricultural sectors of economies.

For a variety of reasons, these estimates are likely to be conservative (Pretty *et al.*, 2000):

- some costs are known to be substantial underestimates (for example, acute and chronic pesticide poisoning of humans, monitoring costs, eutrophication of reservoirs and restoration of all hedgerow losses);
- some costs currently cannot be calculated (for example, dredging to maintain navigable water, flood defences, marine eutrophication and poisoning of domestic pets);
- the costs of returning the environment or human health to pristine conditions were not calculated;
- treatment and prevention costs may be underestimates of how much people might be willing to pay to see positive externalities created;
- the data do not account for time lags between the cause of a problem and its expression as a cost (i.e. some processes long since stopped may still be causing costs; some current practices may not yet have caused costs);
- this study did not include the externalities arising from transporting food from farms to manufacturers, processors, retailers and finally to consumers.

The Value of Positive Externalities

These cost estimates only tell part of the story, as agriculture also produces positive externalities, such as landscape and aesthetic value, water supply, nutrient fixation, soil formation, biodiversity, flood control and carbon sequestration (OECD, 1997). However, there is no comprehensive study to show the aggregate positive side-effects of agriculture.

There have been several studies of the effect of joint agri-environmental policies, with an attempt to put a value on the positive environmental and landscape outcomes (for example, Willis *et al.*, 1993; Foster *et al.*, 1997; Hanley *et al.*, 1998). Benefit estimates for agri-environmental schemes implemented in the UK during the late 1980s and early 1990s are summarized in Stewart *et al.* (1997) and Hanley *et al.* (1999a). Many of the UK's agri-environmental schemes have constituted attempts to restore some of the landscape, habitat and other positive countryside attributes that were lost during the 1940s–1970s intensification phase and to protect those attributes not yet lost. Therefore, benefit estimates for agri-environmental schemes at least offer insights into some of the values society places on particular services agriculture provides in addition to food and fibre.

UK agri-environmental schemes have been designed to deliver benefits in several forms, including wildlife effects, landscape effects, water quality, archaeological sites and enhanced access. Benefits may accrue to those in the immediate area of a scheme, to visitors from outside the area and to the general public (Hanley *et al.*, 1999a, b). The benefit estimates cited by Stewart *et al.* (1997) and Hanley *et al.* (1999a) show that benefits substantially exceed costs, thus confirming the value of the policies. Valuations of 10 Environmentally Sensitive Areas (ESAs) in England and Scotland, plus the Nitrate Sensitive Area (NSA) and Organic Aid schemes, demonstrate a benefit of £45–257 million, over a financial cost to the Treasury of £10.1 million (Hanley *et al.*, 1999a).

The benefit estimates (using a variety of valuation methods: contingent valuation, choice experiments and contingent ranking) per household varied from £2 to £30 for most ESAs, rising to £140–150 for the Norfolk Broads and £378 for Scottish machair grasslands, and were £16–18 per household for the NSA and Organic Aid schemes. The value to local residents showed lower variation than the values to visitors and the public (mostly in the range £2–20, with some large outliers at £150 per household).

Stewart *et al.* (1997) and Hanley *et al.* (1999a) point out that these estimates can only be indicative, and that continued assessment is needed to refine the findings. The estimates may also be confusing the valuation of a total landscape or habitat with the effect of the policy itself. However, if we take the range of annual benefits per household to be £10–130, and assume that this is representative of the average household's preferences for all landscapes produced by agriculture, then this suggests national benefits of the order of £200–600 million. Expressed on a per hectare basis, these indicative figures suggest benefits of the order of £20–60/ha of arable and pasture land in the UK.

On the one hand, these are likely to be overestimates (assuming agri-environmental schemes have already been selected because of their higher value); on the other hand, they could be substantial underestimates, as they omit to value such benefits as pathogen-free foods, uneroded soils, emission-free agriculture and biodiversity-producing systems, as well as focusing on the outcomes of a scheme rather than the whole landscape.

There are too few studies yet to corroborate these data. One study in the UK compared paired organic and non-organic farms, and concluded that organic agriculture produces £75–125/ha ($125–200/ha) of positive externalities each year (with particular benefits for soil health and wildlife) (Cobb *et al.*, 1998). As there were some 2 100 000 ha of organic farming in Europe in 1999, this suggests that the annual positive externalities are of the order of £157–262 million ($262–420 million) (assuming this estimate holds for the many different farming systems in Europe).

In some circumstances, the public benefits provided by farming may exceed the private returns obtained by farmers themselves. Sala & Paruelo (1997) calculate the annual value of carbon sequestration in the US Great Plains to be $200/ha (£125/ha), four times as great as the net private returns to farmers for meat, wool and milk, and about half the market value of land. In effect, these farmers are joint 'carbon farmers' rather than just livestock producers.

An important principle is that it is more efficient to use limited resources to promote practices that do not damage the environment than to have to spend them on cleaning up after a problem has been created. The savings with positive environmental actions can be substantial, as New York State has found out with its support for sustainable agriculture in the 512 000 ha Catskill–Delaware watershed complex (Institute for Agriculture and Trade Policy, 1997). New York City gets 90% of its drinking water from these watersheds. In the late 1980s, though, the city was faced with having to construct a filtration facility to meet new drinking water standards, the cost of which would be $5–8 billion, plus another $200–500 million in annual operating costs. Some 40% of the cropland in the watershed would have had to be taken out of farming so as to reduce run-off of eroded soil, pesticides, nutrients and bacterial and protozoan pathogens.

Instead, the city decided on a collaborative approach with farmers. It funded a Watershed Agricultural Council, a partnership of farmers, government and private organizations with the aim of both protecting the city's drinking water supply and sustaining the rural economy. It works on whole-farm planning with each farm, tailoring solutions to local conditions to maximize reductions in off-site costs. The first two phases of the programme, leading to the 85% target for pollution reduction, cost some $100 million, a small proportion of the cost of the filtration plant and its annual costs. Not only do taxpayers benefit from this joint approach to agri-environmental management, but so do farmers, the environment and rural economies.

Policies to Internalize Externalities

These external costs and benefits raise important policy questions. In particular, should farmers receive public support for the multiple public benefits they produce in addition to food? Should those that pollute have to pay for restoring the environment and human health?

Recent decades have seen considerable progress in setting out challenges and options in the modification of national account systems, so that they take account of both the costs and the benefits of using the environment (Repetto *et al.*, 1989; El Serafy, 1997; Bartelmus, 1999; Turner & Tschirhart, 1999). In addition, there is an emerging and significant literature on the use of indicators to measure progress towards sustainability in agriculture (Bailey *et al.*, 1999; OECD, 1999; MAFF, 2000), and at national level (Pearce, 1993; Jackson & Marks, 1994). In agriculture, the focus has been on a wide range of policy options that are available to encourage changes in farmers' behaviour and practices. These fall into three categories: advisory and institutional measures; regulatory and legal measures; and economic instruments (Table 2). In practice, effective pollution control and optimal supply of desired public goods require a mix of all three approaches, and integration across sectors.

Advisory and institutional measures have long formed the backbone of policies to internalize costs and so prevent agricultural pollution. These rely on the voluntary actions of farmers, and are favoured by policy makers because they are cheap and adaptable. Advice is commonly in the form of codes of good agricultural practice, such as recommended maximum rates of application of pesticides and fertilizer, or measures for soil erosion control. Most governments still have agricultural extension services and employ extension agents to work with farmers on technology development and transfer. Such advisory and institutional measures, though, do not necessarily guarantee outcomes with greater environmental or social benefits.

Regulatory and legal measures are also used to internalize external costs. This can be done either by setting emissions standards for the discharge of a pollutant or contaminant, or by establishing environmental quality standards that relate to the environment receiving the pollutant. Polluters who exceed standards are then subject to penalties. There are many types of standards: operating standards to protect workers; production standards to limit levels of contaminants of residues in produce (for example, pesticide residues in foods); emission stan-

Table 2. Policy instruments for internalizing positive and negative agricultural externalities

Instrument/measure	Policy aimed at increasing supply of positive externalities	Policy aimed at decreasing supply of negative externalities
Advisory/institutional		
Codes of good practice	Set out best practice, but do not guarantee supply	Encourage willing farmers by setting out best management practices
Extension systems (publicly-funded)	Close links between professionals and farmers encourage transformations	Close links allow for ready flow of information and monitoring of farm practices
Participatory processes and farmers' groups	Groups increase farmers' capacity to experiment with sustainable practices (e.g. pest or land management groups)	Groups increase farmers' capacity to reduce polluting practices
Environment audits	Encourage stepwise changes towards increased use of regenerative resources	Encourage farmers to reduce wastes and pollution
Regulatory		
Regulations and laws, (including policies applied by private sector, e.g. supermarkets)	Regulations not used to increase positive externalities	Widespread use, e.g. food standards, non-spraying in riparian strips, water quality and food standards, mandatory practices to limit nutrient leaching, ban on straw burning, protection of habitats
Designations for habitats and species	Neutral	Legal protection of key habitats and species
Economic		
Environment taxes	Do not increase positive externalities (though hypothecated revenue can be used for incentives)	Pesticide taxes, fertilizer taxes, and manure charges in place
Subsidy and incentives reform	Cross-compliant and conditional subsidies coupled with provision of environmental and social goods	Indirect effects—if more sustainable systems emerge
Grants and loans	One-off grants (e.g. for hedgerows or drystone walls)	One-off grants (e.g. for slurry storage facilities or soil conservation measures)
Tradable pollution permits	Not effective	Only one application in agriculture—for water quality improvement (in USA)

dards to limit releases or discharges (for example, silage effluents or slurry); and environmental quality standards to limit levels of undesirable pollutants in vulnerable environments (for example, nitrate or pesticides in water).

However, the problem with such regulations is that most agricultural pollutants are diffuse, or non-point, in nature. It is, therefore, impossible for inspectors to ensure compliance on hundreds of thousands of farms in the way that they can with a limited number of factories. Regulations are also used to limit or eliminate certain farm practices, such as the spraying of pesticides close to watercourses or straw burning in the UK, or the mandatory requirement for farms to complete full nutrient accounts (for example, in the Netherlands and Switzerland). A final use for regulations is the designation and legal protection of certain habitats and species. These can be set at national or international level.

Again, such designations do not guarantee protection, though they draw clear attention to their social value.

Economic instruments are primarily designed to ensure that the polluter bears the costs of the pollution damage and/or the costs incurred in controlling the pollution (the abatement costs). This implies that the free input to farming, a clean or unpolluted environment, is priced and treated as if it were similar to other costs (such as for labour or capital). This is the polluter pays principle, which was accepted by all governments of the OECD in 1972 and later, in 1995, laid down in the Treaty of Rome (Conway & Pretty, 1991; Ekins, 1999). A variety of economic instruments are available for achieving internalization, including environmental taxes and charges, tradable permits and the targeted or coupled use of public subsidies and incentives.

Taxes and subsidies are two sides of the same economic incentives coin. Although either approach can be used to reduce negative externalities from agriculture, and effects on farmers' production and pollution control may be similar, the two approaches will have different effects on farmers' profits and on public budgets (Ekins, 1999; Ribaudo *et al.*, 1999). Recent years have seen a small but significant shift in the use of public subsidies for joint agri-environmental outcomes. Agri-environmental policies in the UK have tended to emphasize the property rights of farmers, implying that farmers should be compensated for alterations in practices which reduce their profits. This is in contrast to many other European countries, where the state is presumed to have greater vested property rights, implying a stronger role for regulation (Hanley & Oglethorpe, 1999). The US philosophy of farm property rights is closer to that of the UK, in that regulation without compensation has been minimal to date.

Tradable permits are systems of quantitative pollution allowances which can be bought and sold in a permit market. The effect is to concentrate pollution control activity amongst those who bear the least costs, thus reducing the costs of compliance. Tradable permits have been used for acid rain and fisheries control (in the USA, Australia and New Zealand), but only recently in an agricultural context—for water quality improvement in the USA. Point-source polluters subject to regulation can purchase emissions allowances from non-point-source agricultural polluters. The concept is based on the idea that the cost of further reducing pollution of a water body may sometimes be lower through adjustments in farming practices than through additional reductions in point-source loadings (Ribaudo *et al.*, 1999).

Three of the most promising policy options both to discourage negative externalities and to encourage positive externalities are dealt with in more detail below: (1) environmental taxes; (2) subsidy and incentive reform; and (3) institutional and participatory mechanisms.

Environmental Taxes

Environmental or 'eco' taxes seek to shift the burden of taxation away from economic 'goods', such as labour, towards environmental 'bads', such as waste and pollution. The market prices for agricultural inputs and products do not currently reflect the full costs of farming. Environmental taxes or pollution payments, however, seek to internalize some of these costs, so encouraging individuals and businesses to use resources more efficiently. Such green taxes offer the opportunity of a 'double dividend' by cutting environmental damage,

Box 1. The effectiveness of environmental taxes in the EU. *Source:* EEA (1999).

- Taxes produce environmental benefits and are cost-effective.
- Examples of particularly effective taxes are those on Swedish air pollution and on Dutch water pollution, and the NO_x charge and tax differentiation schemes for vehicle fuels in Sweden.
- Incentive taxes are environmentally effective when the tax is sufficiently high to stimulate abatement measures.
- A significant contribution to the environmental effectiveness of the charges is provided when the revenues are used for related environmental expenditures.
- Taxes can work over relatively short periods of time (2–4 years), and so compare favourably with other environmental policy tools, though energy taxes can take 10–15 years to exert substantial incentive effects.

particularly from non-point sources of pollution, whilst promoting welfare (EEA, 1996; Smith & Piacentino, 1996; Ekins, 1999).

There is still, however, a widespread view that environmental taxes stifle economic growth. Growing empirical evidence on the costs of compliance with environmental regulations and taxes suggest that there has been little or no impact on the overall competitiveness of businesses or countries, with some suggestion that these regulations and taxes have increased efficiency and employment (EEA, 1996, 1999a; Jarass & Obermair, 1997; OECD, 1997; Department of the Environment, Transport and the Regions (DETR), 1999) (see Box 1).

There are a wide variety of environmental taxes and levies in countries of Europe and North America (see Ekins (1999) for a comprehensive review). These include carbon/energy taxes (for example, in Belgium, Denmark and Sweden); chlorofluorocarbon taxes (Denmark and the USA); sulphur taxes (for example, Denmark, France, Finland and Sweden); nitrous oxide charges (France and Sweden only); leaded and unleaded petrol differentials (all EU countries); landfill tax (Denmark, the Netherlands and the UK); groundwater extraction charges (the Netherlands only); and sewage charges (Spain and Sweden). Environmental taxes have tended not to be applied to agriculture, with the notable exception of: pesticide taxes in Denmark, Finland, Sweden and several states of the USA; fertilizer taxes in Austria (1986–94), Finland (1976–94), Sweden and, again, several states of the USA; and manure charges in Belgium and the Netherlands.

Much attention has recently been focused on the implementation of pesticide taxes (Rayment *et al.*, 1998; DETR, 1999). The ideal situation for a pesticide tax is for the highest costs to be imposed on products causing the most harm to environmental and human health. However, there is no accepted methodology for hazard ranking. There are various options, including a banding system, with pesticides grouped into classes with similar impact, and an *ad valorem* or kilogram-based tax, with tax as a proportion of price or imposed on pesticide use.

An important question remains as to what happens to farmers' behaviour following the establishment of a pesticide tax. In particular, if prices increase, will the use of pesticides fall? The price elasticity of demand is important for determining such environmental effects, and estimates from the Netherlands, Greece, France, Germany, Denmark and the UK put it generally between -0.2 and -0.4, with a few ranging up to -0.7 to -1.0 (Rayment *et al.*, 1998; Waibel & Fleischer, 1998). This seems to imply that it will take a large price change for farmers to change their practices: inelastic demand limits environmental effectiveness, though it is good for generating revenue.

However, there are several reasons why elasticity is probably higher. First, demand is inelastic if there is an expectation that price rises will quickly be reversed, but if farmers come to accept that higher prices incorporating the environmental taxes will remain in place, then further behaviour changes will occur. Second, a well designed package of taxes with regulations, advice and incentives can increase price responsiveness. Third, as innovation increases, more sustainable agriculture options become available to farmers, so promoting further change.

Where taxes have been established, they have been levied on sales price or kilograms of active ingredient used. These taxes vary from 0.7% of sales price (USA) to 36% (Denmark), and have had different effects, the best being a 65% reduction in pesticide use in Sweden since 1985. Revenue raised ranges from £37 million ($59 million) per year in the USA (of which 24% is from California alone), to £12.5 million ($20 million) in Norway and £0.9 million ($1.5 million) in Sweden (Pretty, 1998).

Fertilizer taxes have been introduced in several countries, and are currently of the order of £0.06–0.25/kg ($0.1–0.4/kg) of nitrogen, phosphorus and potassium in Austria, Norway and Sweden, though much lower in the USA (£0.0004–0.0125/kg ($0.0006–0.02/kg) of nitrogen in various states). Most agree that tax packages having the greatest impact on externalities are those combined with other policy instruments (advice, incentives and regulations) and those that are hypothecated—with the revenue raised being reinvested solely to promote more sustainable alternatives.

Subsidy and Incentive Reform in Agricultural Policies

The alternative to penalizing farmers through taxation is to encourage them to adopt non-polluting technologies and practices. This can be done by offering direct subsidies for adoption of sustainable technologies (a potential use for revenue raised from environmental taxes) and removing perverse subsidies that currently encourage polluting activities (Myers, 1998; Potter, 1998; Dumke & Dobbs, 1999; Hanley & Oglethorpe, 1999). An important policy principle suggests that it is more efficient to promote practices that do not damage the environment than to spend money on cleaning up after a problem has been created.

All governments provide some public support to their domestic agricultural and rural sectors. Countries of the OECD provide some £113 billion ($180 billion) support to their farming sectors, which represents some £9700 ($16 000) per full-time farmer-equivalent, or £108/ha ($179/ha) of farmland. There is, though, great variation between countries, from the high level of support in Japan of 71% of the total value of agricultural production to just 3% in New Zealand (Pretty, 1998). Some countries have begun to reduce support prices, and replace them with systems of direct payments. Some have retargeted support to sustainable practices with considerable positive effect. Generally, only a small amount is targeted on environmental improvements, through policies such as the US Conservation Reserve Program, the EU agri-environmental programme and the Australian National Landcare Programme.

Box 2. A selection of agri-environment schemes applied during the 1990s in the EU. *Source:* ADAS (1996), Barrés (1996), Countryside Council of Wales (1996) and Pretty (1998).

- Marketentlastungs und Kulturlandschaftsausgleich (MEKA) scheme of Baden-Württemburg, Germany, offers farmers an à la carte menu of technologies from which to choose, each one earning them 'eco-points'. Each point brings them DM20/ha (for example, using no growth regulator attracts 10 points, sowing a green manure crop in the autumn earns 6 points, applying no herbicides and using mechanical weeding gets 5 points, cutting back livestock to 1.2–1.8 adult units per hectare brings 3 points and direct drilling on erosive soils earns 6 points).
- Initiatives in the Parcs Naturels Régionaux of France are reconciling economic development with nature conservation through an integrated approach, including a process of negotiation and communication between the various local actors, and voluntary contracts for farmers. As the schemes are not imposed, but rather developed through a participatory process, farmers do not feel coerced. Significant examples include the high pastures of the Vosges mountains, and the marshes on the Cherbourg peninsula.
- La Albufera National Park in Spain is a 20 000 ha site of international importance to birds and fish, where farmers are subsidized to reintroduce traditional rice cultivation measures, such as mechanical weeding and premature flooding of rice fields in November–March, and to reduce their use of inputs. The scheme has protected one fish species that lives in the rice fields, and populations of teal and shovelduck are rising.
- The Tir Cymen (now Tir Gofal) Scheme, Wales, rewarded farmers for using their skills and resources to look after the landscape and wildlife and improve their farming practices. Priority was given to activities that offered the most public benefit in environmental terms, with farmers following a whole-farm agreement for 10 years. The scheme benefited farming, wildlife, the environment and local economies.

Common Agricultural Policy Reforms

The main policy instrument for agriculture in the EU, the Common Agricultural Policy (CAP), has undergone gradual greening in recent years. The 1992 MacSharry reforms introduced agri-environmental support under regulation 2078/92, resulting in about 3.5% of total support being diverted to the production of environmental goods. The responsibility for implementation and the degree of support, though, fall to individual member countries, with variable uptake, from the 100% of farmland designated in Austria to just 0.2% in Belgium.

Nonetheless, some agri-environmental schemes have been extremely successful at supporting farm transformations that produce both private benefits for farmers and public ones for the environment and rural communities (Box 2). In the UK, they have mostly resulted in benefits substantially greater than the costs (Harrison-Mayfield *et al.*, 1996; Stewart *et al.*, 1997; Hanley *et al.*, 1999a).

The latest reforms, the Agenda 2000 package agreed in 1999 by the Council of Ministers in Berlin and given substance in Helsinki, sought to deal with continued oversupply of beef and arable products, to prepare for the next round of World Trade Organization (WTO) negotiations, and to pave the way for considerable EU enlargement. These reforms reduced compensation payments, reformed quotas and made provisions for rural development measures. In November 1999, EU agriculture ministers agreed on two new regulations under the CAP, both of which offer expanded opportunities for positive environmental and rural development outcomes.

(1) The Common Rules regulation (EC 1259) introduces environmental protection requirements in relation to support payments, sets out modulation as a

possible measure for reallocating support to rural development and permits member states to decide what measures to take.

(2) The Rural Development regulation (EC 1257) establishes a rural development pillar of CAP, and acknowledges rural development as an integral part of the CAP and a key element of the multifunctional character of European agriculture.

There are still, however, several reasons to believe that agricultural policies will need to change yet further. There are pressures arising from EU enlargement—a large number of central and east European countries have been offered EU membership—yet the EU cannot afford to pay all its farmers similar levels of support. The cost of expanding the CAP to just Poland, Hungary, the Czech Republic and Slovenia is estimated to be an additional £9 billion/year (15 billion Euros/year). There are continuing pressures from the WTO, which stipulates that agricultural support will have to be decoupled from production, and so payments to farmers will increasingly have to be for environmental and social goods if they are to continue to be permitted. There are also pressures from consumers and taxpayers, who pay twice for food: through taxes being used for direct support to farmers; and through having to pay again for the external costs being imposed on nature, rural communities and human health.

US Farm Bill Reforms

Reforms began in the USA with the 1985 'farm bill' and continued with the 1990 and 1996 'farm bills'. The 1996 legislation, the Federal Agricultural Improvement and Reform Act, generally referred to as 'Freedom to Farm', was the most dramatic break with commodity-based farm legislation in the USA since the 1930s. Although driven primarily by government budget concerns, the supposed decoupling of income supports from the production of particular crops held out some hope of additional ecological dividends through crop diversification (Dobbs & Dumke, 1999).

However, initial assessments of the impacts of the 'Freedom to Farm' legislation, while noting the flexibility of individual farmers to shift crops, are not finding any strong tendency towards greater system diversity (Johnston & Schertz, 1998; Karmen, 1998; Lin *et al*., 1999). If anything, the tendency has been to specialize even more in a few major export crops, especially maize and soybeans. In spite of the new planting flexibility enabled by the 1996 policy changes, several factors continue to exert strong pressures on farmers to remain highly specialized, with little ecological diversity in their crop rotations. Those factors include technologies, the current structure of agriculture and access to markets (Dobbs & Dumke, 1999).

Research and technology development, in both public and private sectors, have focused heavily on a few major crops, especially maize, wheat and soybeans in the US agricultural heartland. New seed varieties, such as *B.t.* maize and Round-up Ready soybean, have encouraged more specialized rotations. The evolution of machine technologies towards large and specialized equipment makes it difficult for farmers to diversify into a variety of crops that might require different pieces of expensive equipment.

Various aspects of the structure that has emerged in US agriculture, including ever-larger farm sizes, many spouses and children no longer working on the farm and the complexity of marketing institutions, contribute to farmers' tendency to remain specialized. Moreover, as crop systems have become narrower, easily accessible markets no longer exist for a wide range of crops in any given area. The implication is that policy reform will have to be broad, encompassing areas other than just price support policy, if crop system diversity is to be restored. Stronger positive incentives, in the form of 'green' or 'stewardship' payments for farmers, may also be needed.

However, funding for environmental programmes is meagre in the USA. Most agri-environmental programmes other than the Conservation Reserve Program (which takes land out of production) were combined in the new Environmental Quality Incentives Program (EQIP) in 1996. Funding for the programmes which were consolidated into EQIP had averaged approximately $1 billion/year (1992 constant dollars) over the period 1983–92 (Heimlich & Claassen, 1999). EQIP is funded at only $200 million/year, compared with historic farm income support payments in the USA of $7–12 billion/year (Batie, 1998).

Institutional and Participatory Mechanisms

Although taxes and financial incentives play an important role in encouraging the adoption of sustainable practices, they must be supplemented with additional processes that support communication and learning amongst farmers for maximum impact. For example, many farmers perceive environmental regulations as a constraint and, though they may adopt new practices, they may do so only grudgingly, with a tendency to revert once schemes have ended. Many also do not bother at all with existing schemes: in the UK, for example, less than one-third of the designated area of ESAs has been entered into by farmers, even though they can receive financial support; and in Denmark, there is a large gap between the budgeted area and the amount of land covered by agreements (MAFF, 1997; Just, 1998; PIU, 1999).

The basic challenge is to find ways to encourage voluntary transitions towards more sustainable practices, and to avoid short-term compulsion, which has the attendant danger that compliers will revert to old practices as soon as the policy measures stop. Recent years have seen a rapid growth in interest in the term 'social capital'. It captures the idea that social bonds and social norms are an important part of transformations (Coleman, 1990; Putnam, 1993; Ostrom, 1998). Social capital implies: good relations of trust, regular reciprocity and exchanges between individuals and groups; the presence of common rules, norms and sanctions; and connectedness in the form of networks and institutions (Pretty & Ward, 2001). Some of the benefits of social capital include reduced costs of conducting business (including lower costs of negotiating collective actions and agreements, and increased ability to exploit economies of scale); increased capacity to innovate (such as through membership of farmers' groups which are well connected to research and extension agencies); and improved access to and influence on policies and legislation.

A variety of institutional mechanisms can help to increase social capital and the uptake of more sustainable practices, including: encouraging farmers to work together in study groups; investing in extension and advisory services to encourage greater interaction between farmers and extensionists; and encouraging new partnerships between farmers and other rural stakeholders, as regular exchanges and reciprocity increase trust and confidence, and lubricate co-operation.

Recent years have seen an extraordinary expansion in farmer-participatory or joint management programmes throughout the world. These advances have been centred on participatory and deliberative learning processes, leading to local group formation for watershed/catchment management, irrigation management, micro-finance delivery, forest management, integrated pest management and farmers' research groups. In the past decade, some 350 000 new groups have arisen in these sectors, mostly in developing countries (Pretty & Ward, 2001).

In Denmark, farmers organized into the 720 crop protection groups are easily able to access information from extension systems. These 4300 members (one-seventh of all full-time farmers) have achieved the greatest reductions in both pesticide use (doses and frequency) and input cost (Just, 1998). The strong social capital manifested in groups helps to get costs down for farmers whilst also protecting the environment.

Similarly, in the USA, Practical Farmers of Iowa organizes farmers into groups for experimentation, the sharing of results and the co-adoption of sustainable practices, and has seen members make $80/ha savings over neighbouring conventional farmers (largely from a 50–60% reduction in pesticide and fertilizer use) whilst maintaining cereal yields (Harp *et al*., 1996).

One of the best national examples of rural partnerships and group formation comes from Australia, where a remarkable social experiment has been under way since the end of the 1980s. The National Landcare Programme encourages groups of farmers to work together with government and rural communities to solve a wide range of rural environmental and social problems. By the end of the 1990s, there were 4500 active local groups, with one-quarter to one-third of all Australian farm families as members of groups (National Landcare Programme, 1999; Pretty & Frank, 2000).

The importance of co-learning has been shown clearly by research into the conversion to organic farming in West Jutland, Denmark and in Drôme in south-east France (Assouline, 1997; Just, 1998). Though several factors were important in the transition process, such as the presence of good local pioneers and effective extensionists, it was found that sustainable agriculture spread most quickly amongst farmers organized in groups. In mountainous Diois, farmers chose to work together in groups, so advancing the shift towards sustainable farming; but in nearby Val de Drôme, farmers work less together, and the spread of sustainable technologies is slower.

The Challenge of Integration

The substantial external costs of modern agriculture, varying in the range £49–208/ha ($81–343/ha), and the known external benefits of some agricultural systems (in the UK, in the range of £20–60/ha ($32–100/ha)), pose great challenges for policy makers, farmers and scientists. A range of policy reforms could do much to internalize some of these costs and benefits in prices. In practice, as no single solution is likely to suffice, the key issue rests on how policy makers choose an appropriate mix of solutions, how these are integrated and how farmers, consumers and other stakeholders are involved in the process of reform itself.

The fundamental challenge is to develop more sustainable farm practices that produce enough food as well as maximizing the positive external benefits of agriculture, and then to find ways to encourage farmers to adopt them. Attention will therefore need to be paid to the social and institutional processes that both encourage farmers to work and learn together, and result in integrated cross-sectoral partnerships.

Policy integration is vital. In recent years, there have clearly been an increasing number of policies seeking to link agriculture with more environmentally sensitive management. However, these are still highly fragmented. Such lack of integration in Europe was brought into sharp focus in February 1998 by the then EU Commissioner for the Environment, Ritt Bjerregaard, who drew attention to the major divide between the way the public at large perceive environmental problems and the way legislators deal with them:

> We divide problems into manageable chunks, reflecting the established divisions of competence and responsibility of individual ministries and departments … Citizens expect us to ensure clean air, water, healthy food and protection of wildlife and the countryside and to safeguard these values for the future: this is a wider integrated vision … We have to date made little progress in adapting our policy and decision-making to incorporate this wider integrated vision. (EEA, 1999b, p. 8)

In Europe, most stakeholders agree that the CAP should be further reformed by decoupling payments from farm productivity. A policy framework that integrates support for farming together with rural development and the environment could create new jobs, protect and improve natural resources and support rural communities. The key mechanism is an expansion in environmental payments to farmers, tied to a menu of options for farmers. Such a policy could have many of the elements of the progressive Swiss policy reforms made during the late 1990s—a radical package supported by 70% of the public in the 1996 referendum (Swiss Agency for Environment, Forests and Landscape, 1999).

The Swiss Federal Agricultural Law was reframed in 1992 to target subsidies towards ecological practices, and then amended in 1996 as the Agricultural Act 2002, following a national referendum. Policy now differentiates between three different levels of public support, depending on the sustainability of agriculture. Tier 1 is support for specific biotypes, such as extensive grassland and meadows, high-stem fruit trees and hedges. Tier 2 supports integrated production with reduced inputs, meeting higher ecological standards than conventional farming. Tier 3 is support for organic farming.

There are five minimum conditions necessary for farmers to receive payments for integrated production, the so-called 'ecological standard' of performance:

(1) provide evidence of balanced use of nutrients with fertilizer matched to crop demands, and livestock farmers having to sell surplus manures or reduce livestock numbers;
(2) soils must be protected from erosion: erosive crops (for example, maize) can only be cultivated if alternated in rotation with meadows and green manures;
(3) at least 7% of the farm must be allocated to species diversity protection through unfertilized meadows, hedgerows or orchards;
(4) use of diverse crop rotations;

(5) pesticides have to be reduced to established risk levels.

A vital element of the policy process is that responsibility to set, administer and monitor is delegated to cantons, farmers' unions and farm advisors, local bodies and non-governmental organizations. By 1999, 90% of farms were able to comply with the basic 'ecological standard' (which allows them to receive public subsidies). Some 5000 farms (8%) are now organic (up from 2% in 1991), and most farmers are now expected to meet the 'ecological standard' during 2000. Pesticide applications have fallen by 23% since 1990, and phosphate use is down from 83 kg/ha to 73 kg/ha.

There are now opportunities in the continuing reform processes in both the EU and the USA to incorporate new mechanisms for the better integration of policies, and for better remuneration of farmers for the joint production of multifunctional benefits from agriculture, combined with more targeted internalization of costs to discourage polluting activities.

Conclusions

Agriculture is a complex economic sector, with multifaceted effects on environmental, social and human systems. A range of policy measures and tools are available for internalizing agriculture's negative and positive externalities, to which monetary values and costs are being increasingly well allocated. There are three categories of policy options available for encouraging changes in farmers' behaviour and practices: advisory and institutional measures; regulatory and legal measures; and economic instruments. In practice, effective pollution control and the optimal supply of desired public goods require a mix of all three approaches, integration across sectors and full participation of all key stakeholders in the policy development and implementation process itself.

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