Review

Policy effectiveness and acceptance in the taxation of environmentally damaging chemical compounds

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1. Introduction

Although the use of market-based environmental policy instruments such as taxes and charges has become more prevalent in recent years, the role of environmental taxes and charges in the OECD area is still limited. About 6–7% of the total tax revenues are environmentally related and over 90% of the environmentally related taxes are applied within the energy and transport sectors (EEA, 2000). Within the OECD countries less than 5% of total environmental tax revenues are in turn taxes on chemical substances, products, waste, emissions and virgin natural resources. There exists however a growing interest among analysts and policymakers towards extending the environmental tax base, and many of the proposed schemes include taxes on chemical compounds. One example includes the OECD Environmental Outlook Study...
(OECD, 2001) which investigates a policy mix that would include taxes on chemical use, and that shows that the chosen policy mix could deliver important environmental benefits (e.g., significantly reduced nitrogen loadings) at relatively low economic costs. Proposals for the introduction of environmental taxes on chemicals have also been put forward by, for instance, the European Commission (CEC, 2002) and organizations and government authorities in Canada (Green Budget Coalition, 2005), Denmark (DEPA, 2005), and New Zealand (ERMA, 2004). In some countries the political interest in finding new environmental tax bases is spurred by the presence of a general green tax shift policy.

Before new taxes on chemical compounds are introduced one needs to raise – and attempt to answer – a number of critical questions. One concerns naturally the efficiency of a tax scheme compared to alternative environmental policies (e.g., emissions or technology standards), and this issue is well covered in the existing literature (e.g., Helming, 1997; Kleinhanss et al., 1997). In this paper, however, our starting point is a situation where a tax policy on chemicals is planned to be implemented. At this stage two questions are worth addressing. What is the current use of chemical taxes, and what can we learn from these experiences for future policy designs? Finally, what are the legal and political obstacles towards the implementation of taxes on chemicals? The overall objectives of this paper are therefore to analyze: (a) the economics and politics of taxing chemical compounds; and (b) the future potential for increased implementation of such taxation policies in Europe. While much of the discussion is general in scope, the empirical part focuses on the case of fertilizer taxation in five European countries: Austria, Denmark, the Netherlands, Norway and Sweden.

The choice of fertilizer taxation as an interesting case is motivated for a number of reasons. Although the use of environmentally motivated fertilizer taxes is not very widespread in Europe a few interesting and among them different tax schemes have been implemented over the last two decades. Among these we find tax policies which supposedly have proved to be effective, but also those that have been less effective and even abandoned due to political reasons. This permits a comparative analysis of both the cost effectiveness – i.e., the extent to which given reductions in nitrogen leakages are met at minimum costs to society – as well as the political acceptance of fertilizer taxes in developed countries. The study of national policy choices for fertilizer reduction is also interesting since a European Community (EC) Directive (Council Directive 91/676/EEC) sets EC standards for nitrates in groundwater and surface water. The Directive is a typical example of a traditional command-and-control approach in environmental policy, and it includes, for instance, regulations on how to handle manure in zones particularly vulnerable to nitrate leaching. The discretion of member states to adopt and implement environmental taxes on fertilizers is subject to measures taken at the community level (their legal bases and their contents), and the Nitrate Directive is particularly important in this case (see also Section 2.2). Although this Directive and other community measures do not prevent the implementation of environmental taxes per se, they can provide important hurdles that need to be addressed in the policy design and implementation processes.

The taxation of chemical use is typically motivated by the desire to target downstream external costs in the form of harmful exposure to nature. These non-point source emissions may be difficult and costly to control in those cases where environmental damages vary by location. Therefore, it is often easier to tax the production or the use of chemical compounds upstream. This is a typical situation for most chemicals, and the taxation of fertilizers is a very representative example of this policy dilemma. A number of theoretical studies on fertilizer taxation clearly illustrate the trade-offs involved in either achieving a cost-effective reduction in nitrogen leakage by taxing close to the damage caused on the one hand, or employing a simple tax system with low administrative and monitoring costs by taxing inputs on the other (Section 2.1). The practical experiences reviewed in this paper show that these features of many fertilizer taxes matter for both the cost and the environmental effectiveness of the taxes, but also for the prospects of marketing them as desirable policy instruments in the political arena.

While previous studies on chemical taxation in general and fertilizer taxation in particular tend to focus on either the cost effectiveness of such taxes (see Table 1 and Rougoor et al., 2001) or on the political legitimacy of such taxation policies (e.g., Daugbjerg, 1998, 2000; Vatn et al., 2002), we attempt in this paper to also focus on the interaction between the cost-effective design of fertilizer taxation and the prospects for implementing these tax schemes in practice. The fertilizer case illustrates a number of general policy implementation and design issues, which in turn provide important lessons for increasing the effectiveness and the legitimacy of future tax policies in the chemicals field.

Before proceeding some important limitations of the paper and a number of definitions used should be emphasized. First, we focus solely on fertilizer taxes which aim at influencing environmentally damaging behaviour. This implies, for instance, that we do not discuss the Finnish fertilizer tax that was in place between 1976 and 1994. The objective of this tax was solely to lower production levels of cereals for export and to provide funds to finance export subsidies (Rougoor et al., 2001). Given the difficulties in distinguishing between environmental and revenue-raising (fiscal) taxes, taxes for which fiscal and environmental objectives are combined (e.g., Austria) or inseparable are included in the analysis. Second, a distinction is often made between taxes and fees. Taxes are essentially compulsory unrequired payments to the state budget, while fees are earmarked in the sense that the revenues are spent on related (typically environmental) purposes and often recycled back to the sector on which the fees were levied. In the paper, the term tax is used regardless of whether the revenues are earmarked or simply channelled to the national budget.

In Section 2 we briefly review some theoretical considerations concerning the cost-effectiveness aspects of chemical taxation as well as the policy process involved when implementing such taxes. Section 3 reviews some of the most relevant economic and political experiences of fertilizer taxes in Austria, Denmark, the Netherlands, Norway, and Sweden. In Section 4 we discuss the prospects for increased use of fertilizer taxes in Europe, not the least by drawing from the experiences gained in the above countries. Finally, Section 5
provides some concluding remarks and implications for chemical tax policies in general.

2. **The economics and politics of taxing chemical compounds**

In this paper, we recognize that an increased reliance on chemical taxation in environmental policy requires that such taxes are desirable from an environmental and cost-effectiveness point of view but also that they are politically viable. As was noted above, an important feature of a cost-effective tax is that it can target – as closely as possible – the relevant external damage. For many chemical compounds – including those related to fertilizer use – the harm varies because of the receiving environment’s conditions, so optimal tax rates would be geographically differentiated. The successful implementation of new taxes on chemicals as part of environmental policy not only requires that such policies are perceived as effective means of attaining set goals, but also that it: (a) gains support (and even promotion) from important (international and national) organizations; (b) is not in serious conflict with international agreements; and (c) will be initiated and pushed forward by strong policy coalitions. In this section, we discuss both the economics and the politics of implementing taxes on chemicals, and we use the case of fertilizer taxes to illustrate some of the key concepts.

2.1. **The cost effectiveness of chemical input taxation**

Taxing chemical compounds is often perceived as controversial, and traditionally the use of chemicals has mainly been subject to command-and-control regulations (including bans). In spite of this some analysts maintain that taxes on chemical inputs can be just as desirable (e.g., Macauley et al., 1993; Rendleman et al., 1995; Slunge and Sterner, 2001). A tax policy

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### Table 1 – Selected simulation studies of fertilizer taxation schemes

<table>
<thead>
<tr>
<th>Study and geographical area</th>
<th>Purpose and methodological approach</th>
<th>Main findings and implications</th>
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<tbody>
<tr>
<td><strong>Kim et al. (1999)</strong></td>
<td>A competitive dynamic model is used to examine the type of tax rate that minimizes compliance costs while maintaining a predetermined groundwater quality level.</td>
<td>The net economic benefits are the highest under a variable-unit tax, and slightly lower in the case of a constant-unit tax. A pollution tax results in the lowest net-benefits.</td>
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<tr>
<td><em>(Nebraska, USA)</em></td>
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<td><strong>Brannlund and Gren (1999)</strong></td>
<td>Employs a short-run partial equilibrium analysis to estimate fiscal incomes and environmental impacts from increases in the Swedish nitrogen fertilizer tax.</td>
<td>The estimated fiscal incomes are, due to the presence of low own-price elasticities of demand, increasing until the tax level corresponds to a 200% increase in the fertilizer price.</td>
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<tr>
<td><em>(Sweden)</em></td>
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<tr>
<td><strong>Fleming and Adams (1997)</strong></td>
<td>Assesses the importance of spatial variance in physical parameters in the design of a tax policy to control nitrate concentration in groundwater. A spatial tax, which is differentiated across regions, is compared to a uniform tax.</td>
<td>The spatial tax is a more cost-effective instrument than a uniform tax. Still, the differences are small, and since a spatial tax requires extensive geophysical information, higher costs of monitoring and implementation will be induced.</td>
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<tr>
<td><em>(Oregon, USA)</em></td>
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<tr>
<td><strong>Huang and LeBlanc (1994)</strong></td>
<td>Analyzes the impacts of a residual nitrogen tax, as an alternative to an ad valorem tax, on water quality. Farmers are then taxed for the leaching of residual nitrogen into groundwater.</td>
<td>The impacts on corn production and nitrogen reduction of the tax vary across state; however, the residual nitrogen tax is claimed to be more cost effective.</td>
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<td><em>(Corn Belt states, USA)</em></td>
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<td><strong>Helming (1997)</strong></td>
<td>Uses a spatial equilibrium model to analyze the impacts of a tax on nitrogen surpluses along with other policy options such as taxing fertilizer inputs.</td>
<td>The demand for fertilizers is own-price inelastic, so high taxes are needed to attain significant reductions. In addition, a tax on nitrogen surpluses at the farm level is more cost effective than an input tax.</td>
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<tr>
<td><em>(The Netherlands)</em></td>
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<tr>
<td><strong>Kleinhanss et al. (1997)</strong></td>
<td>A regional production and nutrient balance model is used to analyze the economic and environmental impacts of different economic instruments implemented to reduce the use of fertilizers in Europe.</td>
<td>The model results show that a tax on nitrogen surpluses (kg N per hectare) would only have limited effects within areas where intensive livestock production takes place, but the overall reduction would be considerable.</td>
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<td><em>(Europe)</em></td>
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<tr>
<td><strong>Berntsen et al. (2003)</strong></td>
<td>Evaluates the economic and environmental impacts of different types of nitrogen tax policies. These include, among others, a tax on nitrogen in mineral fertilizer and a tax on the farm nitrogen surplus.</td>
<td>None of the tax policies was the most cost effective for all four farm types considered. A tax on mineral fertilizer favors pig producers, whereas a tax on the nitrogen surplus favors arable farms.</td>
</tr>
<tr>
<td><strong>Rørrstad et al. (1999)</strong></td>
<td>Analyzes the economic and environmental effects of a tax on chemical fertilizers, a tax on nitrogen surplus and a tradable fertilizer permit scheme using the Norwegian EcEcMOD modeling system.</td>
<td>There exists no one-to-one relationship between nitrogen surplus and leaching. Still, a reduction in nitrogen surplus implies in most cases a reduction in leaching. None of the policy instruments considered can target leaching cost effectively.</td>
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<td><em>(Norway)</em></td>
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can, it is argued, be more cost effective and at the same time maintain (or even increase) compliance as it provides the industry with more flexibility in compliance strategies and therefore also faces less resistance. Taxes on chemical inputs are typically motivated by the desire to reduce downstream externalities in the form of harmful impacts on nature. These non-point source emissions are often very difficult and costly to control, at least if the environmental damages caused by the substances are affected by geographical location and the receiving environment’s condition (e.g., nitrogen, pesticides, etc.). It may thus be more effective to tax the production or the use of chemical compounds upstream in the product chain (e.g., Vatn, 1998). Normally there are relatively few wholesalers and importers of a chemical compound compared to the large numbers of actors downstream, i.e., manufacturers of products using the chemical. Similarly, there are fewer of these intermediate producers than there are of the thousands of diverse end-users.

By increasing the price of primary chemicals, incentives are created to minimize use and recycle used chemicals as the cost of the product tax is passed on to consumers. Chemical input taxes also provide incentives to reduce spills, etc., something that a pollution tax does not, although this attribute of chemicals taxation is still under-researched. The implementation of taxes on consumption and use to address downstream pollution does generally not induce a cost-effective allocation of compliance methods, this since they do not provide any incentives to undertake mitigation measures at the user or disposal stage. It “only” provides incentives to reduce consumption (through higher prices) as well as production (through lower profits), and thus lower emissions in these ways. Only in (unlikely) cases where no abatement technologies exist will a tax on output be the best pollution policy (e.g., Sterner, 2003).

The trade-off between costly monitoring of emissions and the cheaper strategy of taxing inputs is an issue that figures prominently in earlier theoretical work on chemical taxation. Table 1 summarizes the scope, approach and results of selected simulation studies of the efficiency impacts of different fertilizer tax schemes (compared to alternative policy instruments). These studies clearly illustrate the trade-offs involved in either achieving a cost-effective reduction in nitrogen leakage by taxing close to the damage caused on the one hand, or employing a simple tax system with low administrative and monitoring costs by taxing inputs on the other. In many instances the latter approach may be the most desirable since it minimizes total costs (the sum of compliance and monitoring costs), but it is vitally important to address the natural geographical conditions. These studies as well as others (e.g., Brady, 2003) also show that the cost effectiveness of a tax approach may depend heavily on the existence of additional policy instruments.

Some authors stress, though, that even taxes on emissions will be inefficient since the level of application is not always proportionate to the resulting pollution (see also Segerson, 1988; Hrubovcak et al., 1990; Hansen, 2002). Most importantly, the damage functions of non-point source emissions, such as pesticide, nitrate and phosphorus pollution, are often characterized by convexity, that is, with sharply rising marginal damage above a critical level (see also below). This strengthens the conclusion that in chemicals policy it is inherently difficult to implement a tax which in any way comes close to reflecting damage done.

Still, the controversial issue in taxing chemicals may lay not so much in that taxes will not target the relevant environmental impacts – although clearly this is also a crucial issue – but rather in the fact that other policy instruments may be more desirable and effective. The use of environmental taxes is based on the assumption that firms compare the marginal costs of reducing chemical use with the tax rate, and then reduce use up to the point where the marginal cost of reduction equals the tax rate. The problem, however, is that the regulator generally has far from complete information about the marginal reduction cost function, and for this reason it becomes difficult to know whether enough (or too much) reduction will be achieved. This is particularly troublesome in the case of hazardous chemicals, for which – as noted above – there may exist critical threshold levels that should not be exceeded. The environmental economics literature recognizes that the existence of major uncertainties about reduction or abatement costs is critical for the choice between different environmental policy instruments. In such a situation, emphasis may be put on controlling quantities rather than prices, especially if the (marginal) environmental damage rises steeply compared to (marginal) reduction costs.1 This implies that command-and-control regulations (and even bans) could be more attractive.

From an economic efficiency point of view taxes on chemicals would therefore be appropriate if the marginal reduction cost curve is known to be steep and the environmental damage curve is not so steep. In practice a mix of regulations could be the best way to control chemical use. For instance, a tax on up-stream consumption to decrease use and promote substitution to other compounds could be combined with labeling at the user stage to inform users about health and environmental impacts and possible actions to mitigate them (e.g., Macauley et al., 1993).

2.2. The political and legal obstacles of implementing chemical taxes

An increased focus on chemicals taxation in many European countries would imply a policy change, and this raises the question of how such a policy change can be pursued and what the main driving forces and obstacles to such a change are. Policy processes are often regarded as logical and linear step-by-step procedures (e.g., assessment of alternatives, recommendation, decision, implementation, and evaluation). However, this view overlooks the dynamics of most policy processes, the time dimension as well as the fact that the process itself will be influenced by different actors (and networks of actors) that are in any way engaged in a specific policy area (e.g., Carlsson, 2000). Sabatier and Jenkins-Smith (1993) develop a policy-analytical method that addresses these concerns. In short their approach relies on an analysis of not only the basic attributes of the policy area and the pros and cons of different policy instruments, but also of to what extent different actors (e.g., organizations, firms, governments, etc.) hinder or support the suggested policy

1 The seminal study on this issue is Weitzman (1974). See also Adar and Griffin (1976) and Stavins (1995).
change. Their model illustrates that the implementation of taxes on chemical compounds is not a straightforward policy process and especially if those that are to be taxed are represented by politically powerful interest groups, who are able to affect not only the design of policy (e.g., Daugbjerg, 1998) but also the problem definitions linking to the policy and (as a consequence) the criteria for evaluating its consequences (Vatn et al., 2002).

An important implication of the above for the case of environmental taxation is that the policy debates concern not only the economic efficiency of the taxes but also distributive and fairness-based arguments. Put bluntly, in the former case the problem definition concerns primarily the incentive impacts of the tax and with little regard for distributive issues, while in the latter case issues concerning the use of the tax revenues as well as possible tax exemptions are often at the forefront of the discussion. For this reason Daugbjerg (1998) notes that political opposition to environmental taxation can be restrained by reimbursing tax revenues to the tax payers, either directly or indirectly by, for instance, subsidizing other activities in the industry. Still, although this ‘earmarking’ strategy often gains a lot of support among the public and the industry – and thus increases the political acceptability of the tax – it can also be questioned both on political and economic grounds. The political problem lies in the fact that earmarking removes funds from parliamentary control, and can thus lead to reduced democratic influence and even increased corruption. Economically earmarking can be inefficient since it does not ensure that the tax revenues are used where their utility is most pressing.

Daugbjerg (1998) also argues that the closer the taxes target the environmental damage involved, the less prevalent will the political opposition be since the tax then has a better chance of being perceived as fair. The search for and the identification of a cost-effective tax policy may thus not only enhance the economic desirability of environmental taxes but it may equally well increase the political legitimacy of the tax policy.

Since most western European countries are members of the European Union, the respective governments may be constrained by community legislation when proposing new taxes as part of environmental policy. In general the implementation of environmental taxes and charges are within the competence of the member states. Still, when a community-wide legislation is absent, environmental taxes should generally be set and implemented so that rates and methods of tax collection are non-discriminating, i.e., imported goods should be treated as domestically manufactured goods. In addition, the tax system must also be proportionate to its objectives; the tax must thus signal a proper balance between the instrument chosen and the objectives of the policy. Thus, overall the tax policies must be in compliance with the rules laid down in the EC-treaty and the criteria laid down by the EC-court. It is also worth noting that since accession to the European Union generally implies less protection and domestic farmers having to compete internationally, the room for implementing stringent unilateral tax policies becomes more limited as price increases then cannot be easily passed on to the consumers.

The main area of conflict is not so much in the implementation of environmental taxes as such, but rather when taxes include national compensation schemes for industry to avoid losses of competitiveness. Such schemes include refund systems, tax reductions and subsidy schemes, and they can be considered state aid (unless explicit exemption has been granted). When community measures exist member states still have the competence to adopt environmental national provisions. However, the competence of member states is in such cases limited and its scope will depend on the content of the provision and of the legal grounds on which the community provision is based. In the case of fertilizer control policy, the EC Nitrate Directive (91/676/EEC) is a community-wide measure for limiting the problems of nitrate leaching, and it applies equally to all member countries. Taxes on fertilizer are neither promoted nor prohibited by the Nitrate Directive; fertilizer taxes can thus complement the Directive but cannot replace the regulations prescribed by it. Still, the difficulties of predicting the impact on environmental quality of the tax may constitute an obstacle towards a tax policy.

3. The experience of fertilizer taxation in Europe

In this section, we briefly discuss some of the relevant practical experiences of fertilizer taxation in five European countries. Cost effectiveness implies that a given environmental goal (e.g., nitrogen leakage) is fulfilled at minimum cost to society. Thus, this concept concerns both environmental goal compliance and cost minimization. For this reason we highlight both the impacts of the taxes on consumption behaviour and ultimately on the environment, as well as the design and the reach of the tax policy. In our assessment of the impact of the tax schemes in the five countries we rely mainly on a comprehensive set of secondary sources, including government investigations and research studies. Some of these past evaluations are summarized in previous work, such as ECOTEC (2001) and Söderholm (2004). Following the discussion in Section 2.2, we comment not only on the design and the outcome of the tax schemes used, but also on the issues related to policy implementation including means of overcoming any political opposition to fertilizer taxation (e.g., earmarking of tax revenues).  

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2 The important point here is that the principle of non-discrimination is applicable to the utilization of tax exemptions as well. This means that selecting certain segments of the business sector and giving them fiscal advantages over other businesses amounts to state aid and is – in the absence of explicit permission – not allowed.

3 An important distinction is that between phosphate and phosphorous. Phosphate (P₂O₅) is the form of phosphorous (P) often used to measure the phosphorous content of fertilizers. Given the atomic-weight proportion, the phosphorous content of a fertilizer is 43% of its phosphate content. The distinction is important to bear in mind since in some countries (e.g., Austria) the taxes are based on kg of phosphate, while in others (e.g., Sweden) they are based on kg of phosphorous.
limited. Nevertheless, according to Hofreither and Sinabell (1998) the tax also raised awareness about fertilizer use and its impacts on the environment among Austrian farmers, and in this way the long-run impacts of the tax may have been more significant. Specifically, the tax had an important signalling effect and reminded farmers of the fact that nitrogen is a cost factor. Thus, Austrian farmers tended to look over their production plans, and part of the decrease in fertilizer consumption can be attributed to efficiency improvements and reductions in the excessive use of fertilizers (Hofreither and Sinabell, 1998). A sizeable part can also however be explained by non-tax impacts, such as changes in production patterns and farmers anticipating higher future prices (ECOTEC, 2001; Bel et al., 2002). Nevertheless, Bel et al. (2002) conclude that overall the Austrian tax had a more significant impact on fertilizer consumption than has been the case in many other European countries, including those surveyed below in this section.

Grain farmers, who were not able to switch to organic fertilizers, experienced the highest income losses as a result of the tax. However, the tax revenues were used to subsidize grain exports easing the tax burden on firms. The impact on the consumers of food products was overall insignificant. The fertilizer industry faced a reduction in competitiveness as the tax led to price increases and substitutes hence became relatively more attractive and consequently industry profits were reduced. Due to the prevailing agricultural import policy in Austria, import penetration was prevented and the producer dominating the fertilizer market was not significantly affected by the tax (ECOTEC, 2001). However, in 1996 as Austria joined the European Union the tax was abolished. It was discriminatory and interfered with the establishment of the single market, and since it was widely believed that the tax had achieved its main objective (raise funds for the grain production sector to subsidize exports) no exception from EU’s state aid rules was sought for (ECOTEC, 2001).

3.1. Austria

In Austria a tax on fertilizers (including potash) was introduced in 1986 with the primary objective of raising funds to support the grain production sector through export subsidies. A secondary goal was however to ensure conservation of the soil through reduced emissions (Hofreither and Sinabell, 1998). The proceeds of the Austrian tax have also been used to stimulate the production of crops, mainly crops that receive their own nitrogen from the air, so called leguminous crops. The initial tax rates were ATS 3.5 (€0.25) per kg nitrogen and ATS 2.0 (€0.15) per kg phosphate, and these were steadily increased until 1994 when Austria joined the European Union and the taxes were abolished. Fig. 1 shows the Austrian consumption of fertilizers (including nitrogenous, phosphate and potash fertilizers) on the left axis, and the average annual tax level on the right axis.

From the time the tax was introduced right up to when it was abolished, total consumption of fertilizers decreased by roughly 3% annually while prices rose in total around 10% (ECOTEC, 2001). There existed no alternative or replacement instrument at the time of the tax system in Austria. The role of the tax in inducing this development can be divided into a direct price effect and a dynamic (long-term) effect on awareness and effectiveness. In 1993 the price elasticity of demand was estimated at −0.2 (Rougoor et al., 2001), 4 and this suggests according to Hofreither and Sinabell (1998) that the observed quantity reduction “could only be expected as the consequence of doubling of fertilizer prices,” (p. 8). This implies that the direct price effect on consumption patterns was considerably lower than the total decrease in use displayed in Fig. 1. According to a groundwater quality monitoring program in place since 1992, the direct environmental benefits of the tax on water quality were also fairly

3.2. Denmark

In 1998 Denmark introduced a tax on nitrogen, and the tax rate was set at DKK 5 (€0.67) per kg nitrogen (on all artificial fertilizers with a more than 2% nitrogen content) (ECOTEC, 2001), and has remained at this level since the introduction. Compared to the Austrian nitrogen tax, the Danish tax level is comparably high. However, some very critical exemptions to the tax exist. Most notably, agricultural and horticultural holdings with a yearly turnover of more than DKK 20000 (€2700) do not have to pay the tax; this implies de facto that all farmers are exempted. Other firms using more than 2000 kg nitrogen fertilizers per year are also exempted from the tax, and for firms using less than 2000 kg nitrogen fertilizers per year the tax revenues can be refunded if total payments exceed DKK 1000 (€135) (Schou, 2003). For the above reasons, the tax is mainly levied on household use (e.g., gardens).

Daugbjerg (1998) notes that Danish farmers have considerable power (compared to, for instance, their Swedish counterparts), and this probably explains the many exemptions in the current tax policy. Taxes on nitrogen appeared on the Danish political agenda in the early 1990s, but in 1994 a government committee “put much emphasis on problems associated with

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4 This estimate is supported in Hofreither and Sinabell (1998). A price elasticity of demand of −0.2 implies that a 10% increase in the fertilizer price induces a 2% decrease in consumption. Compared with many other intermediate products this is relatively low, but of course far from insignificant.
the use of green taxes to reduce nitrate pollution, i.e., lack of cost efficiency, administrative complexity and redistribution of income within the farming community,” (Daugbjerg, 1998, p. 271). This illustrates the importance of addressing distributive issues when implementing a new tax scheme, as well as to target the tax as closely to the harm caused in order to avoid claims about unfair impacts on the farming community.

According to Schou (2003), although households pay a higher price for the use of nitrogenous fertilizers they are generally unaware of the presence of the tax, and the impacts on use have also been limited. This suggests that it may be useful to complement the tax with information campaigns raising awareness of – not only the tax scheme as such – but also of the negative environmental impacts of fertilizer use. The modest tax revenues are not earmarked, so no positive impacts on the environment can be detected from the use of the tax proceeds.

The current Danish tax on nitrogen – with its exemptions – constitutes primarily a complement to the existing ‘control-and command’ regulation of nitrogen use in Danish agriculture (i.e., crop-based quota on nitrogen application). Discussions are however underway about replacing this system by a tax on the nutrient balance for the agricultural sector (Schou, 2003). So far, though, no changes in the design have been made.

In 2005 a new tax on mineral phosphorous when used in animal feed phosphates was introduced, and all tax revenues are recycled back to the agricultural sector through reductions in the land tax for, for instance, farms and nurseries. This is the first tax of this type in Europe (Speck et al., 2006).

3.3. The Netherlands

In 1998 the Netherlands introduced a fertilizer control system that was quite different from the ones used in other European countries, and which essentially targeted nitrogen and phosphate surpluses rather than inputs. The Dutch system was called Mineral Accounting System (MINAS) and the goal was to reduce nitrogen and phosphate losses to a level below certain loss standards (corresponding to EU environmental standards). Under this system every farmer has to maintain records concerning nitrogen and phosphate inputs and outputs. The system represents a combination of a quantitative regulation and an explicit economic incentive (tax). A balance puts. The system represents a combination of a quantitative

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Zeijts et al., 2004; OECD, 2004). Annual environmental expenses for the primary agricultural sector were twice as high during the period 1998–2002 compared to the period 1995–1997 owing to the tightening of fertiliser policy in 1998 (Van Eerdt et al., 2005). The extra costs consisted of higher manure disposal costs (an additional €55 million per year) and higher administrative expenses (an additional €125 million per year). Public spending on implementing the new Fertilizers Act in the Netherlands has mounted from €20 million in 1998 to over €85 million in 2002, out of which a major part can be attributed to MINAS.

The environmental effects of MINAS are still uncertain. MNP-RIVM (2002) concludes that there has been a decline in ground water nitrate concentration over the period 1992–2000, but at this early stage it is difficult to attribute parts of this decline. This is supported in Westhoek et al. (2004), who report that the reduced emissions to the soil have not yet led to a major improvement in environmental quality. Minor improvement can however be seen in nitrate concentrations in the upper groundwater. A significant environmental benefit was however that the number of dairy farms with sandy soils that complied with the EU standard of 50 mg per litre nitrate standard increased from an average of 5% in 1992–1995, to 25% in 1997–1999 and 40% in 2000–2002 (see Table 3). Even so, nitrogen surpluses in sandy soils still need to diminish by 80 kg per hectare in order to achieve the 50 mg per litre nitrate standard for upper groundwater (Van Eerdt et al., 2005).

The MINAS system was abolished in January 2006, as a direct consequence of the EC court decision C-322/00, which stated that the Dutch government had failed to implement certain elements of the EU Nitrate Directive. Westhoek (2005) also notes that the political climate in the Netherlands promoted efforts to make regulations simpler and to have less civil servants. The Netherlands now has a system of nitrogen standards for manure application and total nitrogen and phosphate application comparable to the Danish system (Van Zeijts et al., 2004). This system of use (rather than loss) standards means that farms are no longer assessed on the amount of nitrogen discharged (lost) into the environment (output), but on the amount of nitrogen they use for growing crops (input). The down side is of course that the farms are less flexible in tailoring their management systems to meet the environmental objectives. In other words, while the new system may be easier to manage it will also promote less of cost-effective compliance strategies. To what extent these cost inefficiencies are outweighed by lower administration costs remains, however, an open question and an important issue for further research.

### 3.4. Norway

In 1988 Norway introduced a tax on chemical fertilizers, and from the start the tax was primarily perceived as a way to finance other policy measures. The main purposes of the charge have been to finance environmentally friendly cultivating practices and information measures, while the favorable incentive properties were only a secondary motive for introducing the tax (Nordic Council of Ministers, 2002). In some vulnerable areas, information campaigns were set up. The tax level was expressed as a percentage of the fertilizer price and initially it was only about 1% of the nitrogen price. After an extensive outbreak of algae bloom in the North Sea in 1988, the tax level was raised and corresponded to about 8% of the nitrogen price, but it was still too low to have any important impacts on fertilizer use (Vatn, 2000). Still, the tax raise fueled a debate about how the tax revenues should be used. Most notably, the farmers’ organization wanted to increase its influence over the revenues and therefore allocate them to a specific fund, while the Ministry of Finance argued strongly for channeling the revenues to the general state budget. In the end the Parliament decided to earmark the tax money for environmental programs in the agricultural sector. The revenues from the tax have been “more than fully reimbursed to individual farmers when the environmental subsidies of the agricultural policy are included,” (Daugbjerg, 1998, p. 264).

Proposals for even higher tax levels were published, and the advocates argued that taxes would be a cheap way to reduce nitrate leaching, but that the taxes had to be rather high (100–300% of the nitrogen price) in order to have any substantial environmental effects (Vatn et al., 2002). The tax was raised and in 1991 it reached a level of approximately 20% of the nitrogen price, i.e., at the time about NOK 1.2 (£0.15) per kg N. Prior to this relatively modest increase, the Ministry of Finance argued strongly for the position that if the tax was to be raised substantially, the revenues should be considered

<table>
<thead>
<tr>
<th>Year</th>
<th>Levy free surpluses</th>
<th>Tax rates</th>
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<tbody>
<tr>
<td></td>
<td>Nitrogen (kg N ha⁻¹)</td>
<td>Phosphate (kg P₂O₅ ha⁻¹)</td>
</tr>
<tr>
<td>1998</td>
<td>238</td>
<td>40</td>
</tr>
<tr>
<td>2000</td>
<td>188</td>
<td>35</td>
</tr>
<tr>
<td>2002</td>
<td>165</td>
<td>30</td>
</tr>
<tr>
<td>2003</td>
<td>140</td>
<td>20</td>
</tr>
</tbody>
</table>

Sources: Van Eerdt et al. (2004) and Sjöberg (2004).

* Average of loss standards applied to different types of grass and arable land.

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<tr>
<td>Arable farms</td>
<td>10</td>
<td>30</td>
<td>30</td>
</tr>
<tr>
<td>Dairy farms</td>
<td>5</td>
<td>25</td>
<td>40</td>
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<tr>
<td>Other farms</td>
<td>–</td>
<td>10</td>
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Source: Van Eerdt et al. (2005).
general state income. However, the Ministers of Agriculture and the Environment, respectively, still regarded the tax as a mean of financing other measures in the agricultural sector, and therefore the ‘‘compromise’ was to reduce the tax level, making the conflict about control and responsibility less important,’’ (Vatn et al., 2002, p. 236).

The idea of a substantial increase in the tax level faced strong opposition among some key actors (e.g., agricultural scientists, the Ministry of Agriculture, etc.). The discussion came to centre largely on the fact that the tax did not target the environmental problems as such. Agronomists stressed that the tax could not affect the most polluting activities, and others pointed out that regional differentiation was needed to avoid unfair impacts. The tax was thus argued to be weakly related to the external damage costs; farmers’ costs would be highest for specialized grain producers while nitrogen losses are normally much higher on farms using animal manure. No changes in the tax level occurred between 1991 and 1999 and during this period the tax level remained at €0.15 per kg N. In 2000 the tax was abolished, primarily on the grounds that other measures were claimed to be more effective in reducing nutrient leaching. Thus, in the end the earmarking argument was not sufficient to maintain the nitrogen tax policy. Clearly international trade liberalizations (following GATT) also meant that even a low tax, it was argued, could have negative impacts on the international competitiveness of Norwegian agriculture (Vatn et al., 2002).

3.5. Sweden

In 1984 taxes on fertilizers were introduced in Sweden. The aim of the taxes was to reduce the leakage of nitrogen and phosphorus by reducing the demand for fertilizers, and to finance measures to decrease the negative environmental impacts of the use of chemicals in agriculture. In addition to the tax the there existed also a so-called price regulation charge, whose primary aim was to raise funds for the financing of export subsidies. This charge was, however, removed in 1992. In 1994 the tax on phosphorous was also removed and replaced with a charge on cadmium. The goal of the cadmium charge is to provide incentives to advance better abatement technology, and it is set at SEK 30 (€3.3) for every gram of cadmium that exceeds 5 g per ton of phosphorus. The above taxes are levied on fertilizers manufactured in Sweden and on imported fertilizers. Since 1995, the total tax on fertilizer has been equivalent to approximately 20% of the fertilizer price (SOU, 2003, p. 9). The revenues from the fertilizer tax are considered general state income (although this was a bit unclear as late as in 1995). In practice a large share of the tax revenues have been used to finance R&D measures in the agricultural sector, but the farmers have no control over how the proceeds are spent (Daugbjerg, 1998).

Figs. 3 and 4 show the development of consumption and tax rates for nitrogenous and phosphorus fertilizers in Sweden. The tax rate includes the price regulation charge for the periods during which it was implemented. The consumption of nitrogenous fertilizers has only decreased slightly during the last 20 years while the consumption of phosphate fertilizers has experienced a steady downward trend. With the abolition of the price regulation charge in 1992, nitrogenous and phosphate fertilizer sales increased by 10 and 8%, respectively. However, in 1994 when the tax doubled for nitrogenous fertilizers, consumption levels went back to about the same levels as they were before 1992. At the same time the phosphorus tax was completely removed as the reduction goal of 50% had been met, but consumption still decreased for this type of fertilizer (Nordic Council of Ministers, 2002). The reduction of phosphate fertilizer use after 1994 is partly attributable to the introduction of the cadmium tax; although not implemented to reduce usage of phosphate fertilizer it is believed to have had a negative impact on consumption. It is also worth noting that consumption was at its lowest points for both types of fertilizers in 1991–1992 when the total charge was at its highest level and equalled around 30–35% of the sale price.

The above indicates that the Swedish fertilizer taxes have had real impacts on use, especially the tax on phosphate fertilizers. An evaluation of the impact of the taxes on the use of fertilizers was conducted in 1992. It suggested that the taxes had had some impact on the use of fertilizers and thus on nitrate and phosphate discharges into water; however, the main effect was through the financing of action programs to decrease use (ECOTEC, 2001). Stavins (2000) as well as Jonsson et al. (1997) report that in 1997 the tax on nitrogen had reduced nitrogen demand by about 10%. The relatively modest impact on nitrogen fertilizer use is likely to depend on a low own-price elasticity of demand. Ingelsson and Drake (1998) estimate this to be –0.3, and equally low (and even lower) estimates can be found in other studies (Drake, 1991; Jonsson et al., 1997).
finding and the fact that the relationship between nitrogen input and nitrogen leaching is generally not strong, suggests that in Sweden “a nitrogen tax should be used as only one part of a policy package to reduce eutrophication,” (Ingelsson and Drake, 1998, p. 157).

The effective costs of the taxes on fertilizers are hard to isolate from other factors. In spite of this the agricultural sector has been affected by the taxes in the form of higher costs of fertilizers and since most European countries do not tax fertilizers or have removed their taxes, negative impacts on competitiveness exist (although they are not necessarily significant). Swedish farmers have – due to their limited ability to influence the tax policy processes – not been compensated fully for this loss in competitive strength (Daugbjerg, 1998). A fairly recent government investigation (SOU, 2003, p. 9) concluded that the fertilizer taxes do impose a competitive disadvantage for Swedish agricultural products. In spite of this, however, the assigned investigators proposed that the current taxes (and tax levels) should be retained as they are judged to be environmentally effective.

4. Important obstacles to future implementation of fertilizer taxes

Although the political opposition to fertilizer taxes can sometimes be reduced by redirecting the tax revenues back to the agricultural sector, the foregoing analysis also indicates that additional obstacles to a broad implementation of fertilizer taxes in Europe exist. Earmarking of tax revenues can often “only” ease the overall economic burden of the affected industry, but not necessarily address issues of perceived unfairness as well as any conflicts with other legislative measures. In this section, we discuss briefly to what extent the second-best features of most fertilizer taxes and the EC Nitrate Directive affect the future potential of fertilizer taxation as part of environmental policy in the agricultural sector.

The cost-ineffective features of most fertilizer taxes in Europe are often pronounced. This is most evident in those cases where the tax is levied on the purchase of fertilizer; fertilizer is partly taken up by the crop (and causes thus no environmental damages) and a decline in the use of fertilizer can be compensated by an increase in the use of organic manure. There are substantial differences in the leaching and use of inputs, and the damage from leaching varies because of the receiving environment conditions. Such features represent important obstacles to the increased use of cost-effective taxation schemes. Taxation measures which do not, for instance, address regional differences may still be necessary in order to avoid costly monitoring, but they may be perceived as unfair since farmers who are causing relatively little harm could be taxed higher than farmers with more damaging cultivating practices. Again, this shows that efforts to target the tax – as closely as possible – to the environmental damages incurred may not only benefit the economic efficiency of the tax policy but also the policy’s legitimacy among the affected farmers. Still, as shown in the Dutch case, the dilemma facing policy makers is that these types of efforts will almost always imply high monitoring costs and in this way face opposition from, for instance, the respective Ministries of Finance.

A special case in this instance is ad valorem taxes, i.e., taxes designed as a fixed percentage of chemical prices. In the case of fertilizers these types of taxes are employed in Norway. We have already noted that taxes on physical units may well fail to approximate differential environmental and health impacts, but this problem may be even more accentuated for percentage-based taxes. Most notably, there is no reason to believe that the environmental damages are less pronounced as prices change. As Pearce and Koundouri (2003) observed:

“The risk here is that technological progress in […] fertilizer manufacturing can give rise to price falls, and consequently absolute tax reductions, encouraging more […] fertilizer use.” (p. 3).

Thus, ad valorem taxes will not signal a good correspondence with environmental damage done, this since these damages are not related to the price level. Still, ad valorem taxes are often easy to implement and they do not have to be adjusted to account for inflation (as must an absolute tax whose real value declines with inflation). Also in this case, however, it is possible to foresee that the weak link between tax rate and damage leads to reduced policy legitimacy. If fertilizer prices increase the absolute tax level will also increase even if the environmental damage remains the same.

The Dutch case indicates that the EC Nitrate Directive may (in combination with other EU legislation) de facto impose regulations that make taxation policies less desirable from a national policy maker’s view. The Directive obliges member states to assign areas that are vulnerable to nitrate leaching, establish action plans to decrease leaching, monitor the effectiveness of these plans, and develop codes of good agricultural practice to guide farmers (in particular in regard to manure disposal). In practice taxes on fertilizer use may very well be implemented according to the Directive, but it is important that the tax level is neither too high nor too low. A low tax may – even though part of a broader policy package – lead to non-compliance with the Directive, while a high tax may run the risk of interfering with the single market (i.e., it may not be proportionate).

The EC Court (case C-322/00) did not find the MINAS system to be in non-compliance because of the tax imposed, but we believe it is fair to assert that the specific tax policy used in the Netherlands (although fairly efficient from economic points of view) made it harder for the Dutch to comply with the Directive. The Court primarily found the establishment of loss standards incompatible with the obligation laid down by the Directive, but the Court did not assess explicitly whether the standards under the MINAS system are too high or whether the tax levels imposed when the standards are exceeded are too low. Nevertheless, in practice the risk and the effects of allowing the surplus to be exceeded have to balanced against the consequences for flexibility in reduction strategies. In essence, the Nitrate Directive largely prioritizes risk reduction, goal fulfilment and full compliance with the environmental standards, and with tax solutions it is always hard to ensure full compliance. For instance, according to the Directive it is important to ensure that excess livestock manure is disposed of in a manner that will not be environmentally damaging. In the Dutch case the Court therefore concluded that given the
(too) lax loss standards in the MINAS system, a tax on non-compliance with these standards is not sufficient to remedy the failure to comply with the obligations laid down by the Directive.

The above shows that in the presence of the Nitrate Directive taxes typically need to be complemented with regulations necessary to protect sensitive areas. Thus, in order to avoid costly interference with the European Commission some governments may well find it rational to opt for the implementation of nitrates reduction strategies that in a more direct way meets the requirements of the Directive, rather than to rely heavily on economic incentives. In the absence of this Directive it is easy to anticipate a stronger reliance on economic incentives, perhaps along the lines of the Dutch MINAS system and where loss standards are lower in particularly sensitive regions to deal with the uncertainty problem. The use of fertilizer taxes in the European Union would be spurred on by the implementation of an EU-wide tax. However, the prospects for such a tax solution are probably very slim. The requirement on unanimity makes the adoption of horizontal measures difficult in the Union, not the least since the ambition of member states’ environmental protection policies differ.

5. Concluding remarks and implications

In this paper, we have raised and analyzed a number of issues deemed to be critical for the cost-effective implementation of environmental taxes on chemical compounds, and highlighted their potential importance in the empirical context of fertilizer taxes in Austria, Denmark, the Netherlands, Norway and Sweden. Most of these countries have opted for input taxes, and the outcomes indicate that the taxes have played a role in reducing fertilizer use. Still, the price responses are sometimes judged to be low, and this means comparatively small impacts in terms of quantity reductions. The environmental impacts of the fertilizer taxes have proved hard to assess, primarily since the taxes imposed are not proportional to damage done.

Most of the taxes implemented do not induce a cost-effective allocation of compliance measures since they focus on the consumption or sales of fertilizers rather than on the environmental impacts. This situation is typical for many other tax schemes in the chemicals field, such as the existing taxes on pesticides and solvents in the Nordic countries (Söderholm, 2004). However, in this respect the Dutch system of developing detailed ‘mineral accounts’ for each farm is interesting as it attempts to achieve a closer (although not perfect) proportionality to damage done, but it has also resulted in high administration costs due to the need to regulate animal manure. Similar attempts to tax close to environmental damage are hard to identify for any chemical substance, and research that aims at identifying policy instruments that can induce an optimal balance between compliance and monitoring costs is still very meagre.

The fertilizer tax experiences in Europe provide a number of general implications for other chemical taxation policies. First, our analysis indicates that the choice of tax scheme design matters not only for the cost effectiveness of the policy instrument, but governments can also use policy design to reduce opposition towards environmental taxes. The experience of the European countries in this study indicates that some kind of earmarking of tax revenues can often be effective in increasing the perceived legitimacy of the tax policy, although (as illustrated by the Norwegian case) it may not represent a sufficient condition for political acceptance. To some extent the main impact of the fertilizer taxes have not always rested on the incentives provided by the taxes, but rather on the use of the tax revenues. This is particularly evident when the revenues have been redirected to research and information in which the affected industries have an interest.

Although this runs counter to the public economics arguments against earmarking, one should note that the revenues involved in fertilizer taxation (and indeed in most types of chemical taxation) are minor and (sometimes) of a transient nature. Thus, the disadvantage of earmarking may be a relatively cheap price to pay for gaining some support or understanding and preferably partnership with the concerned industry. Another reason for using tax revenues to further research stems from the fact that the damage from most chemicals are cumulative so that current damage is partly a function of past releases. Water contamination following fertilizer use is a good example of this, but this extends to many other chemicals used in, for instance, solvents and pesticides (Slunge and Sterner, 2001; Söderholm, 2004). If revenues are earmarked, they can be used for clean-up programs and in this way reduce the impact of stock externalities.

Second, the analysis has also illustrated that the more successful policy makers are in designing a taxation scheme that can achieve a closer proportionality to damage done, the less opposition to the tax policy is (ceteris paribus) likely to emerge (at least from those directly affected by the tax). Environmental taxation that targets damages rather than consumption of upstream products tend to promote both cost effectiveness and political acceptance, and we believe it is fair to conclude that so far the attention on such solutions and on how to reduce the associated transaction costs has not been a policy priority. Although governments are likely to pay attention to this problem, most of them quickly opt for the easier option to tax upstream.

One explanation for this is that governments may give undue weight to minimizing monitoring and administration costs (since these are borne directly by government authorities and thus financed by the state budget) even though the total costs to society turn out to be lower with an alternative policy design. Moreover, governments also have incentives to combine fiscal and environmental motives in their taxation policies, and this also calls for broad upstream environmental tax bases. It should also be recognized that taxing close to damage often requires specific monitoring technologies that can measure pollution levels. The development of new technologies – which, for instance, facilitates cheap monitoring of emissions – and innovative policy designs ought to be promoted, but it is somewhat unclear who has the incentive and the willingness to promote and undertake such research and development activities. Private firms cannot be expected to do this. Governments often spend substantial amounts on
funding research on pollution abatement technology but less frequently we see government programs funding research on technology that can facilitate policy enforcement and monitoring.

Finally, our analysis also illustrates that the future implementation of taxes on chemical compounds may run the risk of being in non-compliance with legal obligations, not the least since most legal rules give priority to goal fulfilment rather than flexibility in compliance strategies. The EC Nitrate Directive and the abandonment of the Dutch MINAS system is an illustration of this. Environmental taxes will suffer from the fact that their impacts on environmental quality (and ultimately on goal fulfilment) are hard to assess ex ante since such an assessment requires extensive information about the relevant abatement costs. Given the hazardous nature of many chemicals combined with genuine uncertainty about the marginal abatement costs, it may be motivated to regulate quantities rather than prices. Still, for many other chemicals – including probably those related to fertilizer use – compliance flexibility may be just as important from a policy point of view.

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