ANALYSIS

Pesticide taxation and multi-objective policy-making: farm modelling to evaluate profit/environment trade-offs

Katherine Falconer a,*, Ian Hodge b

a Centre for Rural Economy, Department of Agricultural Economics and Food Marketing, University of Newcastle upon Tyne, Newcastle upon Tyne, NE1 7RU, UK
b Department of Land Economy, University of Cambridge, Cambridge CB3 9EP, UK

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Abstract

Many countries in Western Europe have introduced voluntary programmes to encourage farmers to adopt environmentally more benign practices such as integrated pest management, but more policy action appears to be needed to meet the environmental quality levels now demanded. Input taxes could assist in meeting policy objectives. The issue considered here is the identification of the most appropriate specification of a tax instrument to reduce the environmental problems of agricultural pesticide usage. This paper takes a farm systems approach to evaluation. A case-study illustration is given for a specialist arable farm in the UK, combining an economic model of land use and production with a set of environmental indicators for pesticides. Linking these two components allows the identification of the potential trade-offs between achieving reductions in the environmental burden to a number of ecological dimensions and farm income. Different pesticide tax specifications vary in both the magnitude and the direction of their impacts. The results of the model indicate that either compromises will have to be made in environmental policy, or additional instruments will be required to counter-act the negative side-effects of some instruments. © 2001 Elsevier Science B.V. All rights reserved.

Keywords: Pesticides; Policy evaluation; Environmental indicators; Economic instruments

1. Introduction

The Common Agricultural Policy was established with the aim of increasing production efficiency and self-sufficiency in food production in the EU. A broad range of environmental problems have been attributed to the achievement of its policy goals over recent decades, relating, for
example, to environmental contamination by agro-chemicals, habitat destruction, biodiversity reduction and undesirable landscape change (Skinner et al., 1997). In particular, there are widespread and growing concerns related to the levels of use of chemical pesticides (Reus et al., 1994; World Wide Fund for Nature, 1995; Freeman and Bontin, 1995; McLaughlin and Mineau, 1996). Agricultural pesticides are integral components of modern crop production systems. Unfortunately, however, it seems almost impossible to prevent chemicals that are deliberately introduced into the environment for crop protection from entering and diffusing through the environment. An obvious solution would be to reduce the quantities applied and to use chemicals of lower eco-toxicity, but a balance needs to be struck between greater environmental protection and the continued contribution of agriculture to production.

The fifth European Environmental Action Plan established a reduction in chemical use as a major objective, although no actual goals or limits have been defined and currently member-states are largely free to address their own priorities. The complexity of crop protection and the generally superior information available to the farmer compared to the regulator, on the relative costs of alternative abatement strategies implies that environmental controls should aim to retain as much farm-level flexibility as possible. Policies involving the use of the market mechanism are potentially able to do this. However, although the environmental economics literature suggests that economic incentives may have efficiency advantages compared to command-and-control approaches, their applications have, as yet, covered only a limited range of issues. Only a few Northern European countries have experimented with ad valorem pesticide taxes or volume levies so far, although there is growing interest in the refinement and extension of incentive instruments. For example, in recent years, the UK government has been assessing the options for input pesticide taxation (Department of the Environment, 1997).

Environmental economics can only really make a practical contribution to agri-environmental resource management if linked with more explicit acknowledgement of the real-world characteristics of environmental problems. The heterogeneity of pesticide inputs and the complexity of agro-ecological systems adds an important dimension to environmental policy-making; in this respect, pesticide contamination differs significantly from other agricultural pollution issues such as nitrate contamination.

This paper examines how an analytical modelling approach linking economic and ecological components could assist in achieving acceptable and workable environmental policies. A case-study assessment of the applicability of economic-incentive-based policy options to reduce pesticide contamination is presented. Section 2 discusses the need for multi-dimensionality in policy evaluation and outlines the production ecology approach. Section 3 introduces the case study and Section 4 presents some policy evaluation results. Section 5 discusses the findings, and Section 6 concludes the paper.

2. Environmental complexity and policy multi-dimensionality

Many of today’s most intractable environmental problems are those for which the effects of natural physical and chemical processes are highly uncertain in terms of their extent, degree and duration. The potentially widespread environmental contamination by agricultural pesticides is an obvious example of this. Furthermore, the multi-dimensionality of ecological problems is a critical consideration in the practical policy-making context. Given a complex system of cause and effect, the separate treatment of individual aspects of environmental quality problems by fragmented institutions and policies is unlikely to be appropriate. In particular, given linkages between system components, any intervention is likely to have some unintended effects. Changes need to be evaluated in a wide-ranging way. For example, the interactions between different inputs to agricultural production, such as pesticides and fertiliser, mean that their usage should be considered jointly rather than separately.

Agricultural policy now encompasses a number of aims, and trade-offs between them are in-
evitable. Hence it is important to establish the indirect effects of instruments on other policy objectives and the linkages between instruments, and as part of the policy-making process to exploit complementarities and to avoid conflict or overlap in the achievement of objectives to whatever degree possible. At issue, therefore, is the nature and degree of overlap of the impacts of different agri-environmental policies: identification and measurement of these would provide useful information for real-world policy-making, given multiple objectives and the need to address these simultaneously. Policy development based on promoting the complementarities between schemes and their objectives could permit simpler, more efficient and more practicable frameworks composed of fewer policies.\(^1\) In contrast, implementing piecemeal policies in an inter-dependent economy may lead to an overall loss of efficiency (see Runge and Myers, 1985; Milon, 1987; Braden and Segerson, 1993; Vatn, 1995).

Many of the immediate environmental impacts of agriculture occur at a local level initially, so the analysis here focuses on the details of farming systems. The micro-economic theory of production and consumption provides a useful framework for analysis to shed light on fundamental, firm-level problems of resource allocation. Policy problems are envisaged in terms of alternative allocations of resources, the objective being to find the pattern of production that will contribute most to goal achievement. A high proportion of farm-level agri-environmental policy studies apply optimisation analysis and programming techniques, such as linear programming (LP) (see, for example, Hazell and Norton, 1986). Mathematical programming approaches have important advantages over econometric models using past data where the policy requiring analysis has no historical precedent or if, for any other reason, projections from past data are not possible.

Central to the analysis is a study of the quantity and mix of commodities which farmers could produce, and how (Barnard and Nix, 1979). Any production planning problem can be divided in general terms into several elements (Hazell and Norton, 1986): a set of (quantifiable) objectives; a range of possible enterprises/activities; and a set of limited resource supplies and other constraints. Production functions are assumed to consist of discrete combinations of inputs and outputs, each characterised by a discrete coefficient. With the application of price data, the incorporation of resource availability constraints, and an objective function (or functions), optimal production plans can then be derived.

The optimal plan is that combination of activities that best fulfils the objectives of the farmer and is feasible in terms of the constraints. Optimality can be assessed by reference to the law of equi-marginal returns: if limited resources are not allocated in such a way that the marginal return to those resources is the same in all the alternative uses to which they might be put, profits could be increased by transferring resources, based on the notion of opportunity cost.

The traditional economic optimisation approach can be developed and refined for more useful economic-ecological analysis. De Koeijer et al. (1995) argued that economic analysis should draw on production-ecological knowledge (such as agronomy field trials or environmental monitoring data). Simple ‘production economics’ models provide information about the private costs of policy implementation only, but it is likely that there will be trade-offs between farm and environmental objectives (or policy would not be needed in the first place). Ideally, a framework for policy evaluation would integrate an economic model of farm production with an environmental model. The ‘joint production’ of agricultural outputs and environmental externalities, and the possibility of varying the proportion of different outputs (crops and risks) for different combinations of inputs, provides a useful conceptual framework for analysis when trade-offs are of interest. Both purchased input levels and other resource use should be regarded as the technology sets required to achieve specified output levels. Hence, a systems approach is advocated, i.e. more wide-ranging consideration of the relevant inputs and outputs.

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\(^1\) However, Önal et al. (1998), for example, found that it often becomes difficult to identify workable solutions to land management problems when more than one water quality problem must be addressed.
The next section provides an application of the production ecology methodology to examine economic instruments to reduce the environmental consequences of pesticide usage.

3. Evaluating the environmental choices in taxation options

3.1. The empirical economic model

The research aim was to connect economic farm modelling with ecological models designed to assess the environmental consequences of production activities involving pesticide usage. There have been relatively few pesticide-specific policy evaluations in the literature (see though Oskam et al., 1992; Wossink et al., 1992; Bauer et al., 1995). Furthermore, there have been relatively few studies evaluating the trade-offs between the achievement of different environmental policy goals, rather than just the usual trade-offs between farm income and a single specified objective such as nitrate emissions reductions.

The micro-level is appropriate for primary policy analysis because it is at the level of the individual farm that the actual decisions are made about cropping patterns, production intensity and so on. An East Anglian mainly-cereals farm with combinable break crops was chosen as a case study, using data from Murphy (1995). Cereals account for a significant percentage of arable land in the UK and other European member states and form a key part of most rotations. East Anglia accounts for 40% of the arable crop area in England and Wales, and is also characterised by the greatest intensity of pesticide usage in the UK (Garthwaite et al., 1995), particularly of those chemicals most frequently detected in water resources and in breach of the drinking Water Directive (Drinking Water Inspectorate, 1994). The total rotational area of the farm was set at 250 ha.2 Full details are given in Falconer (1997).

Two versions of the farm model were developed. One model was calibrated to represent current commercial crop production (CONV) and the other was calibrated to represent a more ‘progressive’ farm (PROG), with a wider variety of cropping activities ranging between current commercial practice (CCP) and reduced input rates (50% of the CCP levels or lower) for all pesticide and nitrogen regimes. It is very important to include production functions representing different production options for each crop, for example, ranging from ‘intensive’ (CCP) to ‘ecological’ practices, to reflect the different options available to farmers with as detailed a specification of resource use as possible. However, crop protection is very complex and only limited data were available, so the number of activities considered was restricted to five crop protection input regimes (covering herbicides, fungicides and insecticides) and two nitrogen regimes per crop. The low-input activities for the PROG model were calibrated on the basis of farm trials data collected in East Anglia between 1991 and 1996, which were used to calculate coefficients for yield and variable costs relative to CCP crops. Estimated yields and variable costs could then be obtained for low-input crops without having to use season- or situation-specific data. Numbers of herbicide, fungicide and insecticide spray units (where one unit is one per-hectare application at the labelled dose rate) were assigned to each crop activity on the basis of this information and survey data from Garthwaite et al. (1995). However, the coefficients for the low-input practices were of course, only provisional estimates of the potential differences in yields and variable costs compared to CCP, given the very limited trial data that was available. Yields would be expected to vary over seasons, soil types, rotations, and field operations. Table 1 shows the coefficients estimated for winter wheat as an example.

The model allowed output data to be generated with regard to the crop land allocation; the total farm management and investment income; the total farm spray usage in terms of units, and a measure of the potential environmental impacts of pesticide uses. GAMS (General Algebraic Modelling System) (Brooke et al., 1992) programming
Table 1  
Coefficients for winter wheat (year 1), percentages of current commercial practice, per hectare

<table>
<thead>
<tr>
<th>Current commercial practice</th>
<th>Low pesticide inputs</th>
<th>Low herbicide inputs (other chemicals held constant)</th>
<th>Low fungicide inputs (other chemicals held constant)</th>
<th>Low insecticide inputs (other chemicals held constant)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Yield, full nitrogen level</td>
<td>100</td>
<td>97.64</td>
<td>100</td>
<td>98.40</td>
</tr>
<tr>
<td>Yield, half-nitrogen level</td>
<td>92.15</td>
<td>89.66</td>
<td>92.41</td>
<td>91.49</td>
</tr>
<tr>
<td>Herbicide cost</td>
<td>100</td>
<td>79.65</td>
<td>79.65</td>
<td>100</td>
</tr>
<tr>
<td>Fungicide cost</td>
<td>100</td>
<td>53.79</td>
<td>100</td>
<td>53.79</td>
</tr>
<tr>
<td>Insecticide cost</td>
<td>100</td>
<td>0</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Total chemical inputs, expressed as ‘units’ per ha</td>
<td>6</td>
<td>2</td>
<td>4.5</td>
<td>4.5</td>
</tr>
</tbody>
</table>

* One unit = one per-hectare dose at the recommended rate.

was used to solve the model. The baseline optimal farm plan (i.e. the ‘policy-off’ scenario) for the conventional model was validated using the average crop hectarages for farms of this type and was found to be satisfactory. The total cereal area for the CONV optimal plan matched published estimates from Murphy (1995) for cereal farms, excluding potatoes and sugar beet, reasonably well. The oilseed rape area was relatively high, but probably because no other break crops were grown. The optimal plan for the PROG model included a smaller area of cereals and a larger area of break crops, but gave an equally plausible rotation. The baseline higher management and investment income (MII) achieved by the PROG model compared to CONV suggested that farm incomes could be improved by adopting less-intensive farming practices. This finding is supported by other work, for example, in the Netherlands by Oskam et al. (1992).

3.2. The environmental model: Hazard indicators for pesticides

Ideally, the optimal level of pesticide contamination control would be determined by equating the marginal social benefits of usage with its marginal costs. However, the ecological impacts from pesticide usage cannot be monetarised easily, so alternative approaches are needed. Analysis is further hindered by the fact that clearly quantified relationships between pesticide use and environmental quality are not yet available. The nature of pesticide impacts on the environment following field application depends on a number of factors, such as the amount of pesticide active ingredient applied, the toxicity of the pesticide, its persistence, local environmental characteristics and species presence and vulnerability. Still, some way must be found of assessing at least the direction of likely changes in environmental quality resulting from changes in the optimal farm plan following policy implementation.

The term ‘pesticide’ covers a large and very heterogeneous group of chemicals, in terms of their properties and usage and in terms of their potential for adverse environmental effect. The attributes of individual pesticide chemicals are likely to have a major bearing on the real reductions (or other changes) in terms of pesticide usage and impacts. A fundamental problem is the heterogeneity of pesticides: the diversity of inputs and their effects means that there are no clear targets against which environmental quality improvements might be measured. A variety of broad-ranging indicators are needed for comprehensive environmental assessment. Hence, a set of multi-dimensional summary measures, which can
Table 2
Crop hazard scores, points per hectare

<table>
<thead>
<tr>
<th></th>
<th>Current commercial practice</th>
<th>Low pesticide inputs (other chemicals held constant)</th>
<th>Low herbicide inputs (other chemicals held constant)</th>
<th>Low fungicide inputs (other chemicals held constant)</th>
<th>Low insecticide inputs (other chemicals held constant)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter wheat</td>
<td>25.3</td>
<td>12.2</td>
<td>20.7</td>
<td>18.9</td>
<td>16.7</td>
</tr>
<tr>
<td>Winter barley</td>
<td>14.0</td>
<td>12.3</td>
<td>13.8</td>
<td>12.5</td>
<td>14.0</td>
</tr>
<tr>
<td>Spring wheat</td>
<td>18.0</td>
<td>13.5</td>
<td>14.5</td>
<td>17.0</td>
<td>18.0</td>
</tr>
<tr>
<td>Spring barley</td>
<td>13.7</td>
<td>11.3</td>
<td>13.2</td>
<td>11.8</td>
<td>13.7</td>
</tr>
<tr>
<td>Winter oilseed rape</td>
<td>20.3</td>
<td>14.7</td>
<td>19.5</td>
<td>17.8</td>
<td>18.0</td>
</tr>
<tr>
<td>Spring oilseed rape</td>
<td>19.3</td>
<td>16.1</td>
<td>17.3</td>
<td>19.3</td>
<td>19.3</td>
</tr>
<tr>
<td>Winter field beans</td>
<td>15.0</td>
<td>10.7</td>
<td>12.3</td>
<td>13.3</td>
<td>15.0</td>
</tr>
<tr>
<td>Spring field beans</td>
<td>18.0</td>
<td>9.5</td>
<td>12.8</td>
<td>14.7</td>
<td>16.0</td>
</tr>
<tr>
<td>Peas</td>
<td>18.7</td>
<td>13.6</td>
<td>9.0</td>
<td>9.0</td>
<td>9.0</td>
</tr>
</tbody>
</table>

be condensed into a single index using scoring and weighting, could be useful in suggesting where cropping practice changes could be made for greatest environmental benefit.

It is important to draw the distinction between hazard and impact: impact depends on both hazard and exposure to it. The latter, however, is very difficult to measure or even estimate for most species. Very few data are available on non-target species exposure, or the actual impacts of pesticides on them. As a result, the focus so far has tended to be on hazard, as implied by parameters such as the toxicity or lethal dose rates for non-target organisms, or physico-chemical properties such as water solubility or the soil adsorption coefficient. Similarly, the environmental indicators and summary scores developed here will focus on pesticide hazards.

The chosen methodology used a simple scoring and aggregating approach, similar to that developed and applied by Kovach et al. (1992), for example. The information on statutory product labels summarised in Ivens (1994) was used as a basis for hazard assessment. The environmental and special precautions are supposed to take into account all the available information on pesticide risks and impacts that is used in the approvals process. Nine different ecological and human-health dimensions were identified on an ad hoc basis from scanning all product labels listed in Ivens (1994), i.e. bees, fish, water quality, game and wildlife, skin, eyes, ingestion, inhalation and hazards from organo-phosphorous/anti-cholinesterase compounds. In addition, a tenth ‘general’ dimension was defined on the basis that all pesticides are thought to present some ecological risk by virtue of their nature as biocides.

Scores were given for each hazard dimension according to labelled warnings (such as ‘harmful to fish’ or ‘irritating to eyes’). These scores were then aggregated so all pesticides received a score from 1 to 10, higher scores indicating that the pesticide poses a greater hazard. Simple arithmetic aggregation was used in the absence of any information indicating that an alternative weighting should be used. Table 2 gives hazard scores for the crops in the model.

The disaggregation of the ecological hazards from pesticide usage along different dimensions allows identification of the nature and extent of trade-offs between changes in hazards following changes in crop practices and land use. Trade-offs were expected in the absence of correlations between the environmental and health hazards of the various chemicals potentially used on different crops. Hence it would be impossible to gauge the relative environmental cost-effectiveness of different policies by examining only the changes in the farm-level costs.
This type of indicator, as developed above, has important advantages in terms of its coverage, transparency, adjustability in the light of new information, and the use of available data. It also includes all aspects covered in the approval/registration assessment process, and provides readily-accessible information for products rather than simply the active ingredients (which is useful since mixtures sold as products can otherwise be problematic to evaluate). However, there may well be omissions in the environmental dimensions considered here (for example, groundwater or air quality, of which no mention was made on product labels), and the labelled precautions reflect the nature of the approvals process at the time of the last review of the product. Precautions might be out of date and in need of revision. Furthermore, the yardstick is linear, and takes no account of indirect effects caused by interactions between different dimensions; neither is there any incorporation of environmental or other management efforts (such as the use of pesticide-exclusion strips along watercourses), which may reduce pesticide risks and impacts. Ideally the yardstick should be tailored for specific soil and climatic circumstances too. While the imperfections are acknowledged, at present there are no alternatives, so the yardstick provides at least a starting point for discussion.

3.3. Production ecological modelling for instrument appraisal

The next stage of the analysis involves linking the environmental indices described above to the normative economic model for policy evaluation. A pesticide usage strategy was defined for each arable activity, based on survey data from Garthwaite et al. (1995) relating to the number of units used of each chemical. Activity hazard scores were calculated using this information and the product hazard scores. A total score per farm could then be calculated once the cropland allocation was known. Table 3 summarises the baseline scores for the two farm systems. The overall farm resource allocation model, including both the economic component and the environmental hazard indicators, is summarised in Fig. 1.

A key policy design issue is how closely an economic-incentive instrument aimed at a particular action can be linked to the environmental impacts resulting from the action. The policy targeted on the indicator closest to the policy goal should be most efficient. However, the costs of intervention must be considered; limits on the administrative capacities of government may form a significant constraint on policy-making and implementation. Input volume is a relatively low transactions-cost proxy for environmental burden, and hence a starting point at least for policy design. However, reducing the total volume of pesticide usage may not be the most efficient means of securing reductions in risks to health and the environment given an imperfect correlation between kilograms of active ingredient and environmental burden.

Four economic incentive policy instruments were selected for evaluation: an ad valorem tax (TAX); a fixed levy per spray unit (UNIT), where one unit is one standard (recommended) per-hectare spray application; a per kilogram active ingredient levy (KG); and a levy based on the pesticide hazard score of each individual product (HAZ). Policy instruments were chosen to reflect

| Total baseline farm scores for individual pesticide hazard dimensions |
|-------------------------|-------------------------|
| CONV farm               | PROG farm               |
| Hazard to bees          | 225.0                   | 100.0                   |
| Hazard to skin          | 525.0                   | 218.8                   |
| Hazard following ingestion | 350.0                   | 256.3                   |
| Hazard to eyes          | 387.5                   | 247.5                   |
| Hazard to water quality | 400.0                   | 162.5                   |
| Organo-phosphate warning | 175.0                   | 50                      |
| Hazard to fish and aquatic organisms | 976.6     | 190.5                   |
| Hazard from inhalation  | 62.5                    | 6.2                     |
| Hazard to game and wildlife | 125.0                 | 0                       |
| General hazard          | 1187.5                  | 543.8                   |

3 Note, though, that these systems are only two points along a wide spectrum of agricultural production options.

4 See, for example, Falconer and Whitby (1999).
max \sum_{c} \sum_{f} \sum_{reg} (p_{c,f,reg} \cdot Y_{c,f,reg} - VC_{c,f,reg} \cdot X_{c,f,reg}) \cdot \{Cults_{c,f,reg} \cdot X_{c,f,reg}\} - PERM - OVERLAB - CASUAL - RENT - OVERHEADS

subject to

\sum_{c} \sum_{f} \sum_{reg} X_{c,f,reg} \leq TRA

\sum_{c} \sum_{f} \sum_{reg} X_{\text{rotation}} \leq 15\% TRA

\sum_{c} \sum_{f} \sum_{reg} ROTLIMS \leq ROTCONS

YS_{\text{chem}} = \sum_{d} POINTS_{d}

SCORE_{c,f,reg} = \sum_{c} \sum_{f} \sum_{reg} (YS_{\text{chem}} \cdot CHEMUNIT_{\text{chem}})

YARDSTICK = \sum_{c} \sum_{f} \sum_{reg} (YS_{c,f,reg} \cdot X_{c,f,reg})

UNITS_{farm} = \sum_{c} \sum_{f} \sum_{reg} (UNITS_{c,f,reg} \cdot X_{c,f,reg})

\sum_{c} \sum_{f} \sum_{reg} (X_{c,f,reg} \cdot CULTSREQS_{m,c,f,reg}) \leq (TRACT \cdot MACH_{m})

\sum_{c} \sum_{f} \sum_{reg} (X_{c,f,reg} \cdot HARVREQS_{m,c,f,reg}) \leq HARV_{m}

\sum_{c} \sum_{f} \sum_{reg} (CULTSREQS_{m,c,f,reg} \cdot X_{c,f,reg}) \leq ((MEN \cdot REG_{m}) + (MEN \cdot OVERTIME_{m}) + CASUAL_{m})

\sum_{c} \sum_{f} \sum_{reg} (X_{c,f,reg} \cdot NETFLOW_{q}) + BALANCE_{q-1} \leq OVERDRAFT

where c = crop, f = fertiliser regime, reg = pesticide regime, p = price, y = yield, VC = variable costs, CULTS = cultivations and harvesting costs, X = crop hectares, PERM = permanent labour costs, MEN = number of permanent workers, OVERLAB = overtime labour costs, CASUAL = casual labour costs, RENT = rent, OVERHEADS = other fixed overheads, TRA = total rotational area, ROTLIMS = rotational limits (crop hectares), ROTCONS = rotational constraints, d = hazard dimension, POINTS = hazard score per dimension; SCORE = hazard score per crop-hectare, CHEMUNIT = number of units used per-crop hectare of each chemical; YS = hazard score per chemical; YARDSTICK = total farm hazard score UNITS = spray units, CULTSREQS = tractor time requirements, MACH = tractor hours available, TRACT = number of tractors, HARVREQS = combine time requirements, HARV = combine time available, REG = regular labour hours available for field work, OVERTIME = overtime hours available, CASUAL = casual labour hired, NETFLOW = net cash flow, BALANCE = current account balance, OVERDRAFT = overdraft; m = month; chem = pesticide chemical; q = quarter of the year.

Fig. 1. Summary of the farm-level resource allocation model.
the objectives of current or proposed policy and with immediate relevance to the debate concerning pesticide minimisation as well as wider relevance to the environmental policy discourse. There are important differences between the different economic instruments. TAX places a greater penalty on usage of the more expensive chemicals; HAZ is akin to an ad valorem tax, but involves price increases differentiated across chemicals according to the hazard score; KG penalises pesticides with higher volume applications. UNIT takes account of neither of these aspects, but addresses intensity of pesticide use on a per-hectare basis. The instruments were incorporated into the model through appropriate pesticide cost adjustments. Table 4 gives the ranges evaluated for each instrument. Upper levels were set at those at which farm income fell to zero and land began to be retired voluntarily. All instruments had critical levels below which no change in the optimal farm plan was observed (i.e. the same outcome was observed as in the policy-off scenario); the analysis below relates to levels of tax or levy above these thresholds. Judging the critical value of any instrument is very important: implementation of a tax or levy at a rate below this will simply raise revenue and reduce cash farm income, without achieving any real effects on farm resource allocation (in the short term, at least).

Interest lay in the changes in tax-neutral farm MII (i.e. MII without the deduction of the tax liability)\(^5\) relative to changes in the farm’s pesticide hazard scores, both total and for individual hazard dimensions. Changes in tax-neutral MII were also evaluated against changes in fertiliser expenditures (as a proxy for physical usage and nitrate leaching). The fertiliser criterion was of interest in terms of policy evaluation from a multi-dimensional perspective: if a pesticide reduction policy achieves its goals to the detriment of other environmental aspects (such as nitrate leaching), it is important to have information on these trade-offs. Economic theory predicts that if the usage of either nitrogen or crop protection chemicals is proscribed, usage of the other should also fall, giving rise to a positive environmental side-effect (see Weinschenk and Dabbert, 1987; Berg, 1984; Schulte, 1983).

There are a number of well-known limitations to LP, and therefore the model developed here. LP is a partial equilibrium approach only, with a single objective function. Ideally, risk attitudes and the relative riskiness of different cropping activities would have been modelled too, but data were inadequate for this. Furthermore, the models used here relate to single periods only: ideally, a recursive, or dynamic optimisation, approach would have been used. Another limitation stems from the small range of technologies assumed to be available for any given cropping activity. Understanding the input substitution possibilities is central to the analysis here of the environmental problems of pesticides: policy impacts should differ depending on the compliance options available to producers. There is a complex surface of input combinations that a farmer may take account of when developing or adjusting his farming system, but LP takes this into account discretely rather than continuously. Thus, unless a large number of

<table>
<thead>
<tr>
<th>Instrument Description</th>
<th>CONV Critical levels</th>
<th>PROG Critical levels</th>
</tr>
</thead>
<tbody>
<tr>
<td>TAX on spray expenditures</td>
<td>10%–200% (10% increments)</td>
<td>102.5%</td>
</tr>
<tr>
<td>UNIT levy per spray unit</td>
<td>£5–£85 (£5 increments)</td>
<td>£25.5</td>
</tr>
<tr>
<td>KG levy per kg active ingredient used</td>
<td>£5–£55 (£5 increments)</td>
<td>£14</td>
</tr>
<tr>
<td>HAZ levy per hazard point</td>
<td>£1–£15 (£1 increments)</td>
<td>£1.8</td>
</tr>
</tbody>
</table>

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\(^5\) The tax payments made by the farmer are, of course, simply transfers; interest here lay in the economic effect rather than the financial effect of policy. Transfers could, of course, be paid back to farmers, for example, as a lump sum payment.
activities are included in the analysis, input substitutions may be masked by product substitutions.

The model relates to the short term only; no allowance could be made for the gradual development of the farm plan in the absence of knowledge of future technological change. Furthermore, indicators such as those developed here cannot address specific environmental problems or particular situations, and in most cases they should be supplemented with other, case-specific information to yield maximum benefit in policy appraisal. Caution is advocated when using indicators in new situations very different to those under which parameter values were estimated as values may not be stable across different contexts. However, despite its limitations, the modelling approach linked to environmental indicators here was still considered useful for examining the potential consequences of policy implementation.

4. Results

The empirical work presented used the farm-level models described above to investigate how pesticides might be reduced most cost effectively, at the farm level. The underlying hypothesis was that the cost effectiveness of any one instrument at reducing hazard along different ecological dimensions will vary. Trade-offs might be expected between improvements made to different dimensions, and also between reductions in pesticide and chemical-fertiliser usage. Two-dimensional frontier analysis, involving a simple economic dominance criterion, was applied to distinguish between the outcomes of the various policy instruments. This approach allowed the trade-offs in farm plans between farm income and indicators of environmental burden under different policy scenarios (instrument type and level) to be presented visually. Frontier analysis can assist in the assessment of whether simply reducing the volume of pesticides applied is sufficient to reduce environmental burdens, or whether other policy goals, such as changing the specific type of chemicals used, might perhaps be more effective.

Higher income/pesticide usage (or pesticide hazard) frontiers are preferable, as these indicate that a given policy variable level can be achieved with as high a level of (tax-neutral) farm management and investment income (MII) as possible. A farm plan arising under a given policy scenario is dominant when no other plan can obtain the same or lower spray usage (or level of another policy indicator) with the same or greater returns. Backward-bending frontiers (i.e. with negative gradients) indicate that not only is farm income falling, but the policy indicator value (for example, total farm hazard score) is simultaneously increasing. An important question was whether the rankings of the four charges and levies differed depending on the policy instrument used, or whether one instrument consistently outperformed the others. That is, assuming equal social preference weightings for different types of environmental change, does any one policy achieve better results for one indicator or more with no-worse results for all other indicators of interest?

A concern was that some instruments might be ineffective or even counter-productive to some degree. Policy rankings were found to vary to some degree according to the goal to be achieved, such as reductions in spray expenditure, physical usage and environmental hazard, which complicates practical policy-making. As expected, HAZ was found to be consistently best at reducing the overall farm hazard score with least impact on farm income (see Figs. 2 and 3). UNIT was found to be best at reducing spray unit usage when evaluated using the CONV model, although in the PROG model both HAZ and UNIT appeared to be equally good (with respect to farm MII) at reducing the volume of pesticide applied on the farm. UNIT performed relatively well on balance, which is encouraging as this may also have other attractive features such as transparency and administrative feasibility.

A different policy ranking emerged when evaluating policies on the basis of fertiliser expenditure.

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6 However, if movement is in the other direction, such frontiers would indicate that environmental quality and income increase simultaneously.
reductions rather than pesticide usage reductions, so there are important questions to be answered about social preferences for the relative achievement of different environmental goals, and the permissible trade-offs. Generally, the correlation between fertiliser expenditure and measures of pesticide usage was positive, although a negative relationship was observed following the implementation of some instruments, with policy-making implications in terms of resolving conflicts between the achievement of policy goals for pesticide and fertiliser usage. At low levels of TAX, a slight increase in spray units was observed, and a decrease in fertiliser expenditure for the CONV farm model. Other policy instruments were found to result in decreased levels of both variables as the levy rates increased. TAX was observed to result in decreased levels of both variables as the levy rates rose, while the total farm spray units fell.

The heterogeneity of pesticide products in terms of their adverse environmental potential means that it is important to look beyond aggregated farm-level expenditure and usage reductions and check that positive change is being made in relation to policy sub-goals. Interest lies in the achievement of hazard reductions across a range of ecological dimensions, rather than just making improvements to some dimensions at the expense of others. Figs. 2 and 3 have summarised the trade-offs between total farm-level pesticide hazard and farm income; it is necessary now to examine trade-offs between income and the individual hazard dimensions themselves. The focus was placed on the ecological hazard dimensions rather than the on-farm hazards to health, which would be expected to be ‘internalised’, for example, by wearing protective clothing.

Table 5 summarises the relative performance of the different instruments in terms of their impacts on individual ecological and health hazards. Few trade-offs between hazard reductions for different dimensions were observed for the CONV farm. However, some negative consequences were observed for the PROG model, for example, in terms of hazards from organo-phosphates. Generally, the observed negative effects related more to human health rather than ecological dimensions. Nevertheless, it appears that a pesticide reduction policy based on an ad valorem tax will not necessarily produce desirable changes in usage. It is also encouraging that the frontiers produced un-

![Fig. 2. Scenario frontiers for total farm hazard scores.](image-url)
der the UNIT and HAZ policy scenarios were reasonably similar, given the arguably simpler tax base of the former instrument. Interest also lay in the relative extent to which individual dimensions were improved. Another approach to instrument evaluation and comparison involved the use of a summary index of cost-effectiveness (CE) to indicate the ‘elasticity’ of trade-offs between farm income and policy indicators, both of which are contingent upon the farm-plans resulting under different policy scenarios. This method facilitates evaluation along multiple dimensions of interest, providing different information to the evaluations above.

A unitless cost-effectiveness ratio was calculated using percentage changes in any given dimension X as compared to changes in farm income, rather than absolute changes from the baseline (the policy-off situation). A CE score of 1 indicates unit elasticity, i.e. a 1% change in the criterion value is accompanied by a 1% change in real farm income; a value of 2 would indicate a 2% change in the criterion accompanied by a 1% change in farm income. A larger ratio indicates a more cost-effective policy. However, in the absence of knowledge of social preferences for these changes, it is impossible to identify desirable thresholds of CE for policy instruments to achieve; still, the analysis is useful for comparing policies. A negative ratio indicates that policy implementation gives rise to adverse effects, such as an increase in total hazard, even if other parameters, such as spray expenditure or usage have fallen.

Generally CE ratios for each instrument peaked soon after its critical threshold, and then fell (given rising marginal abatement costs as input use is reduced). However, a range of CE ratios was exhibited for the different dimensions under any given policy instrument, indicating that it is easier to achieve reductions in hazard with regard to some dimensions than others. For example, hazards to bees had relatively high CE ratios under all three policies for the CONV model. Policy design is thus complex, as policy types and levels must be considered simultaneously; then the issue of relationships between different policy evaluation criteria (for example, pesticide and fertiliser usage) and the potential for trade-offs between these must also be considered. The next step would be to consider the ratios in the light of information on social preferences and the priori-
ties of achieving hazards to the various dimensions. The CE trends for each policy with regard to ecological hazard dimensions are summarised in Table 6.

5. Discussion

The policy issue here is how best to reduce pesticide contamination of the environment, especially through encouraging reduced pesticide application levels, with the lowest possible reduction in farm production and incomes, and without prejudicing other environmental policy goals such as reductions in nitrate leaching. In addition, a more immediate issue is how to combine and analyse the available information for useful policy analysis.

One aim of the research was to develop agri-environmental policy evaluation methodology, focusing on the potential of economic incentives to reduce the environmental problems of pesticide usage. The linking of farm-economic and ecological components (through the use of the pesticide hazard indicator) in the model was found to be useful for articulating the trade-offs in policy implementation between different environmental concerns. Existing policy evaluation work has been extended in several aspects within a conceptual ‘production ecology’ framework, i.e. through

- inclusion of low-spray input practices, based on empirical trials data collected in the region;
- inclusion of a physical spray usage component, and
- inclusion of a set of indicators representing pesticide hazards along a number of human health and ecological dimensions.

An important hypothesis was that the choice of environmental or health policy target was likely to be highly important to policy cost-effectiveness since targets are not perfectly substitutable. Objectives such as reductions in the hazards posed by spray usage to different environmental dimensions were achieved with different levels of cost-effectiveness. Imperfect correlations between spray expenditure and hazard imply that ad valorem spray taxation is not the best way of achieving spray usage or hazard reduction goals, and the model results confirm this. Ad valorem taxation may however, be redeemed by its other attributes, such as its relative simplicity and familiarity as a policy instrument.

The differences between pesticides in terms of their potential environmental hazards and impacts, and the relative efficiency of HAZ suggest that compound-specific policies, i.e. differentially-applied taxes or restrictions, should be considered. Interestingly, the results suggested that there are potential trade-offs between the improvements to the different dimensions possible under the policies (e.g. between pesticide and chemical fertiliser contamination reductions). While TAX was the least cost-effective instrument at reducing pesticide usage and hazards, it also had some secondary environmental advantages in being the policy instrument that most cost-effectively reduces fertiliser usage as a positive side-effect. However, further work investigating other agricultural systems is needed to confirm whether this

<table>
<thead>
<tr>
<th>Policy</th>
<th>CONV</th>
<th>PROG</th>
</tr>
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<tbody>
<tr>
<td>Bees</td>
<td>HAZ then UNIT; KG performed better than TAX</td>
<td>HAZ and UNIT; KG and TAX performed badly (no change in hazard levels)</td>
</tr>
<tr>
<td>Water</td>
<td>HAZ and UNIT, then KG</td>
<td>HAZ and UNIT, KG and TAX slightly counter-productive</td>
</tr>
<tr>
<td>Warnings</td>
<td>HAZ and UNIT, then KG</td>
<td>HAZ and UNIT, TAX counter-productive</td>
</tr>
<tr>
<td>Fish</td>
<td>All instruments performed similarly</td>
<td>UNIT then HAZ, KG performed worst</td>
</tr>
<tr>
<td>Game</td>
<td>All instruments performed similarly (KG poorest), TAX marginally better for small reductions in hazard</td>
<td>Not applicable (zero hazard)</td>
</tr>
<tr>
<td>CONV</td>
<td>PROG</td>
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<tr>
<td><strong>Negative CE</strong></td>
<td><strong>Positive CE</strong></td>
<td><strong>Negative CE</strong></td>
</tr>
<tr>
<td>TAX</td>
<td>Warning, inhalation, water, ingestion, eyes, skin (up to around the 170% tax level)</td>
<td>Bees, fish (low CE)</td>
</tr>
<tr>
<td>UNIT</td>
<td>None</td>
<td>Ratios peak soon after critical tax level, then fall; range CE ratios (bees had the highest)</td>
</tr>
<tr>
<td>KG</td>
<td>Eyes and skin (KG levies under £20/kg)</td>
<td>Bee hazard reductions were most cost-effective, also special warnings and water hazards</td>
</tr>
<tr>
<td>HAZ</td>
<td>None</td>
<td>Bee hazard reductions were most cost-effective, also special warnings and water hazards</td>
</tr>
</tbody>
</table>
is a generalisable result. Nevertheless, although the models included only some of the chemicals potentially used in arable production, they were chosen on the basis of survey data to reflect typical strategies for each crop, so results are thought to be reasonably generalisable within this type of production.

Instrument rankings were not consistent over all of the ecological and health hazard indicators. This is an important observation as practical policy-making must take into account and meet multiple objectives, so issues such as which objectives take priority, and the degree to which trade-offs between the achievement of different objectives are acceptable, must be resolved. The next step in practical policy-making terms requires reaching a social consensus on favoured packages of ecological and health hazard reductions. Alternatively, information is needed on how acceptable negative changes to some dimensions are, if other changes are considered positive. A problem is the absence of information on social preferences for environmental protection relative to food production and other activities.

At present, farms are being encouraged to convert to low-pesticide production and further pesticide reductions are still a policy goal. It might well be that conversion to PROG production is sufficient to achieve agro-chemical reduction goals, implying that policy efforts should be focused on encouraging system change to PROG from CONV rather than tax pesticide inputs with farms remaining at CONV.

6. Concluding comments

Concepts such as sustainability imply that ecological problems should not be compartmentalised and examined in isolation: other objectives should be considered simultaneously, i.e. by taking an integrated, multi-dimensional approach to agri-environmental policy formulation and evaluation. Sustainable agriculture has ecological, agricultural and socio-economic dimensions, so trade-offs between different objectives must be recognised and compromises found. The type and degree of trade-offs may vary across different systems, so assessment will always, of necessity, be empirical.

Ideally, policy should aim to create incentives for changes in agricultural systems, such as movement towards integrated farm management. For example, policy might develop based on incentives for conversion to integrated farm management, rather than simply partial changes such as switches between chemicals or crops. Partial changes would probably risk shifting adverse impacts to different compartments of the ecological system, instead of actually reducing undesired effects. Pesticide heterogeneity even means that the effects of pesticide usage reductions are not necessarily positive. It is very unlikely that a single instrument will satisfactorily solve the multi-dimensional problems of pesticide contamination. Advice, extension and education will be central to any strategy.

The complexity of non-point environmental problems is a significant challenge for analysis. The aim of this study was to contribute to the development of effective ways of utilising available knowledge for policy evaluation and management recommendation purposes; linking ecological and economic components would improve quantitative evaluation of the farm-level impacts of different pesticide reduction policies. The meeting of economic and ecological perspectives requires questioning the neo-classical belief that a simple theoretical construct based on relatively simple assumptions can be used to analyse all economic activity, as well as the environmental issues associated with the economic system. Neo-classical economic theory provides a useful conceptual orientation for policy-making and a basis from which to qualify recommendations or make refinements in modelling. However, it is very important to raise awareness of the system within which we exist, produce and consume and the secondary effects of our actions.

The issue is not one of the validity of economics in environmental research, but of the role it can play and the way in which this role takes into account the role of other disciplines. Ecological perspectives suggest that a more complex, multi-criteria approach should be applied, although this would detract from the mathematical simplicity of
the current economic framework. Trade-offs between different environmental concerns, as well as between environmental quality and agricultural production, are inevitable. Therefore, rather than taking a piecemeal approach to policy-making, concerns should be integrated into a comprehensive framework, to exploit positive side-effects of different components and reduce conflicts. Given a complex real world of inter-related components, and high transactions (and policy-making) costs, it is appropriate to recognise multiple outcomes from single instruments. More work is needed to identify the opportunities or mechanisms for joint implementation of policies and the joint achievement of different goals.

Finally, it is important to recognise that models are just tools to assist policy-makers, i.e. starting-points for analysis. In the absence of knowledge regarding social preferences, it is impossible to reach any firm conclusions about the preference of policy outcomes. Trade-offs must be resolved through the political process, unless the outcomes in all dimensions of a particular instrument dominate those of other instruments (for example, if a particular policy can stimulate a greater change (reduction) in environmental impact, at lower cost (foregone income losses) to farmers.

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