Identifying Common Fallacies in the Choice of Environmental Taxes for Agricultural Pollution Control: The Absence of Transaction Costs and the Normality of Agricultural Pollutants

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Abstract

The choice of environmental taxes is one of the central themes in policy design for agricultural pollution control, which dominates both empirical and theoretical research. This paper examines two assumptions very often employed in applied research, namely the absence of transaction costs and the normality of agricultural pollutants. Our results indicate that the well-known superiority of emission taxes over input taxes may not always be valid, when transaction costs are taken into account. Furthermore, the assumption of normal distributed agricultural pollutants overestimate the relative abatement costs.

Keywords: *Pigouvian Taxes, Input Taxes, Transaction Costs, Nitrate Pollution, Lognormal Data, Environmental Policy.*

Introduction

Agricultural activities generate a number of pollutants as by-product of their production processes such as: soil sediments, nutrients, pesticides, greenhouse gases, heavy metals and organisms that cause diseases. According to the existing evidences all those pollutants have devastating effects on human health and on environmental quality (Mainstone and Schofield, 1996). The uncompensated cost imposed to the society by these pollutants, which is usually referred to as a negative externality, results in market failure and resource miss-allocation. A typical solution to internalise negative externalities is the Pigouvian tax.

The majority of agricultural pollutants is classified as nonpoint (diffuse), which means that is virtually impossible to trace them back to a specific source. Therefore, it is not easy to assign responsibility to individual sources for their adverse environmental impacts, which complicates the application of Pigouvian taxes. The unobservability of agricultural pollutants results in policy design impediments characterised by adverse selection and moral hazard issues. Recent theoretical advances that address the issue of non-observable emissions include: ambient taxes proposed by Segerson (1988); random penalty schemes proposed by Xepapadeas (1991); and nonpoint tournaments put forward by Govindasamy

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et al. (1994). However, such schemes have questionable political appeal for two reasons; first, the information requirements for setting and monitoring such policies is prohibitively expensive; and second, they are associated with unresolved legal enforcement problems. Hence, it is not surprising that policy makers usually focus on indirect control measures such as policies that regulate input use and/or management practices.

Input taxes have long been deemed as alternatives to direct taxes on emissions (Common, 1977; Griffin and Bromley, 1982; Holtermann, 1976; Segerson, 1990). The major assumption is that there is a relationship between unobserved emissions and observed inputs, and therefore there is an option to tax inputs instead of emissions. Such a relationship can be defined through meta-modelling (Bouzaher *et al.*, 1993), which is the application of statistical techniques to the data generated by various biophysical models designed to assess the environmental impact of different control scenarios. Under specific assumptions the use of biophysical models may also provide the necessary information to convert nonpoint pollution problems to point ones, overcoming the unobservability of agricultural pollutants and facilitating the use of Pigouvian taxes. These assumptions refer to well designed, reliable and properly validated models, features that are *sine qua non* prerequisites for the "political acceptability" of such models.

A number of studies that combine the use of biophysical models with economic analysis have shown that a tax on emissions is a more efficient policy than a tax on inputs (Johnson *et al.*, 1991; Miltz *et al.*, 1988; Pan and Hodge, 1994). However, what is missing in such comparative studies is an explicit account of the transaction costs associated with different regulatory instruments. The inclusion of transaction costs can modify the relative efficiency of alternative policies (Smith and Tomasi, 1995; Vatn, 1998).

This paper examines whether input taxes can be considered as feasible substitutes to emission taxes, not because emissions are unobservable, a fact that can be overcome to an extent by using simulation models, but because they are associated with lower transaction costs (Romstad, 1999). In this context, transaction costs refer to the costs of assessing the tax base, which comprise the costs of running the simulation models and the monitoring costs needed to validate the model's estimates.

In addition, the assumption of normal distributed agricultural pollutants is relaxed and the likely bias of possible misspecification is explored. Using a modified cost-effectiveness framework, which takes into account the relevant transaction costs of alternative policy scenarios, a comparison between emission and input tax is examined for controlling nitrate pollution for the Kennet catchment in southern England.

The paper is organised as follows. In the next section, a theoretical model is developed that defines the cost-effective nitrate pollution control problem. Section 3 discusses the issue of skewed distributed agricultural pollutants. Section 4 briefly describes the empirical model applied. Section 4 compares the relative abatement and total costs imposed by emission and input taxes respectively. The final section draws some conclusions.

Including Transaction Costs into a Theoretical Model of Instrument Choice

Consider a catchment that has X hectares of agricultural land, located on n farms. Agricultural activities in the catchment are profitable but at the same time generate nitrate emissions that give rise to nitrate pollution.

The social planner seeks to minimise the total costs of firms' compliance with a specific environmental quality standard. Such total costs comprise the costs incurred by all regulated firms, usually referred to as abatement or control costs, and the total transaction costs of designing, administrating, monitoring and enforcing firms' compliance with the environmental standard (Stavins, 1995).

Following Xepapadeas' (1997) classification of pollution control options, we assume that the regulator has three different policy options to control nitrate pollution, namely emission-based, input-based and output-based. These options are denoted as EBO, IBO and OBO respectively. However, only the first two options are examined here, since it is known that output regulation is not a long-run efficient way of controlling pollution (Spulber, 1985).

We define the emission function as a strictly increasing function of the firm's decision vector, say inputs used, x_i. So the emission level, e_i, of the i producer is given by:

$$e_i = e_i\left(x_i, w\right) \text{ with } e_{x_i} > 0, \quad e_{x_i x_i} > 0 \tag{1}$$

where subscripts refer to partial derivatives and the vector w reflects the influence of weather and other site-specific factors on the emission generation, such as elevation, soil type etc. Since these factors are beyond the producers' control, the argument w is dropped without loss of generality.

The regulator's problem can be written as:

$$\min_{e_i} \sum_{i=1}^{n} C_i(e_i) + TC \tag{2}$$

In other words, the regulator's objective is to minimise the economic burden of achieving specific environmental quality targets. Transaction costs, TC, are treated as resource costs since they employ resources such as labour and capital, which have social opportunity cost (see Stavins (1995) for a discussion).

Following Shortle (1990) the abatement costs, $C_i\left(e_i\right)$, are defined as the difference between the maximum profits the firm can earn in the absence of environmental regulation and the maximum profits the firm can earn in the presence of an environmental control policy (restricted profits), that is:

$$C_i\left(e_i\right) = \overline{\pi}_i\left(q_i, e_i\right) - \pi_i\left(q_i, e_i\right) \tag{3}$$

where $\overline{\pi}$ stands for the maximum profit without environmental regulation and π for the maximum profit in the presence of such regulation; q_i is a vector homogeneous products produced by the i_{th} firm and their production generates e_i emissions. The specification of the profit function in (3) was originally introduced by Porter (1974), and it is often referred to it as the i_{th} firm's benefit function from engaging in a pollutant-generating activity (Chambers and Quiggin, 1996; Malik, 1991).

Sometimes, however, it is convenient to define the profit function over the vector of input uses and technological choices. Such an approach is followed by Ribaudo *et al* (1999), which can be written as:

$$\pi_i = \pi_i\left(x_i, a_i, e_i(x_i)\right) \tag{4}$$

Continuous choices are assumed for the input uses, x_i, but discrete choices are available for the technological choices (e.g. farm type/cropping systems), a_i. Furthermore, it is noteworthy that since the input use levels in (4) are those of the unique profit–maximising input demand of (3) given by the Hotelling's Lemma, both formulations are equivalent. Under such a definition of the profit function, a modification of (3) is possible, which can be written as:

$$C_i\left(e_i(x_i)\right) = \max_{x_i}\left[\overline{\pi}_i\left(x_i, a_i, e_i(x_i)\right)\right] - \max_{x_i}\left[\pi_i\left(x_i, a_i, e_i(x_i)\right)\right] \tag{5}$$

Given well behaved production functions (e.g. strict concave functions) and the fact that both output and input prices are determined exogenously, then the maximum expected profits the firm can earn in the absence of environmental control is constant (Beavis and Walker, 1983). Therefore, minimisation of abatement costs is equivalent to maximisation of expected restricted profits (benefits), that is:

$$\min_{e_i} C_i\left(e_i\right) \equiv \min_{x_i} C_i\left(e_i(x_i)\right) \equiv \max_{e_i} \pi_i\left(q_i, e_i\right) \equiv \max_{x_i} \pi_i\left(x_i, a_i, e_i(x_i)\right) \tag{6}$$

Transaction costs can be defined quite broadly to refer to any costs associated with establishing, administering, monitoring and enforcing a government policy or regulation. As Krutilla (1999) argues, any policy related costs other than the costs of pollution abatement, or producer and/or consumer welfare effects arising from policy-induced output changes, would qualify under the residual transaction costs label.

In our model, transaction costs depend on the pollution control policy and the various aspects of that policy, such as the amount of monitoring it requires, M_j, and how easy is to implement and administrate, A_j. Thus, the transaction costs function can be written

$$TC_j = M_j + A_j \tag{7}$$

where $j = e$ denotes emission based policies and $j = x$ denotes input based policies.

A more detailed disaggregation of the transaction costs can be found in Herath and Weersink (1999) and in McCann and Easter (1999), where extra costs related to the design and enforcement of the policy are included. In the context of diffuse pollution and when taxation is considered to be a policy option monitoring costs, M_j, are found to represent the largest component of transaction costs (McCann and Easter, 1999). In addition, the administrative costs, A_j, are usually represented by a specific proportion of the collected taxes (Kohn, 1991; Polinsky and Shavell, 1982).

Emission Based Policy Option (EBO)

Polluting firms are forced to comply with a given environmental standard if they are confronted with an emission tax per unit of emission released into the environment. The regulator's responsibility is to define and impose such a cost-effective emission tax. To do so, the regulator can use the derived profit function and the transaction costs function to solve the problem:

$$\max_{e_i} \sum_{i=1}^{n} \pi_i\left(q_i, e_i\right) - TC_e \text{ s.t. } \sum_{i=1}^{n} e_i \leq \tilde{e} \tag{8}$$

where $\tilde{e}$ stands for the maximum admissible level of emissions from the society's point of view The Kuhn-Tucker conditions for a global optimum, ignoring any corner solutions and given that $\partial \pi_i\left(q_i, e_i\right) / \partial e_i = -\partial C_i\left(e_i\right) / \partial e_i$, imply that:

$$\left[-\frac{\partial C_i\left(e_i\right)}{\partial e_i} - \frac{\partial TC_e}{\partial e_i} - \lambda \right] = 0 \tag{9}$$

where λ is the shadow price of the environmental constraint The implication of (9) is that the incorporation of transaction costs leads to different decision rules than the standard solution which requires that the marginal abatement cost should to be equal to the shadow price, λ, of the environmental constraint (Krupnick *et al.*, 1998). In turn, the cost-effective Pigouvian tax is given by that shadow price. However, from (9) it is obvious that having accounted for the transactions costs, the cost-effective Pigouvian tax is equal to the marginal abatement cost plus the marginal increase in the transaction costs.

Consequently, the imposed Pigouvian tax forces each individual firm to solve the problem:

$$\max_{q_i, e_i} \pi_i\left(q_i, e_i\right) - \lambda e_i \tag{10}$$

Input Based Option (IBO)

Under the assumption that the regulator has perfect information about the emission function, then emission restrictions can be translated into inputs restrictions as Figure 1 illustrates.

MB stands for the marginal benefit function while VMP denotes the value of the marginal product. An emission tax, t_e, reduces the emission level from e to e^*, and an input tax, t_N, reduces the input use from x to x^*. The input price is denoted by w.

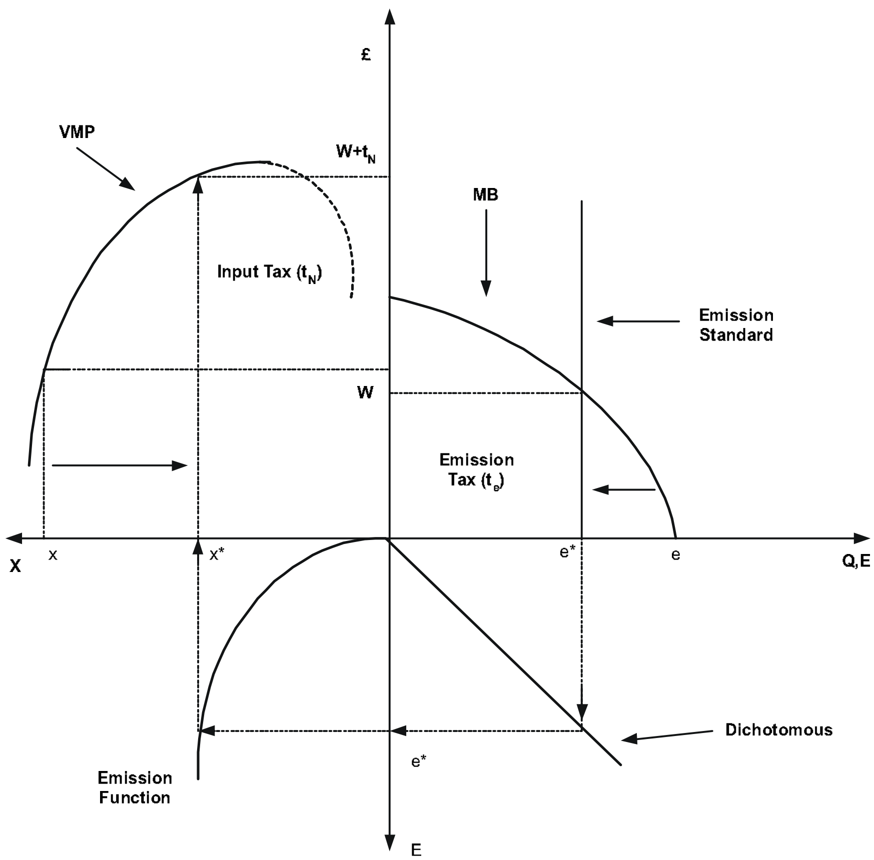

Figure 1. The relationship between emission and input restrictions (and taxes) under perfect information

Under IBO, polluting firms are forced to comply with a given environmental standard if they are confronted with an input tax per unit of inputs used. As in the EBO it is the regulator's responsibility to define and impose such a cost-effective input tax, by solving the problem:

$$\max_{x_i} \sum_{i=1}^{n} \pi_i \left(x_i, a_i, e_i(x_i) \right) - TC_x \text{ s.t. } \sum_{i=1}^{n} x_i \leq \tilde{x} \tag{11}$$

where $\tilde{x}$ represents the maximum amount of input that can be used. Such input constraint can directly be derived by the emission constraint given the knowledge of the emission function (1), something that is illustrated in Figure 1. The Kuhn-Tucker conditions for a global optimum, ignoring any corner solutions, imply that:

$$\left[-\frac{\partial C_i\left(e_i(x_i)\right)}{\partial e_i}\frac{\partial e_i}{\partial x_i}-\frac{\partial TC_x}{\partial x_i}-\lambda_1\right]=0 \tag{12}$$

The shadow price of the input restriction, λ_1, gives the cost-effective input tax (Hahn, 1995). Facing the input tax defined by (12) each individual firm forced to solve the problem:

$$\max_{x_i}\left[pf_i(x_i,a_i)-\left(w+\lambda_1\right)x_i\right] \tag{13}$$

By manipulating (9) and (12), it is possible to derive the relationship between the different shadow prices of EBO and IBO, which is:

$$\left[\lambda+\frac{\partial TC_e}{\partial e_i}\right]\frac{\partial e_i}{\partial x_i}=\lambda_1+\frac{\partial TC_x}{\partial x_i} \tag{14}$$

According to (14) the relationship between cost-effective input and emission taxation does not only depend on the return to scale of the emission function, as identified by Stevens (1988), but also on the relative marginal transaction costs of the two policies. It is evident that a cost-effective input tax is equal to a cost-effective emission tax as long as the emission function exhibits constant returns to scale, $\partial e_i/\partial x_i=1$, and also when the two policies are characterised by the same marginal transaction costs.

On the Skew Distribution of Nitrate Emissions.

The estimation of expected value of the total emissions raised an interesting computational issue. If we assume, either explicitly or implicitly through the Law of Large Numbers (Parzen, 1960), that the aggregate level of nitrate emissions is normally distributed, then its mean value represents a computationally trivial case (Shortle, 1990). In such a case, we have:

$$E\left[\sum_{i=1}^{n}e_i\right]=\sum_{i=1}^{n}E[e_i] \tag{15}$$

where E stands for the expected value.

However, if the underlying distribution of the aggregate level of the nitrate emissions is lognormal, then the arithmetic mean is not the minimum variance unbiased estimator (Gilbert, 1987). The same is valid for the geometric mean as well (Landwehr, 1978). Therefore, we explicitly consider the case that nitrate emissions follow a skew distribution, such as the lognormal, and we use the expected value for such a distribution. The scientific justification for such an assumption is given by Ott (1990) and subsequently was followed by a number of studies such as Schmoyer *et al.* (1996) and Rosen *et al.* (1998). Then, by using the method of matching moments (MoMM) (Thompson, 1999) the sum of the nitrate emissions under the assumption of lognormality is given by:

$$E\left[\sum_{i=1}^{n} e_i\right] = \ln\left(\sum_{i=1}^{n} E[e_i]\right) - 0.5\ln\left(\frac{\sum_{i=1}^{n}\operatorname{var}(e_i)}{\left(\sum_{i=1}^{n} E[e_i]\right)^2} + 1\right) \tag{16}$$

where $E[e_i]$ and $\operatorname{var}(e_i)$ are the mean and variance of the individual nitrate sources empirically estimated from the simulation models.

It seems that there is a general consensus in the literature that the lognormal model is the most appropriate one for environmental impact analysis (Cooper *et al.*, 1996; Rabl and Spadaro, 1999). Nevertheless, since the appropriate statistical tests to discriminate between normal and lognormal distribution require a very large sample size (>1000) (Gingerich, 1995), which was not available, we decided to examine both normal and lognormal assumptions and explore the relative bias of the likely misspecification.

Brief Description of the Empirical Model

The application of the theoretical model to the Kennet catchment requires representation of nitrate emission functions and abatement costs within an economic model. The approach adopted is to divide the catchment into land classes characterised by different production and emission possibilities and to use an aggregate non-linear programming model to simulate producers' responses to different nitrate pollution control measures. The empirical model proceeded in three stages. First, various data sets on agricultural production and land uses were combined within a Geographical Information System (GIS). A GIS was used to classify land classes based on their soil properties, to identify the total area of the catchment and to spatially distribute agricultural activities.

Second, nitrate emission function for all the major soil/crop combinations were estimated econometrically from nitrate emission estimates derived from the simulation models: SUNDIAL for the arable land, and N-CYCLE for the grassland. SUNDIAL (SimUlation of Nitrogen Dynamics In Arable Land) is a PC based version of the Rothamsted Nitrogen Turnover Model (RNTM) which was used to estimate nitrate losses given a set of agronomic parameters and weather conditions (Smith *et al.*, 1996). Similarly, N-CYCLE is an empirical mass-balance, which simulates the nitrogen cycle in grassland systems developed at the Institute of Grassland and Environmental Research (IGER) (Lockyer *et al.*, 1995). In addition, a simple hydrological model, TOPCAT (Quinn and Antony, 1996) was used to estimate the concentration of nitrates in the drainage water. The pattern of nitrate emission estimates for the Kennet catchment was consistent with the range of values for nitrate emissions found in the literature (Bradbury *et al.*, 1993; Whitehead, 1995).

The third stage assembled the production and pollution information within a non-linear optimisation framework. A non-linear programming model was used since both the production and the emission functions are non-linear. Crop production functions were taken from the literature (England, 1986). The whole catchment was represented as a single profit maximising firm, which is a typical specification of sector models within a mathematical programming framework (Hazell and Norton, 1986). Output and input prices refer to the 1995-1996 production period. Figure 2 depicts the model structure of the empirical application, as flow chart that links data, estimates and models. Similar integrated modelling analysis for diffuse pollution control is followed by Bouzaher and Shogren (1995), Vatn *et al.*, (1997) and Wu *et al.* (1995).

The associated transaction costs with the two policies examined comprise the costs of monitoring and administrating them. The reported monitoring costs of

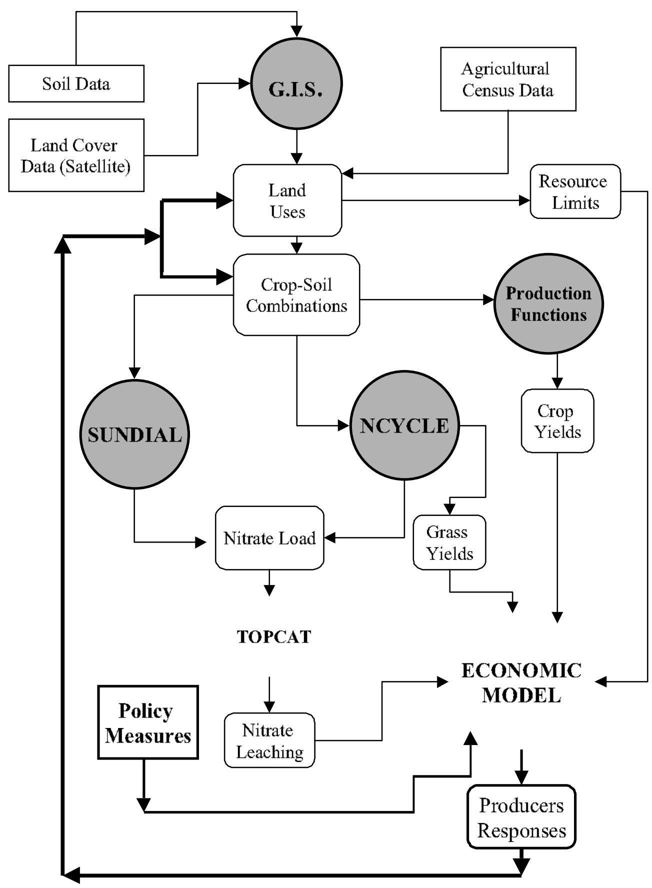

Figure 2. Modelling Framework for Nitrate Pollution Control

the Nitrate Sensitive Areas scheme (NSA's) and the Environmental Sensitive Areas (ESA's) are taken as a proxy for transaction costs associated with emission taxes and input taxes respectively. NSA's is an EU policy that involves periodical nitrate emission monitoring in order to assess farmers' compliance with the scheme (M.A.F.F., 1998). On the contrary, ESA's is an EU scheme that involves input monitoring (M.A.F.F., 1997). The per hectare monitoring costs of ESA's scheme were £52,19 for the period 1995-1996, while the relevant costs of the NSA's were £103,63 (Agriculture-Committee, 1997).

Following Polinsky and Shavell (1982), the administration costs associated with policies that involve taxation are assumed to be a specific proportion of the relevant tax payment. For example Kohn (1991) uses the value of 0,5%, while anecdotal sources for the UK suggest a range of values between 0,5% and 1% (Falconer, 1999). In terms of our case study the value of 1% was used.

Results

Based on the integrated modelling framework briefly described in section 4, this section examines the relative efficiency of emission versus input taxes in achieving specific reductions in the concentration of nitrates in the drainage water. These reductions are chosen to be 10%, 20% and 30% of the baseline solution, where the baseline solution corresponds to the unrestricted profit maximising problem. Regulations expressed as percentage reductions are very common in applied research (McKitrick, 1999).

Figure 3 gives the abatement costs under emission and input taxation when the tax payment, which is a transfer payment, is not included in the relevant abatement costs. Since transfer payments do not contribute to economic welfare,

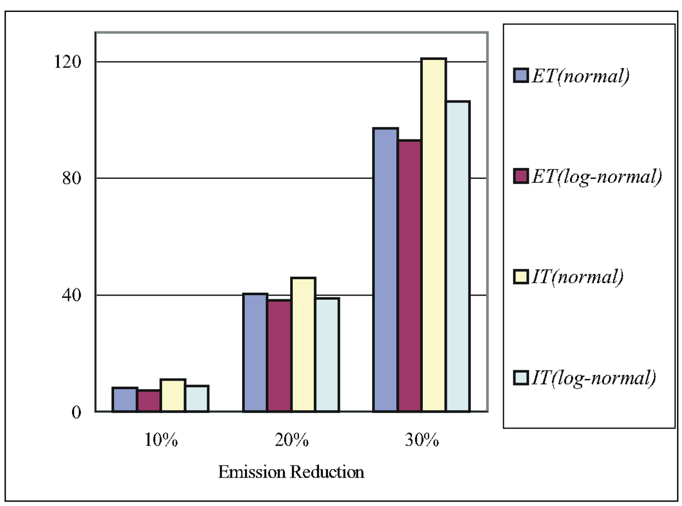

Figure 3. Relative Efficiency of Emission Taxes versus Input Taxes Based on the Abatement Costs (£ 000´s) Definition: ET: Emission Tax; IT: Input Tax

they are excluded from the social welfare costing (Prato, 1998). These abatement costs represent the resource costs from society's point of view in meeting the specific emission reduction targets. Strictly speaking, since markets are distorted due to the various ways of government intervention, these costs are not the 'true' resource costs

The results in Figure 3 are the expected ones for both the instrument choice comparison and the parametric assumption of the distribution of the random variable, that of nitrates. Since the emission function is found to be a convex function in inputs and to exhibit increasing returns to scale, an emission tax outperforms an input tax. For all the different levels of emission reductions examined an emission tax results in lower abatement costs than an input tax.

In addition, the assumption of normally distributed emission rates results in higher abatement costs than the assumption of lognormally ones. Again, this is an expected result since it is known that the normality assumption overestimates lognormal data (Al-Khalidi, 1994). Nevertheless, the relative misspecification bias is not that important.

By contrast, the relative superiority of emission taxes over input taxes as shown in Figure 3 is dramatically altered when the comparison between emission and input taxes is based on the total cost, which comprise the resource, the monitoring and the administration costs. Figure 4 shows the relative total costs of pollution control imposed by emission and input taxes.

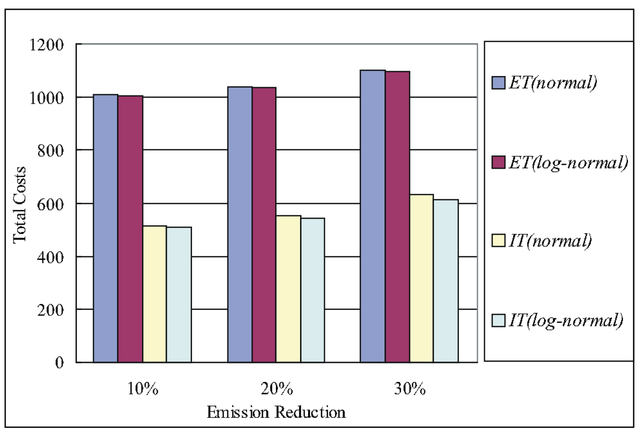

Figure 4. Relative Efficiency of Emission Taxes versus Input Taxes Based on the Total Costs (£ 000´s) . Definition: ET: Emission Tax; IT: Input Tax.

As is shown in Figure 4, emission taxes are not the most efficient instrument when the relevant transaction costs are brought into the equation. This is an interesting result that contradicts the conventional wisdom of the majority of environmental economics textbooks, in which emission taxes are presented as the instrument that can force environmental compliance at a minimum cost. Thus, our empirical results support Vatn's (1998) proposition which stresses that statements that put forward the superiority of emission taxes, explicitly or implicitly assuming zero transaction costs, create confusion and errors both in defining the problem and in the search of the appropriate control instrument.

Conclusions

The purpose of this paper was twofold. First, to examine both theoretically and empirically the incorporation of transaction costs into economic analysis. As was shown in section 2, such incorporation of transaction costs determines the efficiency ranking between emission and input taxes. Second, to empirically examine the likely bias of abatement cost estimates under the often employed normality assumption of agricultural pollutants.

The results presented here provide some empirical evidence that the extensively advocated superiority of emission taxation may not always be valid. In real-life policy comparisons, when the total cost considerations should form the basis for policy rankings we have shown that emission taxes are not the best instrument. If this is the case, and assuming that our results are not context or site specific, then it is legitimate to argue that the exclusion of the relevant transaction costs can result in quite misleading policy recommendations.

Finally, the assumed normality, either explicitly or implicitly, of nitrate emissions overestimates the relevant abatement costs. Suffice to say that such bias is negligible without affecting the relatively efficiency of the taxes examined. On the contrary, the cost of being wrong by ignoring the transaction costs is immense, which can alert the efficiency ranking of alternative control policies.

Notes

From (6) it is evident that

$$\frac{\partial \pi_i(x_i, a_i, e_i)}{\partial x_i} = -\frac{\partial C_i(e_i)}{\partial x_i} = -\frac{\partial C_i(e_i(x_i))}{\partial x_i} = -\frac{\partial C_i(e_i(x_i))}{\partial e_i}\frac{\partial e_i}{\partial x_i}.$$

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