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1. Introduction

Agriculture contributes negative and positive externalities to society, that is, beneficial and detrimental changes in human wellbeing to third parties for which they are not generally compensated or charged. Beneficial externalities include the creation of amenity and landscape and negative externalities include pollution of surface and groundwater. In so far as parts of agricultural subsidies compensate for beneficial externalities, they are said to be 'internalised' and should not be the subject of further policy measures. However, agricultural subsidies also add to the negative externalities by expanding output and encouraging environmentally detrimental farming practices. Comprehensive attempts to value these externalities in the UK and to compare them to the true value added of the agricultural sector are to be found in Hartridge and Pearce (2002) and Pretty et al. (2000). A particular feature of the negative externalities is the damage done by nutrient pollution and by pesticides. Nutrient pollution refers to water pollution mainly from nitrates and phosphorus, concentrations being elevated by leaching from soils of fertilisers and animal manure and slurry. A similar leaching process occurs with pesticides. Significant repositories for these leached pollutants are surface waters and groundwater.

In the UK, the main source of nitrate pollution in freshwater is agriculture, with some river pollution being due to sewage. Sources of agricultural nitrogen are inorganic fertilisers and livestock manure and slurry, As far as phosphorus is concerned, sewage discharges and agriculture both contribute, but agriculture provides the major input from manure and slurry and inorganic fertilisers. Eutrophication of water results from excess nutrient loading. Eutrophication contributes to losses of biological diversity and nitrate concentrations in drinking water may be a health hazard. Notable characteristics of these sources of pollution are (a) that some of the manure and slurry applications to land are designed as waste disposal systems, rather than as intended fertiliser; and (b) the sources of the waste are dispersed or 'non-point'. The former characteristic raises issues of policy integration - for example, manure disposed of to land results from restrictions on alternative means of disposal. The second characteristic also raises a complex policy issue, namely how to design a system of controls for pollutants the sources of which are hard to identify and measure. This paper is concerned with the second issue and, in particular, with the extent to which market-based approaches, such as taxes, could be used to control non-point pollution.

2. Nitrate and pesticide pollution in the UK

Table 1 summarises data on nitrate, phosphorus and pesticide pollution of freshwater in the UK. Average concentrations of phosphorus in surface waters fluctuate considerably, but the 2000 concentration is below that of 1980. Nitrate concentrations for surface waters have increased 1980-2000. Pesticides in both surface water and ground water have increased when measured in terms of the percentage of samples about specified concentrations. (Nitrate pollution of groundwater is not shown in Table 1 due to an ambiguity in the units of measurement). Approximately 35% of English rivers (by length) contain more than 30 $mgNO_3/l$ (roughly the European Union standard) and 60% contain more than 0.1 mgP/l. The corresponding percentages for Wales are 2% and 8%, and for N.Ireland, 0% and 27%. Overall, then, nutrient and pesticide pollution of freshwater has not shown improvement.

Table 1 **Freshwater pollution by nitrates, phosphorus and pesticides (Great Britain average).**

	1980	1990	1995	1999	2000
Surface water:					
Orthophosphates	0.40	0.64	0.50	0.40	0.33
(mg/l(P)	15.90	16.20	17.40	17.80	17.00
Nitrates (mg/l(NO_3)			9.90	14.80	
Pesticides[1]			3.50	3.40	
Pesticides[2]					
Groundwater:					
Pesticides[1]			19.10	35.40	
Pesticides[2]			17.30	31.60	

Source: www.defra.gov.uk/environment/statistics/des/index.htm

Notes: 1 - % of samples of Mecocrop, a typical pesticide, over 100 ng/l;
2 - % of samples of Mecocrop over 500 ng/l. Figures for phosphates and nitrates are averages for Great Britain.

3. The policy context: nitrates and pesticides

Policy on nutrient and pesticide pollution in the UK has three central features. The first consists of guidance issued by DEFRA (formerly MAFF) to farmers about 'good practice' in the management of these pollutants. The second involves a voluntary agreement between DEFRA and farmers with respect to pesticides. The third involves land zoning for 'Nitrate Sensitive Areas' (NSAs) and Nitrate Vulnerable Zones (NVZs). For discussion purposes, the first and second aspects can be treated as one.

3.1 Voluntary agreement on pesticides

The current voluntary agreement on pesticides emerged from an initial proposal in 1997 by the Labour Government to introduce a pesticides tax. Early formulations of such a tax were contained in a consultant's report (ECOTEC, 1998) and the Budget in 2000 entertained the real possibility of such a tax. The Crop Protection Association (CPA), acting for farming interests, expressed their opposition to such a tax and sought its replacement with voluntary measures. CPA’s own April 2000 proposal for self-regulation was revised in October 2000 in response to Government criticism that the proposals would not be sufficiently effective. The CPA proposals included information programmes, promulgation of best practice on disposing of waste pesticide and spraying procedures, and a separate agreement with the water industry covering a limited number of farms in sensitive water catchment areas. The revised package was also rejected by the Government. A third attempt to secure a voluntary package was made in 2001. This time a more pro-active stance was taken on biodiversity conservation, and the package included a proposal for crop protection management plans, along with detailed targets for pesticide reduction. Significantly, the package was developed with consultation from other parties, including environmental bodies who had been critical of the first two packages. Implementation would similarly be overseen by a multi-stakeholder committee. This third proposed package was welcomed by environmental groups. In June 2000, the then DETR commissioned a further consultancy report on the design of an effective voluntary agreement (EFTEC et al. 2002). This report commented on the likely effectiveness of the *second* set of CPA proposals and then set out its own proposals for a more effective voluntary agreement. 22 measures were proposed along with 85 actions, and 35 of the actions derive from the original CPA proposals.

The problem with voluntary approaches is that they have a high risk of not working, or, if they do work, they do so at unnecessarily high compliance cost. The literature on voluntary agreements is now extensive – see ten Brink (2002). The identified advantages of voluntary agreements can be summarized as follows:

(i) regulatory costs are avoided since the industry is in a better position to judge the appropriate actions and adjust them flexibly and in a targeted fashion. This could be important in industries where profit margins are low, or competitiveness is a vital concern, or where incomes are threatened, as with agriculture;
(ii) they can be devised so as to involve stakeholders, reducing the potential for conflict over regulation[1];
(iii) they generate better and more extensive information (learning) about environmental risks;
(iv) they may be easier and faster to implement than regulations.

However, there are considerable risks with voluntary agreements. These can be summarized as follows:

(i) they may not be effective because they lack mandatory targets and the credibility of the threat of strict regulation in such contexts may fade with time;
(ii) they may easily be 'captured' by polluters in their own interests;
(iii) there is a risk of free-riders if the industry agency implementing the measures does not have full coverage of polluters, and further free-riding may arise from parties within the agreement through non-compliance;
(iv) they may be used as a barrier to new entry into the industry;
(v) their effectiveness may be immeasurable due to the difficulties of defining the 'business as usual' baseline situation against which performance needs to be measured. It may similarly be difficult to measure performance relative to some alternative policy instrument.

Studies of the economic efficiency (cost minimisation) and environmental effectiveness (whether targets are met) of voluntary agreements suggest an inconclusive picture. The empirical evidence is consistent with the theoretical literature which shows that efficiency and effectiveness depends critically upon bargaining power, implicit allocations of property rights, the credibility and severity of the threat of alternative policy, commitment of the parties, stakeholder involvement and free-riding (e.g. see Segerson, 1998). The literature does suggest that agreements are likely to be less efficient than market-based instruments, but perhaps no worse in terms of environmental effectiveness. On the other hand, Dosi and Zeitouni (2000) are clear that existing voluntary approaches in the European Union for non-point pollution have not

[1] Interestingly, 'stakeholders' are usually defined in terms of those groups and institutions that can inhibit a regulation from being effective, whether those groups are elected or self-appointed. The general public often appears not to constitute a 'stakeholder group'.

brought about substantial reductions in pollution trends. The inconclusiveness of the empirical literature derives in part because most agreements are relatively recent, but in other cases there are problems of data, and of determining the baseline against which goals are to be assessed. In so far as the baseline can be determined, some agreements have done no better than the baseline and some have improved upon it. 'Soft benefits' are excluded from tests of efficiency and effectiveness, i.e. the extent to which agreements foster greater trust and understanding between stakeholders. Potentially offsetting this benefit is an analogous cost if the agreement is seen as a sign of 'weak' government. Probably the most useful outcome of the literature to date has been the development of guidelines for incentive design within agreements (e.g. EFTEC et al. 2002; OECD, 1999). In the event that consistent incentives could be guaranteed, then it is possible that voluntary agreements will perform well relative to other policy approaches.

In November 2002, The House of Commons Environmental Audit Committee issued a severely critical report on the voluntary initiative on pesticides (Environmental Audit Committee, 2002). They declared that the initiative lacked clear goals for reducing pesticide usage, was already a year behind in developing Crop Protection Management Plans, and that the actions added little or nothing to the baseline of what would have happened anyway given that other schemes also exist. They regarded the 30% take-up target for 2006 as ‘insufficiently challenging’ and that Government lacked criteria by which to judge whether the initiative was a success or not. They doubted that the voluntary agreement contained real incentives to change farmer behaviour, and that DEFRA’s approach to funding a Steering Group for the initiative was ‘miserly’. Finally, the Committee called for more investigations into the practical design of economic instruments. The Committee’s conclusions very much bear out economists’ doubts about voluntary agreements. To be effective, they have to contain carefully targeted incentives, and the pesticides agreement does not contain those.

3.2 Land zoning

Nitrate Sensitive Areas (NSAs) were introduced in 1990 in the UK in response to the European Union Drinking Water Directive (80/778/EC) which set a limit on nitrate concentrations in drinking water of 50 mg/l^{-1}. In NSAs, farmers are paid to adopt different land management schemes than those they would normally practise and which impose risks on surface and groundwater. Effectively, farmers are seen as possessing the property rights to traditional land management. Typical schemes involve converting arable land to unfertilised, ungrazed grass; converting to low input grass; and converting high input arable to low input arable.

In 1998, a new zoning scheme set up 66 Nitrate Vulnerable Zones (NVZs) in response to the EU Nitrate Directive of 1991 (91/676/EEC). NVZs are not solely concerned with health risks as is the case with NSAs, and hence the coverage is broader at some 8 per cent (600,000 ha) of England's land area. Farms in the NVZs are subject to controls, through an Action Programme on rates and timing of fertiliser application. In 2000, the European Court of Justice ruled that the UK was not in compliance with the Nitrate Directive and that further action was required. In 2001 DEFRA issued a consultation document (DEFRA, 2001) setting out only two options for compliance with the Nitrate Directive: (a) all farms in England would be designated as being within NVZs, or (b) 80% of England would be classified as within NVZs while designations in Wales and Scotland would be far lower. The former wider coverage option is estimated to have compliance costs of some £36 million, the latter £27 million. Offsetting savings in terms of improved use of nutrients make the net costs £32 million and £23 million (DEFRA, 2001). No full cost-benefit study was carried out.

There are two basic reasons why the zoning approach is used. First, NSAs and NVZs reflect a 'management agreement' approach to pollution control. This approach hinges critically upon the issue of who has the 'property rights' to agricultural land. If it is the public, then farmers should expect to face the implications of the 'polluter pays principle' and be regulated or taxed for generating negative externalities. But if the farmers have the property rights then it is for society to pay them not to undertake environmentally and health damaging activities. The management agreement approach tends to assume that farmers have the property rights, and hence they should be paid, at least in part, to participate in voluntary restrictions. On this basis, farmers inside NVZs are partially paid to comply. The payment is only partial because EU laws on State Aid forbid subsidies in contexts where there is a legal requirement to comply. Outside of NVZs, voluntary codes of good practice still apply.

The second reason for adopting the zoning/voluntary code of practice approach is because the pollution in question is diffuse in nature. Since it is not possible directly to measure rates of nitrate leaching farm by farm, it is not possible to identify precisely who the polluters are. The management approach works by appealing to, and negotiating best practice, without the regulatory burdens of direct controls, which would be untargeted blunt instruments.

But the problems with zoning as an efficient solution to the problem of nitrate pollution are several. First, there are technical issues relating to the extent of the land to be designated as a NVZ. For any given site in excess of the 50 mg/l limit, areas upstream need to be designated as NVZs. The question is how far upstream these designations should extend. Previously, the rule of thumb was that the designation would extend as far as points where the threshold is not breached. Under the new proposals, all upstream sources would be designated NVZs regardless of whether they exceed the threshold or not. There is a risk in this procedure that designations will 'over-comply' with the requirements, adding an excess cost burden to the regulation. The second problem builds on this: zoning may not be the minimum cost policy anyway, i.e. zoning is not being compared with alternative procedures. A third problem relates to the failure of the zoning policy to adopt an integrated approach. Areas with restricted nitrogen applications will face the potential of surplus manure stocks which will have to be transported to sites where there is a 'manure deficit' in the sense that manure applications to land will not exceed the nitrogen budget. The transportation process will itself create environmental problems, so that reduced nitrate in water is effectively being traded with increased noise, traffic emissions etc. In the same vein, many farmers accept sewage sludge for spreading on the land and it has been suggested that, faced with nitrogen problems, they may now refuse to allow sludge to be disposed of in this way. If so, this would effectively generate a further environmental problem of what to do with sewage sludge. A final issue is the role that can be played by any economic instrument - tax or tradable permit - if strict command and control standards are already in place. If zoning is designed to ensure that maximum permissible nitrate concentrations prevail at each site, then trading in nitrate 'credits' would not be permitted. Trading only works if some sources can go over a limit and some under. With geographically fine-tuned standards in place, there would be no incentive to trade. This is an instance of a more general problem of how to superimpose market based instruments on top of existing legislation (Pearce, 2002).

In light of these problems, market-based approaches to nutrient and pesticide pollution are worth close investigation. Since the Nitrate Directive is law, the UK has little option but to comply as best they can through the zoning procedure, but it is possible that a market-based approach could be integrated with some form of zoning. The essence of the market-based approaches is that they attempt to overcome the problem of non-point pollution by tackling the inputs directly. We first review the theory underlying choice of policy instrument in the non-point pollution context, and then look at the experience with some market-based approaches.

4. The theory of non-point pollution control

Pollution is essentially an economic problem. Pollution involves risks to health, nuisance, disamenity and loss of biological diversity. All these impacts constitute losses in human wellbeing and constitute 'externalities'. There are also costs of reducing pollution, in terms of monitoring, treating polluted effluents, cleaning up after pollution accidents and to polluters in terms of forgone output or profits. These costs are not equally borne by those affected, largely because pollution affects different individuals and groups in different ways. Pollution control is, thus, also a matter of equity and social justice as well as economic efficiency. The private costs of an activity, such as applying a fertiliser, are those incurred by the farmer. To these may then be added the total of external costs that occur because of the activity, for example the extra costs to water consumers and any costs of ill health. This results in the true inclusive social cost of the activity. The consequences are illustrated in Figure 1.

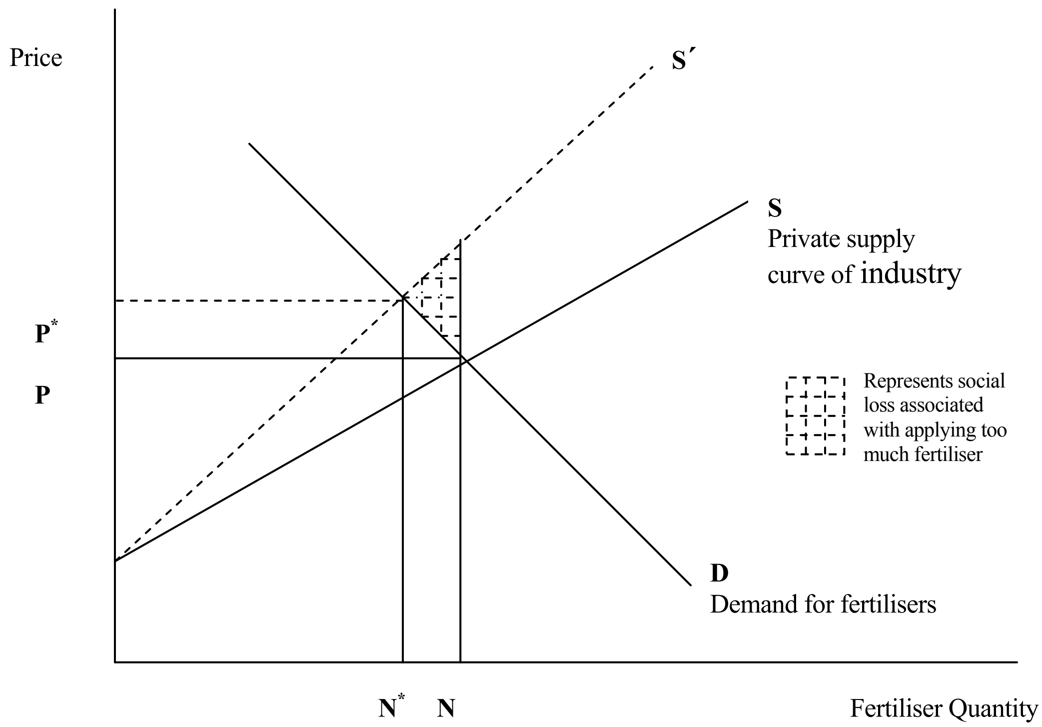

Figure 1: The impact of the inclusion of social costs of fertiliser use on the supply and demand relationship.

Line S represents a private supply curve of the fertiliser industry, and line D the demand for fertilisers from farmers. At price P the industry sells N amount of fertiliser – the supply is equal to the demand. But if external costs are added to produce a true social cost the price at any given quantity will increase. The new supply curve is then S′ and the socially desirable outcome is now N^*, where the demand corresponds to the social, not private, costs. The difference between N^* and N represents the excess of fertiliser used because the current market price, P, does not account for the external costs of fertiliser pollution.

There are several market-based approaches that have the potential to address agricultural pollution. An indirect method is persuasion combined with technical assistance to facilitate changes in behaviour. A more direct stimulus is setting product, design or environmental performance standards to which farmers must comply. The aim is to affect farmers' choices of inputs and production and pollution control practices towards the socially optimal ones. An example of such direct regulation is pesticide registration, which restricts pesticides available to farmers and sets conditions of use.

Alternatively, farmers' behaviour can be influenced through the use of economic incentives. Major options are taxes or liability for damages to discourage environmentally harmful activities, subsidies to encourage pro-environment behaviour, tradable permits to ration environmentally harmful activities, and contracts in which environmental authorities purchase specified pro-environmental actions ('paying for ecological services').

4.1 Input tax/subsidy compared to run-off tax/subsidy and ambient tax/subsidy schemes

A number of hydrological process and statistical models have been developed in order to overcome the difficulty of measuring pollution flows from non-point sources. These can provide the policy maker with information on the relationship between production choices and emissions, which can be used to construct the economically efficient input tax/subsidy scheme. We briefly describe Griffin and Bromley's (1983) model, which can be used to design optimal instruments, based on estimated emissions and input use. In this model polluters are risk-neutral, profit maximizers, and cannot collectively influence input and output prices. The objective of the policy-maker, as indicated in Figure 1, is to minimize the social costs of pollution (i.e. the sum of private costs and external costs), or equivalently to maximize the difference between the expected benefits of polluting activities and the expected costs of the resulting pollution. Net social benefits to be maximized are given in the equation below:

$$SNB = \sum_{i=1}^{n} \pi_{Ni}(x_i) + \sum_{k=1}^{p} \pi_{Pk}(e_k) - D[a(r_1,\ldots,r_n,e_1,\ldots,e_s,\zeta,\psi)]$$
$$s.t.\ W(a) \le T$$

where $\pi_{Ni}(x_i)$ is the *ith* farm's expected profit, restricted on emissions, for any choice of inputs x_i from non-point source production (emissions measured using proxies); $\pi_{Pk}(e_k)$ is the *kth* farm's expected profit, restricted on emissions, for any choice of inputs e_k from point source production (emissions measured with certainty); *D* is the environmental damage cost; a_i (.) is the *ith* farm's site specific characteristics (also called the fate and transport function); $r_i(x_i, a_i)$ is the *ith* farm's non-point emissions; ζ represents the natural background levels of pollution; ψ

represents watershed characteristics and parameters; and *W(a)* is the performance measure indicating a target *(T)* defining the maximum level of acceptable ambient degradation.

The solution of this problem requires that two conditions be satisfied:[2,3] (a) the marginal gain in profits from the use of any input on any farm must equal the marginal environmental opportunity costs; and (b) the marginal gain in profits from emissions must equal the marginal external cost of emissions. Given the externality problem described in Figure 1, condition 2 will not be satisfied without intervention. If *input-based tax instruments* are used, then farmers will maximize after-tax profits by equating marginal gain in pre-tax profits from the use of each input with the corresponding input tax rates, $\partial \pi_{Ni} / \partial x_{ij} = \tau_{ij}$.[4] If *a farm-specific tax-based estimated runoff* is used as an instrument, then farmers will maximize after-tax profits by equating the marginal gains in pre-tax profits from the use of the input, with the marginal cost of the increased tax payment, $\partial \pi_{Ni} / \partial x_{ij} = \tau_j \partial r_i / \partial x_{ij}$.[5]

The literature on non-point agricultural pollution sources recognises, although rarely develops relevant applied models, that some additional parameters should be taken into account when constructing economic instruments for agricultural non-point pollution. These include: (a) the stochastic nature of emissions and imperfect information about the fate and transport of pollutants and (b) asymmetry of information between the regulatory agency and polluters about polluters' control costs.

In effect, the case-specific importance of these parameters indicates the most cost efficient instrument to be used in each particular situation.[6] In a deterministic model the choice of base is unimportant (Griffin and Bromley, 1983). This does not hold when runoff is stochastic (Shortle and Dunn, 1986). In particular, both input-based instruments, as well as instruments based on estimated run-off, provide farmers with incentives to consider how their choices affect their expected profit and will lead farmers, as already indicated above, to the maximization of after-tax profit. However, input choices do not only alter expected profits. They also affect the variance and skewness of profit. The variance of profit is directly proportional to the Arrow-Pratt measure of risk aversion, while the skewness of profit is directly proportional to the measure of downside risk aversion. Both these measures of the Arrow-Pratt and downside risk aversion are used in the calculation of the risk premium. That is, if the farmer is risk averse, this premium corresponds to the percentage of his/her profit the farmer is willing to forgo (pay) in order to avoid the risk emerging from the use of each particular input in his/her production.

2 Although the static analysis to follow offers many insights relevant for the dynamic case, additional policy implications can come from a dynamic analysis, with the most significant being how policies optimally evolve over time to ensure cost-effective rates of investment in pollution control equipment and environmental improvements (see, Xepapadeas, 1991, 1992, 1994; Kim et al. , 1993; Dosi and Moretto, 1993, 1994; Tomasi et al., 1994).

3 For the sake of simplicity our analysis does not consider entry/exit effects of policy instruments. However, since these policy instruments affect producers' profits they subsequently affect their decision to remain, enter or exit the regulated market. These considerations entail additional instruments that are designed to influence entry and exit without distorting input choices; e.g. a lump-sum tax to the extra-marginal producers (which ensures that they are better-off when they do not produce) and a lump-sum subsidy to the marginal producer whose decision to produce is adversely influenced by the magnitude of cost-effective taxes.

4 Alternatively, the cost-effective allocation may be attained using farm-specific input standards that limit the use of pollution-increasing inputs to no more than their optimal levels and require the use of pollution-control inputs at no less than their optimal levels.

5 Alternatively, for an estimated runoff standard, farms maximise profit subject to estimated runoff being restricted at or below a target level.

6 Given restricted information on costs and benefits from environmental quality degradation, cost-effectiveness is the popular criterion adopted in instrument design.

Hence, although input-based and estimated run-off based instruments are potentially able to achieve the optimal use of agricultural inputs in a deterministic framework, that cannot do so in a stochastic framework as they cannot capture higher moments of profits which are affected by input choice and also affect the welfare (utility) of risk-averse farmers operating in a uncertain/stochastic framework. However, this is a real world aspect of the problem at hand, which should be taken into account if intelligent agricultural policy is to be designed and implement (see Groom et al., 2002).

Another important difference between input-based instruments and estimated runoff-based instruments - also related to the issue of stochasticity and imperfect information about the fate and transport of pollutants - is that the former allow for differential targeting of inputs whereas the latter do not. Differential targeting of inputs enables better fine tuning of input risk effects. On the other hand, however, estimated runoff instruments have the advantage of transmitting more site-specific information to producers about their environmental pressures, relative to input-based instruments.

When asymmetry of information between the regulatory agency and polluters about polluters' control costs exists, producer responses to various instruments cannot be predicted accurately. As a result, it becomes impossible to design tax/subsidy incentives that will exactly satisfy the optimal solution of the externality problem. Although input tax/subsidy schemes and contractual arrangements that can elicit farmers' specialised knowledge have been described in the literature, there is little empirical literature on the performance of alternative agricultural pollution control instruments under conditions of public uncertainty about polluters' control costs.

Ambient tax/subsidy schemes involve paying farm-specific subsidies when the ambient pollution concentration falls below a target and charges firm-specific taxes when the ambient concentration exceeds the target. These schemes shift monitoring from the source of emission to the receptor. The same problems that limit the cost-effectiveness of estimated runoff-based incentives limit the cost-effectiveness of ambient-based incentives (Horan et al., 1998). Ambient tax schemes, however, have substantial appeal compared with input tax schemes as: (1) they do not need to devise farm-specific policies (except from the lump-sum charges to induce optimal entry and exit); and (2) if point sources exist in the relevant region, they optimally co-ordinate control of point and agricultural sources without the need to devise source-specific policies.

Additional limitations on the effectiveness of ambient taxes are introduced by the following factors. First, strategic considerations complicate the design of these instruments as rewards and penalties depend on groups' performance. Secondly, the shift of the burden of information from regulators to producers might not be optimal given the limited technical information and capacity of the typical producer. This is again a problem relevant to imperfect and asymmetric information. Thirdly, ambient taxes cannot produce a first-best outcome when polluters are risk-averse (Horan et al., 1999). Such considerations have led Weersink et al. (1998) to suggest that ambient taxes are more effective when applied to small and well-monitored watersheds, with homogeneous farms and small time lags between polluting activity and degradation impacts.

4.2 Liability rules

Under liability rules, victims of externalities have the right to sue for damages in a court of law. There exist two important classes of liability rules. The first one is the strict liability rule, under which polluters are held liable for full payment of any damages that occur. Strict liability rules are similar to (non-linear) ambient taxes. The extent of a farmer's liability depends on the damages that arise as a result of ambient pollution level (which is a function of site-specific characteristics, e.g. soil type and topography) and his beliefs regarding the probability that he will be sued (which itself depends on site-specific characteristics and a vector of random variables that may influence this probability). The second set of liability rules concern negligence: polluters are only liable if they fail to act with the 'due standard of care' (Segerson, 1995). This standard is defined in terms of either damages or input use.

As far as economic performance is concerned, both strict liability and negligence are limited in all the ways in which ambient-based incentives are. Moreover, uncertainty about the success of the litigation process (depending on the characteristics of agricultural pollution), together with the high expenses involved, reduces the effectiveness of this instrument. As argued by Horan and Shortle (2001), liability rules are probably best suited to the control of pollution related to the use of hazardous materials or non-frequent occurrences such as accidental chemical spills.

4.3 Point / non-point trading

The main attraction of trading schemes is their potential to achieve the environmental goal at a lower social cost than 'command and control' instruments. These schemes involve each polluter receiving a number of pollution permits that specify allowable emissions for the permit owner. Permits are transferable and can be traded among owners; thus a market for pollution is created and relevant property rights are introduced. Farmers with high (low) marginal pollution control costs will purchase (sell) permits from farmers with low (high) marginal costs and as a result emit less (more). The end result is that the maximum total allowable level of pollution is met at lower cost than if trading was not allowed (Baumol and Oates, 1988; Handley et al., 1997).

This is the textbook version of trading schemes. Unfortunately the reality for non-point emissions is not that straightforward. The difficulty in accurately monitoring them (at reasonable cost) and their stochastic nature complicates the design of the appropriate trading schemes for non-pollution in agriculture. Point / non- point systems that have been developed to date involve point sources trading increases in emissions for reductions in estimated loadings from non-point sources. Point sources are given (auctioned or purchase) the permits and agricultural non-point sources voluntarily enter the commitment to 'sell' additional allowances of permits to point sources (in actuality to be compensated for their reduction in loadings). As a result some control responsibility is transferred to non-point sources as well. Alternatives to trading mean loadings are trading inputs that are correlated with pollution flows, trading reductions of cropland in fertilisers-intensive uses. Theoretical research has demonstrated that emissions-for-inputs trading systems can be designed to provide greater economic efficiency (transactions cost aside) than emissions-for-estimated loadings trading-schemes because they are better able to manage the variability of non-point loads (Horan and Shortle, 2001).

Another relevant issue in the design of trading schemes is the rate at which non-point allowances are traded for point source allowances (Shortle, 1987). This rate should not be one to one as non-point inputs and estimated loadings are imperfect substitutes for point source emissions; rather the rate should be a function of risk and relative contributions to ambient pollution. Faeth (2000), the last volume of an interesting series aiming at providing answers to the mandates of the 1972 US Clean Water Act,[7] argues in favour of a program where point-point and point/non-point trading would have a shared responsibility to undertake remediation actions not coupled to point source regulatory requirements. This, he argues could take one of several forms. A first option could be that agricultural conservation subsidies, or some share of them, could become part of the pool of funds available for a joint trading program. Farmers, municipalities or industrial sources who generated credits could sell them through a single program sponsored by the government or another broker in conjunction with point sources who wishes to purchase and apply credits. Credits purchased by government conservation funds would be retired.

As a second option, farmers could only generate credits after they have met a minimum standard for agricultural practices. For example, farmers who currently had sound manure management practices that were below standards to ones well above standard could generate a partial credit. While such a program requirement would be entirely voluntary, it would have the benefit of rewarding farmers who undertake sound management on their own, and would still provide an incentive for farmers who had not yet made the change It would have the disadvantage, however, of reducing cost-effectiveness.

Finally, state agencies could apply a performance requirement to all sources, including agriculture, and allow point sources and farmers the option of meeting the requirement though trading. Point and non-point sources would have access to the same pool of credits, and conservation subsidy funds could be applied as before to offset some part of farmers' costs or the cost of operating the program. Farmers who had previously undertaken conservation practices would be in compliance and have no further obligation. Others who could make inexpensive reductions would do so and perhaps do more than their obligation. Some farmers with high costs or high-valued crops who wished to continue their current practices could purchase credits from point or non-point sources to meet their obligation.

While trading has economic potential, there are some uncertainties associated with trading that need to be acknowledged and accounted for. The first and perhaps most important aspect of trading that would involve non-point sources is that there is a great deal of uncertainty involved because the loads are tied to weather events. While point sources produce fairly regular flows across seasons and even years, non-point sources do not. Loads are highest during rainy seasons and years with high precipitation, and conversely lower at other times. For this reason, a reduction in the load from non-point source may not be equivalent to that from a point source. Therefore, it is important that water quality is monitored to make sure that expected improvements are realised and water quality goals are met.

[7] In *Fertile Ground: Nutrient Trading's Potential to Cost-Effectively Improve Water Quality*, Paul Faeth, develops a framework to assess the cost-effectiveness of various policies and combinations of policies to reduce phosphorous loads in specific watersheds and argues that policy approaches incorporating nutrient trading programs are dramatically less expensive than conventional approaches and can achieve comparable benefits. In *Growing Green: Enhancing the Economic and Environmental Performance of U.S. Agriculture*, Faeth and the team he managed integrated voluminous amounts of data into an analytic framework that assessed the profitability and environmental impacts of alternative cropping systems. In *Agricultural Policy and Sustainability: Case Studies from India, China, the Philippines, and the United States*, Faeth and several co-authors found that farm policies are usually stacked against resource-conserving farming methods.

Another consideration is that trading programs can be expensive to put in place and operate if poorly designed. Regulatory paperwork, information gathering and the process of identifying partners to trade with can create transaction costs that are prohibitive and make a trading program ineffectual. Administrative oversight needs to be sufficient to ensure good performance, but not so burdensome as to inhibit trading. Registration of trades should be efficient so that, partners can easily hook up, report their trades, and get approval. When numerous non-point sources are involved, some sort of broker –for example, a co-operative - needs to be organised to co-ordinate the sale of credits and to verify them using standard techniques. For all these reasons, trading should occur within a regulatory program where rules and methods are standardised and appropriate review can be cost-effective, not permit-by-permit, which is expensive.

4.4 Some additional theoretical remarks

Although we focus on incentives, the empirical literature has shown that there are instances in which standards might be preferred (e.g. Abrahams and Shortle, 2000). Briefly, these instances are: (1) cases of extremely hazardous pesticides with less risky substitutes; (2) cases when techniques exist that have the potential to yield significant environmental gains to farmers with little or no cost (e.g. use of septic systems for domestic wastewater treatment in rural areas; (3) cases in which a high degree of certainty over the level and geographical distribution of polluting inputs is desirable (e.g. pollutants for which damages are potentially large or irreversible, such as toxic substances). Moreover, it is worth mentioning that our analysis adopts the unrealistic assumption that agricultural commodity and input markets are not distorted. The widespread presence of agricultural subsidies in OECD countries is sufficient to show that markets are distorted. As such, first-best rules are no longer optimal and second-best rules, which take into account the effects of policy designs on costs of market distortions, are needed.

5 Experience with market-based approaches: pesticide taxes

There is fairly extensive experience with pesticide taxes in OECD countries. Table 2 summarises these taxes. Table 3 shows estimates of the price elasticity of 'demand' for pesticides. Demand here is taken to refer to either the quantity of pesticides used or the frequency of their use[8].

The problems with designing pesticide taxes are several. First, Table 3 confirms the impression that the price elasticities of demand for pesticides are low[9]. Hence taxes will do little to reduce pesticide use unless they are set at very high rates (relative to the pesticide price). Both Norwegian and Swedish official reviews of effectiveness reached this conclusion, although both also agreed that it is difficult to disentangle tax effects from other policy effects (Andersen et al. 2001). This suggests that the effectiveness of taxes rests on the uses of the tax revenues. The Danish experience suggests that recycling revenues back into agriculture severely reduces the effectiveness of the tax, and this explains the switch of policy in 1998 compared to the 1996 tax

[8] The two will tend to be related but some of the literature reports impacts on frequency of use and some of it on total quantity used.

[9] The exception appears to be Gren's analysis for Sweden (Gren, 1994) but she notes the fact that her estimates are higher than other Swedish studies. van den Bergh et al. (1997) report a high value for one of the studies they review but do not say which one it is.

where revenues were return to agriculture as reductions in land taxation. Using revenues to further research or encourage changes in farming practice would appear to make more sense. Damage from pesticides is cumulative so that current damage is partly a function of past pesticides releases. This will be especially true of water contamination by pesticides. If revenues can be hypothecated, they can be used for groundwater clean-up programmes, so that revenue-raising taxes nonetheless have an externality reducing function.

Table 2 **Pesticide taxes**

Country	Year	Tax rate	Revenues	Comment
Denmark	1982	20% P+T		Pesticides in small volumes
	1989	3% P+T		Pesticides
	1996	27% P+T	Recycled to reduce land taxes	Insecticides
		13% P+T		Herb/fungicides
		3% P+T		Preservatives
	1998	35% P+T	Recycled to organic farming + pesticide reduction	Insecticides
		25% P+T		Herb/fungicides
Finland	1988	2.5% V		To cover administration fees
	1998	3.0% V		
Iceland	1998	38 € tonne		
Norway	1988	2.0% P		
	1989	3.0% P	NOK 20m	
	1990	17.0% P	Nearly all revenues recycled	Tax + regulatory fee
	1991	19.0% P		
	1996	22.5% P		
	1998	24.5% P		
	1999	1.4 NOK/ ha		Regulatory fee
		0-150 NOK/normal dose/ha		Damage-varying tax
Sweden	1984	4SK/kg/ai		Env'mental. Tax
	1988	8SK/kg/ai		Env'mental. Tax
	1994	20SK/kg/ai		Env'mental. Tax
	1986	29SK/ha/dose		Regulatory tax
	1990	38SK/ha/dose		Regulatory tax
	1991	46SK/ha/dose		Regulatory tax
	1992	29SK/ha/dose		Regulatory tax
	1992	0SK/ha/dose	Regulatory tax phased out	

Source: adapted from Andersen et al. 2001.
Notes: P = price, T = tax. Thus, 20% P+T means the tax was set at 20% of the tax-inclusive price, or 25% of the tax-exclusive price. V is value added. NOK = Norwegian krone. Ai = active ingredient. SK = Swedish krone. Ha/dose = one normal dose per hectare = 'hectare dose'.

Table 3 **Price elasticities of demand and use for pesticides**

Country	Study	Elasticity	Comment
Denmark	Dubgaard 1987	60% -> -21% F 120% -> - 43% F -0.07 Q -0.61 Q -0.35 Q	 Herbicides/barley Fungicides/barley Fungicides/wheat
Denmark	Rude 1992	60% -> -13/-19% F 120% -> -20/ -28%F	Updates Dubgaard
Denmark	Jensen and Stryg 1996	-0.33 Q	
Sweden	Swedish Board of Agriculture 1992	-0.20 to -0.50 Q	Herbicides
	Gren 1994	-0.93 -0.52 -0.39	Herbicides Insecticides Fungicides
Meta-analysis	Van den Bergh et al. 1997	-0.05 to - 1.53	Various
UK	Bailey and Rapsomanikis 1999	-0.39	
	ECOTEC 1997	-0.55 to -0.71[1] -0.28 to -0.45[2]	
France	Carpentier 1994	-0.30	Specialised arable farms
Netherlands	Oskam 1995 (quoted)	-0.12	

Notes: elasticities are to be read as follows: 60% ->-21% F means that a 60% tax (% of pesticide price) gives rise to a 21% reduction in frequency of use. Q refers to quantity and the figure quoted is therefore a conventional price elasticity of demand. 1: estimating using ECOTEC methodology I. 2: estimate using ECOTEC methodology 2.

Second, Table 2 shows that most taxes have been designed as percentage of pesticide prices. The risk here is that technological progress in pesticide manufacturing can give rise to price falls, and consequently absolute tax reductions, encouraging more pesticide use. Taxes per unit active ingredient can also fail to approximate differential environmental and health impacts (Dubgaard, 1991; Archer and Shogren, 2001). The theoretical solution here is to express the tax as an absolute sum per unit of toxicity-weighted ingredient. Securing the toxicity weights is potentially feasible through the use of health-risk coefficients (Bolt, 2000) and health or ecological risk coefficients (Archer and Shogren, 2001). In practice, capturing the 'true' marginal damage from different forms of pesticide is complex due to other factors affecting damage - e.g. ground and weather conditions, ecosystem variation, and so on. But two studies suggest it is possible to secure estimates of differentiated tax rates. Bolt's study sets a tax equal to:

$$t = b.\left(E.\frac{1}{L}.\frac{1}{k}.P\right).V$$

where b is a slope factor for a dose-response function linking exposure to chemicals to the probability of cancer; E is the estimated daily intake of an individual averaged over a lifetime of exposure; L is average lifetime; k is kilograms of pesticide use; P is population at risk and V is the monetary value of a statistical life. The resulting tax rates vary from 7 pence per kilo of Trifluralin to over £59 per kilogram for Amitraz. The most sophisticated toxicity-weighted tax study is that of Archer and Shogren (2001). They show results for a flat-rate tax for herbicides; a tax that is then differentiated by information about health risks from herbicides reaching water supplies; a tax that is even further differentiated according to the probabilities that herbicides will reach water supplies; and tax that is finally differentiated according to all these factors plus information about tillage practices. Unlike Bolt's study, however, there is no estimate of actual damage, so that the variation in tax rates is calibrated on a benchmark tax of $1 per (US) pound of one herbicide, Atrazine. However, the effects of the postulated taxes are shown in terms of monetary changes in farming returns per acre and in terms of changes in environmental indicators of water quality. This permits trade-offs to be shown between reduced net returns and improved environmental quality.

Third, there are few examples of *actual* taxes being differentiated by toxicity. The Norwegian reforms of 1999 appear to come closest to this. Even though the overall demand for pesticides is not reduced significantly by a tax, a toxicity-differentiated tax may be effective if substitution between pesticides will occur in such a way that the overall toxic impact of pesticides will be reduced. In short, pesticide use and toxicity could be 'decoupled' by a pesticide tax. The problem with pesticide tax studies is that few of them simulate the 'cross price effects' of such a policy, i.e. they do not look closely at substitution between types of pesticides (or between pesticides and other inputs such as fertilisers and land). Bailey and Rapsomanikis (1999) simulate such a toxicity-weighted tax for the UK in the period 1992-1998. Overall price elasticity of demand for pesticides was consistently low and never greater than –0.39. However, cross price elasticities between the 'banded' pesticides (banded according to toxicity) were greater than the 'own' price elasticities, suggesting that farmers might switch between types of pesticide. But faced with a large tax on highly toxic pesticides, taxes on medium to low toxicity pesticides have a greater effect, suggesting that the high toxicity pesticides are more 'necessary' for agricultural production. A toxicity-weighted tax may not therefore achieve much by way of reduction in high toxicity pesticides. Moreover, Bailey and Rapsomanikis suggest that significant pesticides taxes could be welfare-reducing unless the size of the externality associated with pesticides is above some 40,000 Euro per tonne. The only available estimate of damage based on a willingness to pay study, also for the UK, is of damage equal to 20,000 Euro per tonne (Foster et al, 1998), i.e. only half that required to justify a toxicity-weighted tax in economic terms. The importance of substitution to high-concentration low-dose pesticides is also stressed in Andersen et al. (2001) in evaluating the taxes in Finland and Norway. Substantial reductions occur in the tonnage of pesticides in both countries in response to various policy measures, including the taxes, but Andersen et al. doubt that this is associated with significant reductions in environmental and health damage.

Three central conclusions can be derived from documented experience with pesticide taxes. First, that actual taxes may have had little impact on reducing pesticide damage, even if they have helped to reduce pesticide use. This is due to the low elasticities of demand and the behaviour of farmers in switching to low dose higher toxicity pesticides - i.e. it is important to know both overall price elasticities and cross elasticities of demand. Second, that toxicity-weighted taxes can be computed. This is evidenced by the work of Bolt for the UK but especially so by the work of Archer and Shogren in the US. Fears that complex tax structures would be administratively difficult to operate may be well-founded but it would be useful to review the Norwegian experience following their switch to a toxicity-weighted tax structure in 1999. Third, it is important to assess cross-elasticities as well as own price elasticities for aggregate pesticides or individual pesticide types.

6 Experience with market-based approaches: fertiliser taxes

Table 4 lists fertiliser taxes that are in place or which have been introduced in the recent past, while Table 5 reports estimates of fertiliser price elasticities. Fertiliser taxes should have two effects. First, fertilisers can be expected to be over-used due to risk aversion among farmers. This means that farmers will prefer to over-use fertiliser rather than under-use them, the latter option being associated with risks of unacceptable crop yield reductions. Hence a tax should reduce fertiliser use without giving rise to yield reductions, and especially so where other technologies are available for replacing artificial fertilisers (e.g. leguminous crops). Second, even if fertilisers are used optimally from the standpoint of the farmer's interests, crop yield reductions may be justified as the price to be paid for reducing environmental externalities.

It is important to note that the Netherlands and Denmark are developing detailed 'mineral accounts' for each farm in the country. A mineral account records the application of nitrogen to soils through fertilisers and animal manure, the net take-up of nitrogen by crops, and hence the net excess balance. The net balance is effectively the run-off of nitrates from the farm. To a some extent, then, the underlying problem of non-point pollution - namely the difficulty of allocating ambient pollution to sources - is overcome.

Rougoor et al. (2001) analyse the effects of the taxes in Sweden, Austria and Finland. In the case of Austria, the levy is thought to have had a significant 'signalling' effect through raising awareness that fertilisers are environmentally damaging. In Sweden, it is estimated that the tax reduced demand for fertilisers in 1991-2 by 15-20%. Jonsson et al. (1997) estimate that the nitrogen tax reduced financially optimal dosages by about 10 per cent. They suggest indirect effects through the use of recycled revenues to fund research etc. was more significant. However, recycling of revenues ended in 1994 when the charge became an official 'tax'. The Finnish experience is limited to a period of just two years since before 1992 the tax was set at a very low level and it was abolished altogether in 1994. The effect in 1992 was significant but there was a growth in set-aside land at the same time. It is suggested by Rougoor et al. that the resulting net price elasticity was low at -0.11. However, this needs to be interpreted as a very short-run elasticity and long run elasticities would certainly have been higher. Hansen (1999) briefly evaluates the Danish experience and suggests the nitrogen tax (which covers fertilisers and manure) will help to solve regional nitrate problems, but that local problems will require more careful controls on animal stocking rates.

The experience of those countries that have introduced fertiliser taxes is that they appear to have played some role in reducing fertiliser use[10]. However, the price elasticity estimates are low and this suggests comparatively little effect in terms of quantity reductions. There is some suggestion that revenue recycling may have been more effective, with revenues redirected to research and information.

In terms of environmental effectiveness there is a problem similar to that for pesticides in terms of the tax being proportional to damage done. Whereas pesticides vary in their toxicity by design and also according to the conditions in the receiving environment, fertiliser damage tends to vary mainly because of the receiving environment conditions. On the face of it this suggests that flat rate fertiliser taxes should be more efficient than flat rate pesticide taxes which, as we have seen, need to be toxicity-weighted. Nonetheless, flat rate fertiliser taxes may still be inefficient. Brännlund and Kristrom (1999) simulate desirable fertiliser taxes for Sweden and show that the rates should vary regionally from 0.45 SK/kgN to 3.15 SK/kgN. Even this differentiation, they suggest, hardly captures the geographic variation in damages. Hence there is a problem of correctly targeting the tax. Brännlund and Kristrom's analysis appears to be the only one available making some allowance for differential ecological impact. In principle, however, it should be possible to differentiate a tax according to nitrate sensitivity indicators. The obvious problem is how the resulting arbitrage between regions would be avoided, i.e. fertilisers being bought in low tax areas and transported to high tax areas.

[10] On the comparative politics of introducing fertiliser taxes in Denmark and Sweden see Daugbjerg (2000). The former made several attempts prior to 1998 to introduce such a tax, without success.

Table 4 Fertiliser taxes

Country	Year of tax	Tax rate	Revenues	Comments
Austria	1986-1994	1986:€0.25 kgN 1988:€0.35 kgN 1991:€0.47 kgN	ASch 1 billion 1991	Used to subsidise grain exports. Abolished on entry to EU
Norway	1988	1996: NOK1.2/kg/N NOK2.3/kg/P	NOK 165m 1996	
Sweden[11]	1985	1985: €0.12 kgN 1986: €0.14 kgN 1987: €0.16 kgN 1988: €0.20 kgN 1991: €0.24 kgN 1992: €0.27 kgN 1993: €0.07 kgN 1995: €0.20 kgN (1992: SK 5kg/P)	SK 300m 1996	Combined environmental and regulation tax, latter abolished in 1992. Reclassified as a tax in 1994 without revenue recycling
Belgium	1991	Levied on N and P content of manure	€3.3 m 1993	Applies in Flemish part of Belgium
Netherlands		<125kg/P/ha tax = 0 125-200 kg/P/ha: tax = 0.25Gl/kg. >200 kg/P/ha tax = 0.50 Gl	Gl 35m 1993	Based on P production per ha
Denmark	1998	5 DK/kgN		Levied on 'excess' N if mineral accounts kept. Otherwise on all N. N ceilings set by government
Finland	1976 - 1994	1976-1992 small 1992 €0.44 kgN 1990 €0.25kgP	€ 45 m 1994	Aimed at export subsidies. Abolished on entry to EU

Sources: Nordic Council of Ministers (1996); Hansen (1999); OECD (1995);Rougoor et al. (2001); Brannlund and Gren (1999)

[11] Detailed schedules of the Swedish taxes are given in Jonsson et al. (1997).

Table 5 Price elasticities for fertiliser demand

Country	Study	Price elasticity	Comment
Austria	Rougoor et al. 2001	-0.20	
	Becker 1992[1]	-0.29	
Sweden	Drake 1991[2]	-0.17 to - 0.25	
	Jonsson et al 1997	-0.12 to - 0.51	
	Brännlund and Gren 1999	-0.12 to -0.51	Range covers different regions
Finland	Bäckman 1997	-0.11	Very short run elasticity - see text
Netherlands	Oskam 1995	-0.43	
Various	Burrell 1989	-0.10 to -1.10	

Notes: 1 in German, quoted in Rougoor et al. (2001). 2 in Swedish, quoted in Rougoor et al. (2001).

7 Tax policies in the context of farm income support

Most studies that evaluate the effectiveness of pesticide and nitrogen tax policies either say nothing about the farm income support context in agriculture, or assume it is given and that it has no interaction with tax measures. As such, they share the same problem noted earlier, namely that first-best policies are not feasible. An exception is Shortle and Laughland (1994) in the context of the USA. Their argument is that any tax would have implications for farm support policies - hence policy co-ordination would be needed between environmental tax and output subsidy measures. The net effects depend on the way any subsidies are adjusted as the tax is imposed. Adjustments might involve raising subsidies to compensate farmers, or the subsidy regime might be changed to one of lump sum transfers based on acreage. Dosi and Zeitouni (2000) similarly express the view that groundwater protection needs to be integrated into reform of the Common Agricultural Policy. The main message is that the net welfare effects of environmental taxes in the agricultural sector cannot be estimated properly without modelling the interactions between such measures and farm income support policy - a message about 'joined up' government.

8 Tradable permits in mineral surpluses

Where mineral accounts exist - i.e. where it is possible to trace inputs and outputs of specific polluting minerals such as nitrogen - the conditions lend themselves to trading. The Netherlands permits trading in surplus animal manure. Farmers with below-limits manure allowances can sell to farmers already at the limit of their application rates. There is also very limited experience in the USA relating to point-non-point trading for phosphorus in certain States (Horan and Shortle, 2001; US EPA 2001). Schemes exist for the Tar Pimlico estuary in North Carolina, the Dillon Creek Reservoir in Colorado and Cherry Creek, Colorado. Young and Karkoski (2000) describe a scheme in the San Joaquin Valley, California where farmers trade land drainage allowances. However, in this particular case the amount of drainage load is monitorable due to the prevailing system of pipes and canals. They note that the scheme was introduced in 1994 and that within five years pollution loads were substantially reduced.

The obvious problem with trading regimes is what non-point sources would trade with point sources. The schemes in question involve point sources trading increased emissions with estimates of reduced loadings from agriculture. Agricultural sources enter the trade on a voluntary basis and with compensation for abatement efforts they can demonstrate they have made. In effect, then, monitorable emissions are traded for expected values of reduced loadings, the sources being stochastic. Trading therefore tends to take place on the basis of non-unitary ratios, i.e. one unit change in point emissions is traded for more than one unit expected value of non-point emissions. At the moment there appear to be no trading systems in which point source trade emissions with *inputs* (fertilisers, pesticides) to non-point sources. Shortle and Abler (1997) suggest that trading point emissions for non-point expected value emissions will be less efficient than trading point emissions for non-point emission reductions.

US EPA (2001) expresses a mix of optimism and pessimism for future trading prospects in water loadings in general, the optimism being based on the potential for cost savings, the pessimism being based on the limited potential for trading if technology-based standards are already in place and the complexity of designing trading systems.

9 Conclusions

Both pesticide and fertiliser taxes could play some role in addressing the problem of non-point agricultural pollution in the UK. The main problems that need to be addressed can be summarized as follows. First, the low elasticities of demand for pesticides and fertilisers suggest that taxes will have little direct effect on reducing demand. This is of course a problem common to other UK environmental taxes such as the landfill tax and the aggregates tax. The experience of other countries suggests that such taxes may nonetheless have a 'signalling' effect in reducing demand, especially if farmers fear future rises. Possibly more important is the indirect effects of reducing demand via information and research activities financed by hypothecated taxes. In other words, there will be a high income effect which generates revenues that could be hypothecated to well-targeted environmental programmes.

Second, the need, in both cases, to capture as far as possible the geographic and product variation in damage. For pesticides the product variation should be capable of being captured through toxicity weighted taxes, i.e. taxes would vary by commercial pesticide product. A single flat rate tax would appear to be inefficient, although it might be justified via its revenue raising effects. However, the limited work available suggests it is also very important to model cross-substitution between pesticides which face different tax rates. Little information appears to be available on this issue but what there is, is instructive. Geographical variation in ecosystem sensitivity to pesticides is probably not capturable in a pesticide tax. This problem is common to fertiliser taxes: for fertilisers, the main cause of geographic variation in damage is ecosystem sensitivity. Beyond one experiment for Sweden, there appears to be no work comparable to that on pesticides which investigates the potential for geographically varying taxes. Since the tax is an input tax, any variation would in any event set up an arbitrage market, blunting the point of the tax. Hence a fertiliser tax would in all probability be blunter instrument than a toxicity-differentiated pesticides tax.

Third, the extent to which, even if there are problems with these taxes, they perform better or worse than the alternatives. The main alternatives in the UK, and almost everywhere else, are some form of voluntary agreement and land zoning to cover application rates. Both can be judged inefficient and the theoretical literature suggests they are more inefficient than market-based approaches. Fourth, the practical scope for taxes given that they would have to be superimposed on the increasingly complex legislative structures governing pesticides and, especially, nitrates. The issue is whether these instruments can induce 'beyond best practice' emissions. Finally, the possibility that some form of trading could take place with respect to manure loadings (and sewage sludge). A detailed life cycle assessment would be required to avoid a situation where one environment problem is resolved by trading at the expense of creating another problem. Trading regimes appear to exist in the Netherlands for surplus manure, but their effectiveness is not known. Trading nutrient loadings also exists in limited form in some states of the USA.

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