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Economic Instruments in Chemicals Policy

Past Experiences and Prospects for Future Use

Patrik Söderholm

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Content

Preface

The Nordic Council of Ministers regularly publishes reports on the use of economic instruments in Nordic environmental policy. The present report investigates the potential for increased use of economic instruments, not the least taxes and charges, in chemcials policy. It discusses some theoretical aspects of such policy instruments, and reviews the practical experiences of existing chemicals taxation in the Nordic and other European countries. A final chapter analyzes the potential role of economic instruments in phasing out three different chemical substances and products.

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Luleå, May 2009

Summary

The objective of the report is to analyze the potential for increased use of market-based policy instruments in chemicals policy. Specifically, the report: (a) provides a conceptual discussion of the role of different market-based instruments in controlling pollution based on chemicals production and use; (b) outlines a comprehensive overview and analysis of the European experiences of taxes and charges in chemicals policy during the last decades; and (c) proposes and evaluates a set of carefully selected economic instruments targeted at specific chemical compounds and products. The cases chosen include the use of: (a) different types of two-stroke oils; (b) the substance nonylphenol (NP) and its ethoxylates (NPEs), which break down into NPs; and (c) ethylene glycol.

In chapter 2 of the report we briefly review the economic-theoretical aspects of policy instrument choice in controlling chemicals use. Here we rely heavily on the lessons from the environmental economics literature and comment in particular on the scope for using different types of economic instruments in the chemicals field. The use of economic instruments is typically motivated by the desire to reduce downstream externalities in the form of harmful impacts on nature, and from an economic efficiency point-of-view it is desirable to target these damages as closely as possible. Still, these non-point source emissions are often very difficult and costly to control, not the least if the environmental damages caused by the substances are affected by geographical location and the receiving environment's condition. It may thus be more effecient to tax the production or the use of chemical compounds upstream in the product chain.

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. The literature also recognizes that the existence of major uncertainties about abatement costs is critical for the choice between different environmental policy instruments. Emphasis may then be put on controlling *quantities* rather than *prices,* especially if the (marginal) environmental damage rises steeply compared to (marginal) reduction costs. This implies that command-and-control regulations or a mix of quantitative regulations and economic incentives could be the best ways to control chemical use.

Still, in many instances economic incentives such as input taxes could work well in isolation. Moreover, a deposit refund system for used chemicals also represents one mechanism to consider. Handlers or recyclers then pay a deposit when accepting the compound, a deposit that is returned as a refund at proof of recycling or legal disposal. This raises the costs of illegal disposal and rewards appropriate disposal or recycling.

We also pay attention to the policy aspects of implementing incentive-based instruments. These issues include policy legitimacy and acceptance and the public economics issues raised when tax revenues are earmarked for specific purposes. 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 (e.g., R&D efforts) 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.

Previous research shows that behavioural change following a price increase may differ depending on whether the change is a result of an environmental tax increase or simply a regular market effect. Thus, taxes may not just signal economic information but also moral concerns about the personal responsibility to contribute to the provision of public goods. In evaluating different policy designs for the management of chemical risks, it is useful to pay explicit attention to how these types of signalling – or attention-seizing – effects can be achieved and even strengthened. Economic instruments and information are thus essentially policy complements rather than substitutes.

In chapter 3 of the report we review and survey the experiences of economic instruments in chemicals policy in different European (and in particular in the Nordic) countries. Two groups of chemicals are covered in the analysis; chemicals used in agriculture (fertilizers and pesticides), and hazardous chemicals (in particular chlorinated solvents). In practice the most commonly used economic instrument is an input tax (or fee). Overall the analysis illustrates the trade-off between costly monitoring on the one hand and the achievement of a cost-effective allocation of abatement measures on the other. Most of the taxes implemented in practice are second-best instruments as they focus on the consumption or sales of the respective chemicals. Still, the Dutch tax on fertilizers and the Norwegian taxes on pesticides show that there exist ways in which the tax system can be designed to achieve a closer proportionality to damage done.

The analysis has also illustrated that the more successful policy makers are in designing an economic incentive scheme that can achieve a closer proportionality to damage done, the less opposition to the policy is (ceteris paribus) likely to emerge (at least from those directly affected by the policy). For instance, environmental taxation that targets damages rather than consumption of upstream products tends to promote both cost-effectiveness and political acceptance. However, 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.

The experiences from the European countries indicate that some kind of earmarking of tax revenues can be effective in increasing the perceived legitimacy of the policy. To some extent the main impacts of some of the 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 most types of chemical taxation are minor and often of a transient nature. Thus, the disadvantage of earmarking may be a relatively cheap price to pay for gaining support from – and preferably partnership with – the concerned industry. Another reason for using tax revenues to further research stems from the fact that the damages from most chemicals are cumulative so that current damage is a function of past releases. If revenues are earmarked, they can also be used for clean-up programs and in this way reduce the impact of stock externalities. Still, earmarking can never be an excuse for the implementation of otherwise inefficient economic instruments.

A key output of the report is the analysis of three cases of chemical use in which the implementation of economic instruments potentially could help reduce negative health and environmental impacts. In chapter 4 we discuss – for each of the cases – the nature of the chemicals use, the characteristics of the involved suppliers and consumers, as well as potential designs of different economic instruments. The chapter also outlines brief impact assessments of the key policy instruments identified. The importance of policy design concerns both the economic impacts of the instruments as well as the legitimacy of the policy. The cases should not be look upon as first-best cases that deserve to be prioritized in future policy decisions; rather they have been chosen so as to illustrate different types of challenges in relying more extensively on economic instruments in chemicals policy. In each of the three cases there are clear challenges in the implementation of the proposed instruments; overall these are linked to problems of implementing taxes and charges which can closely target the relevant environmental impacts.

In the NPE case, it is difficult given the existing scientific knowledge to identify an economic instrument that can properly address the NPE-content of imported products (clothes, textiles, cleaning agents etc.), one of the most important sources of NPE in Sweden and the other Nordic countries. This problem deserves additional research before efficient policy instruments (above information campaigns) can be put in place. An input tax based on the amount of NP and NPE used in the production of products and emission charges are discussed; these could be potentially interesting instruments in countries in which NPE use is still significant in important domestically produced products and point sources. In principle an input charge could be placed on all sectors that use either NPs or

NPEs as raw materials, including the sectors for which the Marketing and Use Directive applies. However, the charge should also cover sectors where few specific measures exist, such as electrical engineering applications, fuel and oil additives, photographic materials etc.

A differentiated taxation of two-stroke oils represents a rather promising policy measure in addressing the water pollution and health impacts from the use of oils in two-stroke engines, and is arguably more efficient than both a scrap premium on two-stroke engines and a tax rebate. The tax policy is also fairly easy to implement and given the existence of readily available (greener) substitutes the compliance process ought to be smooth. No significant economic impacts on the user collective (e.g., owners of private boats and snowmobiles) are expected. The challenge in this case probably lies in achieving significant consumer responses following the tax increase, i.e., decreased demand for regular – non-biodegradable – two-stroke oils, given the low share of two-stroke oils in household budgets. For this reason it probably becomes essential to combine the explicit economic incentive with information campaigns about the impacts on, not the least, the marine environment.

The final case represents an output tax on the use of ethanediol-based glycol and it is also straightforward, given the availability of a reasonable substitute (prophanediol-based glycol). Still, in this case the health problems following oral ingestion of ethanediol are diffuse and differ widely across different consumer groups. Unlike the two-stroke oil case, in which the environmental impacts are fairly strongly correlated with the consumption of the product in question, the health impacts of one litre of ethanediol-based glycol will be zero in many cases. This suggests that the implementation of a uniform per litre tax on this glycol only is a second-best policy instrument that does not promote a cost-effective reduction in oral ingestion incidents. Still, due to the limited impacts of previous information campaigns and bottle design recommendations, a tax could represent a first step towards a more permanent reduction of this problem. Also in this case, though, informative policy instruments should constitute important complements to the economic incentives provided by a tax.

In sum, an important condition for future successful use of economic instruments in Nordic chemicals policy is that damages done are targeted as closely as possible (without however imposing high administrative costs). In many cases, instruments that address the use and/or production of chemical substances or products are reasonably cost-effective second-best measures; it is however important to avoid, for instance, environmental taxes that have mainly fiscal rather than incentive-based impacts. The implementation of economics instruments on chemcials use must also be preceeded by a detailed account of the likely substitiution behaviour, including an environmental assessment of of the substitutes.

1. Introduction

1.1 Background and motivation

Environmental policy in the 1970s was mainly based on “command-and control” (CAC) regulations, such as firm-specific emission limits and mandatory technology requirements. However, since the late 1980s policy makers have paid increased attention to market-based policy instruments, such as environmental taxes and charges, tradable permit schemes, deposit refund systems etc. In some policy areas, though, the use of these incentive-based policy instruments is still limited. Chemicals policy represents one of those areas. In Europe about 6–7 per cent of the total tax revenues are environmentally related and over 90 per cent of the environmentally related taxes are applied within the energy and transport sectors (EEA, 2000). Less than 5 per cent 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 policy makers 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 this 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), Sweden (SEPA, 2004) 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, but overall the interest in other types of market-based instruments (e.g., deposit-refund systems, tradable allowance schemes) as complements to the existing regulations in chemicals policy has been limited.

The research literature has also devoted attention to the potential efficiency advantages of increased use of, for instance, taxes on chemical compounds and products as a complement to existing CAC regulations (e.g., Slunge and Sterner, 2001; Söderholm and Christiernsson, 2008; Hammar and Drake, 2007; Swedish Chemicals Agency, 2008c).[1] These previous studies point out that the economic rationale for taxing chemical

[1] See also Roos (1999), Malcolm et al. (2007), and Welker-Hood et al. (2007) for additional studies arguing for increased use of economic instruments in chemicals policy.

compounds is related to the internalization of negative externalities in the form of hazardous emissions and waste disposal. It is however difficult to regulate the damage at the point where it arises, and for this reason taxes on chemical inputs (or the production of chemicals) may be good second-best alternatives. The overall contributions of much of the earlier policy-oriented work in this field are also to: (a) provide a review of the (fairly limited) experiences of chemical taxes in the OECD, and in the Nordic countries in particular (e.g., Söderholm, 2004; Speck et al., 2006); and (b) identify a number of important issues that have to be addressed before specific market-based policy instruments are selected and implemented in the chemicals field (e.g., Hammar and Drake, 2007). Nevertheless, even though the above studies stress the importance of evaluating the effectiveness of market-based policy instruments on a case-by-case basis, little is said about specific policy proposals and the use of different economic instruments in the case of carefully selected chemicals and/or products (see, however, Swedish Chemicals Agency, 2008c). This report targets these gaps in the literature on chemicals policy.

1.2 Objectives of the report

The overall objective of the present report is to analyze the potential for increased use of market-based policy instruments in chemicals policy. Specifically, the report: (a) provides a conceptual discussion of the role of different market-based instruments in controlling pollution based on chemicals production and use; (b) outlines an extended overview and analysis of the European experiences of taxes and charges in chemicals policy during the last decades; and (c) proposes and evaluates a set of carefully selected proposals for economic instruments targeted at specific chemical compounds and products.

Concerning the last point outlined above, the report identifies three cases in which economic instruments could successfully complement other regulations restricting use and reducing any negative impacts. The cases chosen include the use of: (a) different types of two-stroke oils; (b) the substance nonylphenol (NP) and its ethoxylates (NPEs), which break down into NPs; and (c) 1,2–ethanediol (ethylene glycol). The selection of these products and substances are further motivated in section 4.1. In brief these cases represent substances and products which have proved to be harmful to human health and the natural environment, and they highlight different implementation challenges in the use of economic instruments – mainly taxes – in chemicals policy. These challenges include, for instance, the availability of attractive substitutes and the imports of specific substances in consumer products. The ambition is to present an in-depth analysis of these cases, which ultimately can form the basis for future decisions on policy instrument selection in chemicals policy.

1.3 The Nordic approach to chemicals regulation

This report devotes the main attention to the experiences and the future use of market-based policy instruments in the chemicals field in the Nordic countries, Sweden, Finland, Norway and Denmark. These countries have long been forerunners in international chemicals policy. Their concerns have involved the contamination of waterways caused by persistent and bioaccumulative pollutants as well as chemical exposures from everyday products. Two basic principles underlie the chemicals initiatives in the Nordic countries: (a) the substitution of safer products for potentially harmful products; and (b) precautionary actions even though the nature and magnitude of risks are not fully known (e.g., Karlsson, 2006).

The policies implemented in the Nordic countries include a variety of regulatory and voluntary tools to reduce hazardous chemical risks, involving education, technical assistance, taxes and fees (see also chapter 3), as well as chemical phase-outs. The focus has been on the develop-ment of long-term goals and action plans for reducing the impacts of broad classes of chemicals, as well as rapid screening processes to prioritize chemicals for reductions (Lowell Center for Sustainable Production, 2003). For instance, one of the main goals of the 1999 "Swedish Environmental Quality Objectives", which outlines 16 environmental quality objectives to be attained by 2020, is to achieve "a non-toxic environment".

Nordic policies have also been characterized by the establishment of lists of chemicals of concern to guide business and government decision-making. Government authorities then work with business and procure-ment agencies to assist them in avoiding these chemicals. Some countries are providing technical support and initiating demonstration projects on alternatives as critical steps in developing safer alternatives. These measures have made the phase-out objectives, sometimes enforced by bans, attainable in practice.

Finally, it should also be noted that the Nordic countries have spearheaded several international initiatives to more effectively protect their environment, and to harmonise international standards. These programs have had an important influence in the development of, for instance, the REACH process (see also section 2.4).

1.4 Scope and definitions

Before proceeding, a number of important limitations of the study need to be stressed. Scholars typically provide a rather broad definition of *market-based policy instruments* such as, for instance, "regulations that encourage behaviour through market signals rather than through explicit directive," (Portney and Stavins, 1990). Typically, this definition embraces a wide range of different instruments, taxes, fees, tradable allowance schemes but also information-based instruments such as eco-labels and campaigns. Information is, together with command-and-control approaches, a commonly used instrument in chemicals policy. In this report, however, we focus on the case of *economic instruments*, i.e., those that make explicit use of *price* signals to producers and consumers. As has been noted above, these instruments are so far rarely used to control chemicals use and production. This choice of scope is not meant to downplay the role of information in chemicals policy. We believe that information is crucial in this sector (see also, Hammar and Drake, 2007), but it must be acknowledged that economic instruments also provide information, not only about the economic cost and benefits associated with a particular use but also, in some cases, signalling effects about the characteristics of the underlying good (e.g., Ghalwash, 2007). In addition, economic and information-based policy instruments can serve as important complements, at least if presented as coherent policy packages.

The use of economic instruments is motivated for several reasons (e.g., fiscal motives, cost-covering charges), but in this report we focus solely on those economic instruments that aim at influencing environmentally or health damaging behaviour (e.g., reducing pollution, waste etc.). Still, the revenue-raising character of some economic instruments may be of interest also from environmental and policy acceptance points-of-view (see also chapter 2) (Hammar and Drake, 2007; Söderholm and Christiernsson, 2008). For this reason a distinction should be made between *taxes* and *fees (or charges)*. Taxes are essentially compulsory unrequited 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. The latter approach may significantly reduce the opposition towards increased use of economic instruments in environmental and health policy, and also be used for funding R&D efforts and/or clean-up activities.

Finally, although we highlight the potential for increased use of economic instruments in chemicals policy from a Nordic perspective, it should be noted that in the discussion on specific proposals for new economic instruments (chapter 5), most of the attention is paid to the Swedish situation. This is mainly due to the limited data availability for the other Nordic countries in the cases chosen. Still, this part of the report comments briefly on the situation in these other Nordic countries, and stresses in particular that the problems faced in Sweden are equally relevant for these countries.

1.5 Approach and outline of the report

In line with the objectives of the study, this report comprises three major parts. Chapter 2 discusses briefly the economic-theoretical aspects of policy instrument choice in controlling chemicals use; here we rely heavily on lessons from the environmental economics literature and comment in particular on the scope for using different types of economic instruments in the chemicals field. We also pay attention to the policy aspects of implementing incentive-based instruments such as policy legitimacy issues and the public economics issues raised when tax revenues are earmarked for specific purposes.

In chapter 3 we outline the current use of economic instruments, primarily taxes, in the chemicals field in the OECD area. In this survey we follow OECD (1997) and make an important distinction between *ex ante* and *ex post* analyses of environmental taxes. *Ex ante* evaluations are made in advance based on economic modeling and simulations, and are best described as conditioned scenarios (Skou Andersen et al., 2001). The *ex ante* analyses referred to in this report are in most cases based on economic models that capture the essence of an economy or a market, and with whose help one can identify possible outcomes if specific policies are implemented, and compare these to some predetermined criteria. An important feature of *ex ante* evaluations is that they permit a consistent comparison between different policy designs and instruments. *Ex post* evaluations, on the other hand, are based on historical analyses of the actual effects of existing or previously implemented policies. The *ex post* evaluations surveyed in this report tend to rely on secondary data and compare environmental outcomes before and after the introduction of the policy, in most cases taxes on products or substances.

Chapter 4 presents and motivates the three case studies selected for a more in-depth analysis of the potential application, design and consequences of different types of economic instruments. The analysis involves both an economic analysis of the effectiveness (environ-mental and health impacts) and the cost-effectiveness of the proposed policy instruments (including administrative costs), and to some extent a political and legal analysis of possible obstacles to their implementation. The markets for the chosen chemicals and any related products will be analysed. It is of particular importance to assess the demand and supply situations (including imports and exports). Furthermore, the substitution possibilities – including both imports and product substitution – as well as the competitive situation for the producers will also be assessed.

An important tool in this research endeavour is the products database maintained by the Swedish Chemicals Agency; it includes more than 120000 products, including information about production, exports and imports since (at least) the early 1990s. Supplementary data for the remaining Nordic countries (Denmark, Finland, and Norway) have been retreived from the so-called SPIN database, maintained by the Nordic Council of Ministers.

Finally, chapter 5 provides some concluding remarks and implications from the investigation, and outlines a number of recommendations for the future implementation of economic instruments in chemicals policy.

2. The political economy of environmental management in chemicals policy

2.1 Introduction

Economic instruments targeted at chemical compounds and products are often perceived controversial, and traditionally the use of chemicals has mainly been subject to command-and-control regulations (including bans) and information-based policy instruments. In the introduction to this report, though, we noted that during the last decade the policy interest in environmental taxes on chemical compounds has increased. Nevertheless, the present policy suggestions are still strongly characterized by traditional regulations.

A recent report from the Swedish Chemicals Agency (2008a) analyses the so-called product choice principle, and how its use can be developed and intensified. The principle (as expressed in the Environmental Code of Sweden) states, roughly, that economic agents (e.g., companies and households) should in their use of chemical products substitute less-damaging products for those which impose greater danger to human health and the natural environment. Clearly, this requires substantial monitoring and enforcement activities, and in a concluding section the authors of the report state that such activities cannot alone induce environmentally benign product substitution in private companies and complementary policy instruments are therefore necessary. Still, economic instruments are never mentioned in the report; instead the authors solely discuss the role of public procurement, R&D activities and information. They note, for instance:

> Apart from monitoring activities other policy instruments and actions are needed to induce companies to employ the product choice principle. Knowledge about chemicals and their hazardous features is of great importance. Companies that gain knowledge about chemical compounds in products can often take further action to reduce risks by replacing a hazardous substance with another or by choosing a less damaging product. […] This can be achieved by developing tools that support companies or provide recommendations for use. (author's translation) (Swedish Chemicals Agency, 2008a, p. 11).

Also in the case of households a lot of attention is paid to the role of information in inducing product substitution; the authors even stress that legal rules about information on the chemical content in different products is the *most effective* policy instrument to enable consumers to influence product substitution (Ibid.). In this report we challenge this ”conventional wisdom”, not the least due to heavy cognitive burdens put on the individual consumer to process this information.

Some analysts maintain that, for instance, taxes on chemical inputs can be just as desirable (e.g., Macauley et al., 1993¸ Rendleman et al., 1995; Slunge and Sterner, 2001). In brief, policy instruments based on information campaigns are too toothless, and the traditional command-and control approach limits the flexibility in using different compliance measures. Economic incentives often have better track records in these respects. It is also important to acknowledge the importance of the legitimacy of the policy, and also here economic instruments can perform better than the traditional approach (Slunge and Sterner, 2001).

From an economic efficiency point of view the implementation of economic instruments in the chemicals field can be motivated if: (a) there exists a market failure, thus a situation in which market forces alone cannot provide enough of environmental goods and quality; and (b) other environmental regulations (e.g., technology standards, emissions limits etc.) are less efficient than, for instance, taxes or charges. In this chapter we address both these issues, and we also devote attention to the political economy aspects of economic instruments in chemicals policy (e.g., use of tax revenues, policy acceptance etc.). Before proceeding, however, some comments on the key characteristics of chemical substances and their environmental impacts are in order. Overall the discussion in this chapter provides a necessary starting point, both for the analysis of past experiences of economic instruments in chapter 3 as well as for the evaluation in chapter 4 of new policy proposals for chemical substances and products previously regulated through other means.

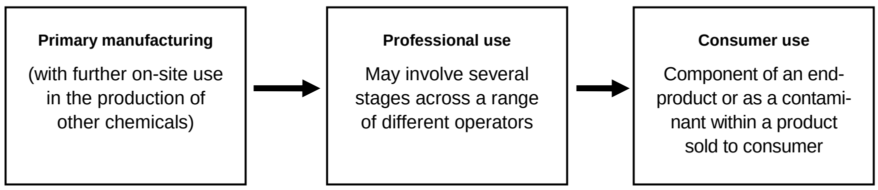
2.2 Key characteristics of chemical pollutants

The characteristics of environmental pollutants and their sources can heavily affect the choice of regulatory measures, and ultimately if these approaches are accepted by the government and the relevant stakeholders. We will here focus on four important characteristics.

First, different chemical pollutants differ with respect to the degree to which they persist, bioaccumulate and are toxic. The persistence of a chemical relates to how long it will remain in the environment before it biodegrades (or is transformed). This implies that persistent pollutants will continue to cause damages to varying degrees, depending on the assimilative capacity of the environment, even if emissions have ceased. Bioaccumulative pollutants are those that, through their presence in the aquatic environment, sediments or soils, build-up in flora and fauna. A chemical substance that is both persistent and bioaccumulative will pose an even greater potential for damages to continue beyond the length of time over which the pollutant is emitted to a given medium. Finally, the more toxic the substance, the greater the potential harm over the long-run.

Second, many chemicals have complex life-cycles associated with their production, use and disposal (see Figure 2.1), and negative environmental impacts can arise at different stages along the product chain (each also involving disposal of waste products). It is important to consider the full range of possible means of chemicals entering the environment, if the aim is to achieve risk reduction in the most cost-effective manner (see also section 2.3). Moreover, even though there may be several sectors of use, chemical activity may only pose danger to the environment in a limited selection of these.

Figure 2.1 The life-cycle of chemical compunds



Source: RPA (2002a).

Third, some pollutants stem from point sources (e.g., releases from production and processing facilities), while some come from diffuse sources (e.g., consumer use such as paint products and detergents). As discussed in the next section these characteristics affect the ability to closely monitor emissions, and thus influence the choice of policy instrument (Vatn, 1998). Related to the above is the extent to which there exist significant spatial variations in environmental and health damages. Ultimately this implies that for many chemicals it can be very difficult to design policies that closely target damages done without incurring substantial monitoring and transaction costs. For instance, in many cases the damages depend on the conditions of the receiving environment (e.g., nitrogen).

Fourth and finally, the level of uncertainty surrounding the risks posed by a chemical and the frequent lack of data on the risks posed by relevant substitute products or processes will also influence policy choices, and we return to this issue below.

2.3 The economics of environmental management of chemicals use

The use of economic instruments is typically motivated by the desire to reduce downstream externalities in the form of harmful impacts on nature, and from an economic efficiency point-of-view it is desirable to target these damages as closely as possible. In practice this means, for instance, that emission taxes (or a corresponding emissions allowance scheme) will perform better than taxes on the production or use of a chemical substance. Nevertheless, these non-point source emissions are often very difficult and costly to control, not the 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.). The targeted chemical may exist in many products as well as in a wide variety of end uses. It may thus be more effecient 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 (at least in part) 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 (see also chapter 3). 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 (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. 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.[2] 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.[3] 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, deposit refund systems for used chemicals could be one mechanism to consider. Handlers or recyclers then pay a deposit when accepting the compound, a deposit that is returned as a refund at proof of recycling or legal disposal. This raises the costs of illegal disposal and rewards appropriate disposal or recycling. Following the above, a system like this could also be incorporated into an excise tax on sales, as a way of reducing the administrative costs of the scheme.

Proceeding along the same lines we can note that Dinan (1993), Fullerton and Wu (1998), Walls and Palmer (1997, 2001) and Palmer and Walls (1999) argue that a tax on virgin materials (i.e., chemicals use or production) cannot correct for the external costs resulting from waste disposal. These authors find that no single tax can generate the optimum level of both downstream and upstream waste disposals and that multiple policy instruments are necessary to fully internalize the externalities. Palmer and Walls (1999) argue that a combination of a tax on intermediate goods and recycling subsidy would provide the appropriate incentives. This system is thus very similar to a deposit-refund system in which the deposit acts like a tax on the virgin material, while consumers who recycle get the refund back.

Furthermore, a tax on up-stream consumption to decrease use and promote substitution to other compounds could be combined with labelling at the user stage to inform users about health and environmental impacts and possible actions to mitigate them (e.g., Macauley et al., 1993).

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[2] This can however be addressed by trial and error, i.e., *ex post* adjustments in the tax rate.

[3] The seminal study on this issue is Weitzman (1974). See also Adar and Griffin (1976) and Stavins (1995).

2.4 The politics of environmental management of chemicals use

An increased focus on the use of economic instruments in the chemicals field in Europe would imply a major 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 the above 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 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. Another reason for using tax revenues to further research or clean-up activities stems from the fact that the damage from most chemicals are cumulative so that current damage is a function of past releases. 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 the highest and/or where national priorities currently are the most pressing. Of course, there is a clear trade-off between using the tax for revenue-raising purposes and for providing incentives to reduce chemical use. Put bluntly, one important reason for why the use of the tax revenues in many cases have led to substantial environmental impacts (see chapter 3), is because the incentive effects of the tax have been very modest.

Daugbjerg (1998) 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 (Ibid.). 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. It is also worth noting that since accession to the European Union generally implies less protection and domestic producers 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).[4] 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. For instance, 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

[4] 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.

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. The latter concern did play a critical role in the abandonment of the Dutch fertilizer tax in 2006 (see further section 3.2.3).

An important component of EU chemicals policy is also the chemicals management scheme REACH. It involves first of all the registration of all chemicals in commerce marketed over one metric ton per year. Manufacturers and importers of chemicals will over time be required to submit a registration dossier, and also downstream users will be required to provide information under the scheme. The information includes, for instance, properties of the substances, intended uses and exposures as well as hazard classifications. Furthermore, chemicals produced over 100 tons per year (about 5000 substances) and those of particular concern will undergo an evaluation process. Chemicals of great concern based on their inherent hazardous characteristics will have to undergo an authorization process to continue their use. This will be made on a case-by-case basis considering socio-economic impacts, necessity, substance risk for that use, as well as economic and technical feasibility.

These evaluations can result in proposals for accelerated risk management measures, including restrictions and bans but also different types of economic instruments. The latter are explicitly mentioned in the REACH guidelines as one strategy to implement when other means are not economically feasible (by, for instance, leading to restricted uses of the substances relocating outside the EU).

2.5 Economic incentives, information and signalling effects

Information is a standard policy instrument in chemicals policy, but it has not always proven to be effective in inducing behavioural change (see, for instance, section 4.4). A stronger reliance on economic instruments in chemicals policy should not imply that less emphasis is paid to information. On the contrary, economic instruments and information campaigns can be combined in comprehensive policy packages, not the least if the revenues from any taxes are earmarked to improve environmental quality or promote research. Moreover, taxes and other economic instruments may also work via their attention-seizing function (Speck et al., 2006). Taxes focus attention of industries and consumers on the options and costs of reduced environmental impacts.

Following this idea, recent empirical research (e.g., Ghalwash, 2007; Berkhout et al., 2004) shows that behavioural change following a price increase differs depending on whether the change is a result of an environmental tax increase or simply a regular market effect. Thus, taxes may not just signal economic information but also, we argue, moral concerns about the personal (or indeed corporate) responsibility to contribute to the provision of public goods. Furthermore, Hilden et al. (2002) investigate the Finnish pulp and paper and chemical industries, and conclude that the implementation of an environmental tax prompted firms to enter voluntary agreements.

In evaluating different policy designs for the management of chemical risks, it is useful to pay explicit attention to how these types of signalling – or awareness-raising – effects can be achieved and strengthened. Economic instruments and information are essentially policy complements rather than substitutes.

3. Experiences of environmental taxes in European chemicals policy

3.1 Introduction

In chapter 2 we argued that economic instruments for selected chemical compounds and products can represent relatively efficient policy instruments to reduce negative health and environmental impacts. In this chapter we review and survey the experiences of such instruments in different European (and in particular the Nordic) countries. Two groups of chemicals are covered by the analysis; chemicals used in agriculture (fertilizers and pesticides) (sections 3.2–3.3), and hazardous chemicals (in particular chlorinated solvents) (section 3.4). In practice the most commonly used economic instrument is an input tax (or fee). Such a tax represents one way of reducing pollution from non-point sources when direct emission control measures are difficult and costly and the implementation of first-best instruments thus becomes unfeasible (e.g., Romstad, 1999). In the analysis we devote quite a lot of attention to the case of fertilizer taxes, partly since this represents a case where the impacts of earmarking, signalling effects and earmarking of tax revenues are well illustrated. Section 3.5 provides some summarizing remarks and implications for the future use of economic instruments on chemical compounds and products.

3.2 The experience of fertilizer taxation in Europe

Nitrogen and phosphorus fertilizers both contribute to eutrophication. This comes about when nutrients, especially nitrogen and phosphorus, and sediments are deposited in surface waters such as streams and lakes through sedimentation and erosion and then reach coastal and marine waters. Eutrophication of coastal waters is caused by excess nitrogen loadings, while eutrophication of freshwaters is caused by excess phosphorus loadings. Phosphorus fertilizers also contain cadmium. This heavy metal accumulates in soils and groundwater, and could cause cardiovascular diseases, liver problems and reproductive disorders. Another problem with nitrogen is that it can reach groundwater through a leaching process; high concentration of nitrates in drinking water is believed to have negative health effects.[5]

In this section we discuss some of the relevant practical experiences of fertilizer taxation in five European countries. Since cost-effectiveness implies that a given environmental goal (e.g., nitrogen leakage) is fulfilled at minimum cost to society it is useful to comment both on environmental goal compliance and cost minimization. Thus, 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. Furthermore, 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). The assessment relies 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).

Before proceeding, however, it is useful to take a brief glance at the literature attempting to model the economic impacts of different types of fertilizer tax designs. Table 3.1 summarizes the scope, approach and results of selected simulation studies of the efficiency impacts of different fertilizer tax schemes. Since the environmental damages caused by the substances are affected by geographical location and the receiving environment's condition it may be very difficult to control emissions cost-effectively, and the simulation 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. As is shown in the next sub-section input taxation is the most commonly used approach in practice.

[5] An important distinction is that between phosphate and phosphorous. Phosphate (P_2O_5) is the form of phosphorous (P) often used to measure the phosphorous content of fertilizers. Given the atomic-weigh proportion, the phosphorous content of a fertilizer is 43 per cent 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.

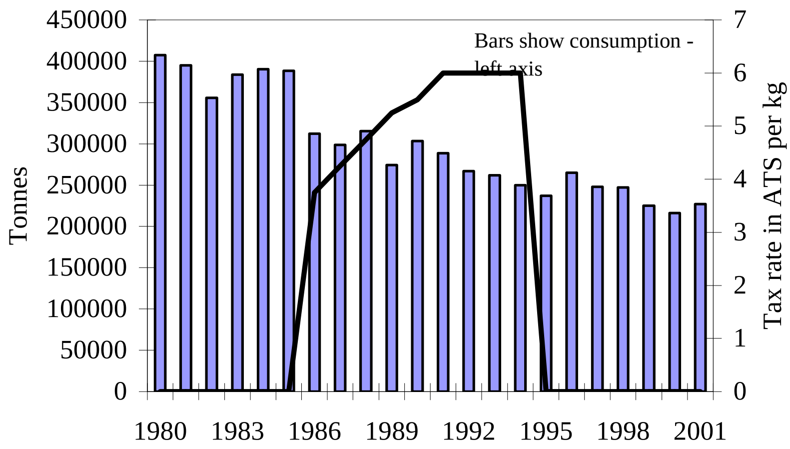
Table 3.1 Selected simulation studies of fertilizer taxation schemes

Study and geographical area	Purpose and methodological approach	Main findings and implications
Kim et al. (1999) (Nebraska, USA)	A competitive dynamic model is used to examine the type of tax rate that minimizes compliance costs while maintaining a predetermined groundwater quality level.	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.
Brännlund & Gren (1999) (Sweden)	Employs a short-run partial equilibrium analysis to estimate fiscal incomes and environmental impacts from increases in the Swedish nitrogen fertilizer tax.	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.
Fleming & Adams (1997) (Oregon, USA)	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.	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.
Huang & LeBlanc (1994) (Corn Belt states, USA)	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.	The impacts on corn production and nitro-gen reduction of the tax vary across state; however, the residual nitrogen tax is claimed to be more cost-effective.
Helming (1997) (The Netherlands)	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.	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.
Kleinhanss et al. (1997) (Europe)	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.	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.
Berntsen et al. (2003)	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.	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.
Rorstad (1999) (Norway)	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.	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.

3.2.1 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 (EUR 0.25) per kg nitrogen and ATS 2.0 (EUR 0.15) per kg phosphate, and these were steadily increased until 1994 when Austria joined the European Union and the taxes were abolished. Figure 3.1 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.

Figure 3.1 Total annual fertilizer consumption and tax rates in Austria, 1980–2001*



**Total fertilizer consumption refers to the amount of the nutrients nitrogen, phosphate and potash consumed. During the time period of the tax policy, ATS 1 equalled about EUR 0.07.*
Sources: EEA (2003) and Sjöberg (2004).

From the time the tax was introduced right up to when it was abolished, total consumption of fertilizers decreased by roughly 3 per cent annually while prices rose in total around 10 per cent (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),[6] and this suggests according to Hofretiher and Sinabell (1998) that the observed quantity reduction "could only be expected as the consequence of a 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

[6] This estimate is supported in Hofretiher and Sinnabel (1998). A price elasticity of demand of -0.2 implies that a 10 per cent increase in the fertilizer price induces a 2 per cent decrease in consumption. Compared with many other intermediate products this is relatively low, but of course far from insignificant.

in Figure 3.1. According to a groundwater quality monitoring program in place since 1992, the direct environmental benefits of the tax on water quality were also fairly 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 (Ibid.). 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 has had a more profound 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 (Ibid.).

3.2.2 Denmark

In 1998 Denmark introduced a tax on nitrogen, and the tax rate was set at DKK 5 (EUR 0.67) per kg nitrogen (on all artificial fertilizers with a more than 2 per cent nitrogen content) (ECOTEC, 2001), and has remained at this level since the introduction. Compared to the Austrian nitrogen tax, the Danish tax level is fairly high. However, some very critical exemptions to the tax exist. Most notably, agricultural and horticultural holdings which are registered in the farmer's mandatory fertilizer account do not have to pay the tax, since they are regulated by individual non-transferable quotas (see last paragraph in this section); this implies *de facto* that all bigger farms are exempted and the tax is targeted towards use of nitrogen at smaller farms and in other sectors. Firms using more than 2,000 kg nitrogen fertilizers per year for use in industrial production are also exempted from the tax, and for firms using less than 2,000 kg nitrogen fertilizers per year in industrial processes, the tax revenues can be refunded if total payments exceed DKK 1,000 (EUR 135) (Schou, 2003). For the above reason, the tax is mainly levied on household use (e.g., gardens).

Daugbjerg (1998) notes that the Danish farmers can exert considerable political 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 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," (Ibid., 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 to avoid "unfair" impacts on the farming community.

According to Schou (2003), although households pay a higher price for the use of nitro-genous 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 ear-marked, 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. So far, though, no changes in the design have been made. Still, 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).

3.2.3 Finland

A fertilizer tax in Finland was implemented in 1976 and later abandoned in 1995 when the country joined the European Union. Initially this tax was used to lower production levels of cereals for export and raise funds to finance export subsidies (Rougoor et al., 2001). However, as of 1990 an environmentally motivated tax on phosphorous fertilizers was introduced, thus aiming explicitly at reducing the phospourous content of the fertilizer mixes. This tax was thus based on kg P used (Peltola, 2002). From 1992 and onwards the original fertilizer tax (based on N) and the new environmental tax (based on P) were combined, and the multiple goal of raising export subsidy funds and decreasing phosphorus use was retained.

Previous studies indicate the existence of a relatively price elastic nitrogen demand and with substantial spiollovers on the use of phosphourous. Thus, taxing nitrogen (as ingredient of a fertilizer mix) could have important impacts on both nitrogen and phosphorous use in Finland, but only at relatively high tax levels. The original fertilizer tax was however signficantloy lower than the corresponding taxes in, for instance, Austria and Denmark (however with fewer sectoral exemptions than in Denmark). The tax rate on phosphate was higher but this tax was only in place for five years and the impacts have not been studied in any detail.

When the taxes were abolished in 1995 the so-called Finnish Agri-environmental Program (FEAP) was introduced, and it relies essentially on a command-and-control approach to pollution control in the agricultural sector. In 2007, though, a study on the possible need for a new fertilizer tax was published.

3.2.4 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 phos-phate surpluses rather than inputs. The Dutch system was called MINAS (Mineral Accounting System) 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 at the farm level is set up and a tax is paid per kilogram that exceeds a certain levy-free surplus per hectare. Table 3.2 provides an overview of the levy-free surpluses for nitrogen and phosphorus as well as the tax rates levied on “excess emissions”.

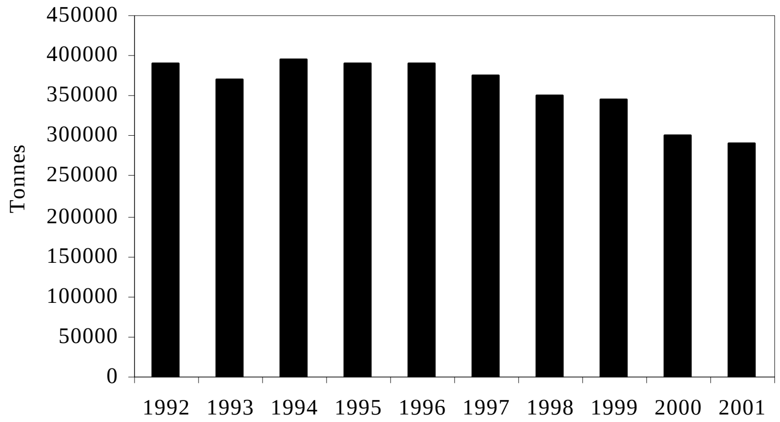
Table 3.2 Fertilizer taxes (levies) in the Netherlands

	Levy free surpluses		Tax rates	
Year	Nitrogen* (kg N ha^{-1})	Phosphate (kg P_2O_5 ha^{-1})	Nitrogen (EUR/kg)	Phosphate (EUR/kg)
1998	238	40	0.7	1.1
2000	188	35	0.7	2.3
2002	165	30	0.7	2.2
2003	140	20	2.3	9.1

*Average of loss standards applied to different types of grass and arable land.
Sources: Van Eerdt et al. (2004) and Sjöberg (2004).

The tax revenues are not earmarked, and are thus simply added to the state budget. All farm types (except cultivation under glass) are included in MINAS, but the impacts on different types of farms have often been quite different (see below). Figure 3.2 shows the consumption of fertilizers in the Netherlands over the time period 1992–2001. When the MINAS system came into effect a reduction of about 4 per cent took place during the first year. Since then the levy free surplus has been lowered while higher tax rates have been gradually implemented (see Table 3.2). In 2001 consumption had decreased by approximately 26 per cent compared to its 1996 level. According to a report by the European Fertilizers Manufacturers" Association, the relatively high tax level played an important role in inducing this reduction (ECOTEC, 2001). CBS (2003) and Westhoek et al. (2004) support this conclusion and report that the reduction in use also resulted in declines in the total nitrogen and phosphate surpluses on agricultural soils. It is evident that in those farms that did not have to join the system until the year 2000, nitrogen leakage reductions were considerably more modest than for the farms which joined the system already in 1998 (Westhoek et al., 2004).

Figure 3.2 Annual nitrogen (N) fertilizer consumption in the Netherlands, 1992–2001*



Sources: EEA (2003) and Sjöberg (2004).

Since the MINAS policy is applied broadly across virtually all significant farm types and targets surpluses (rather than inputs), it has a good potential for being relatively cost-effective compared to a tax on fertilizer consumption only. The empirical evidence on this issue is however mixed. The dairy farmers have managed to lower their nitrogen surpluses by reducing the use of chemical fertilizers at a relatively low cost, and these farms" contribution to total reductions are also significant (CBS, 2003). The economic effects on farms with intensive livestock production (pig and poultry farms) have been more profound, primarily since the average manure disposal costs per farm increased more than five-fold during the

1990s, much due to the MINAS system (Westhoek et al., 2004).[7] This implies that MINAS also induced reductions in manure output. Even though MINAS overall provided incentives for a fairly cost-effective reduction of nitrogen and phosphorous surpluses,[8] these gains are at least partly outbalanced by the relatively high administration costs (Van 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 EUR 55 million per year) and higher administrative expenses (an additional EUR 125 million per year). Public spending on implementing the new Fertilizers Act in the Netherlands has mounted from EUR 20 million in 1998 to over EUR 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 per cent in 1992–1995, to 25 per cent in 1997–1999 and 40 per cent in 2000–2002 (see Table 3.3). 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 groundwaters (Van Eerdt et al., 2005).

Table 3.3 Per centage of farms in sandy areas with nitr ate concentrations of 50 mg l^{-1} or less in upper groundwater

Sandy areas	1992–1995	1997–1999	2000–2002
Arable farms	10	30	30
Dairy farms	5	25	40
Other farms	-	10	20

Source: Van Eerdt et al. (2005).

[7] MINAS is intended to ensure the responsible removal of all manure that cannot be used in accordance with loss standards on the intensive livestock farm that produced it. This includes disposal to other farms, manure processing and exports. Since the average intensive livestock farm consists of 10 hectares of land, by far the majority of the manure must be removed (Van Eerdt et al., 2005).

[8] The cost-effectiveness of MINAS can however be questioned in the case of pig and poultry production. MINAS expresses surpluses in units per surface area, and does not therefore fully account for variations in nitrogen and phosphate stocks in animals, animal feed and animal manure (OECD, 2004). Some variations were, though, acknowledged in the sense that no fixed numbers for the excretion per animal were used.

The MINAS system was abolished in January 2006, as a direct consequence of the EC court decision in case C-322/00, which stated that the Dutch government had failed to implement certain elements of the EU Nitrate Directive. Westhoek et al. (2004) also note that the political climate in the Netherlands promoted efforts to make regulations simpler and to have less civil servants. The Netherlands now have 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.

3.2.4 Norway

In 1988 Norway introduced a tax on chemical fertilizers, and from the start the tax (or rather the charge) 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 per centage of the fertilizer price and initially it was only about one per cent 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 eight per cent 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 per cent 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 per cent of the nitrogen price, i.e., at the time about NOK 1.2 (EUR

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 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 EUR 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. International trade liberalizations 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).

3.2.5 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 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 (EUR 3.3) for every gram of cadmium that exceeds 5 grams 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 per cent of the fertilizer price (SOU 2003:9). 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).

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 per cent, 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 per cent had been met, but consumption still decreased for this type of fertilizer (Nordic Council of Ministers, 2002). 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 per cent of the sale price.

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. Drake and Hellstrand (1998) report that the cadmium tax reduced the application of of phosphorous fertilizers by roughly 1–2 per cent, and it has represented a reasonably cost-effective approach to restricting cadmium fluxes to soils.

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 per cent. 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)). This 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 competitive effects 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 2003:9) 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.

3.2.6 Summarizing comments

Pollution caused by fertilizers is dispersed, and this raises the complex policy issue of how to design efficient economic instruments to achieve reduced environmental damage. Most countries have in practice opted for input taxes, and overall the experiences show that they have played some role in reducing fertilizer use. Still, the price responses are low, and this means comparatively small impacts in terms of quantity reductions. The environmental effecttiveness of the taxes is very hard to assess, primarily since they are typically not proportional to damage done. In the case of fertilizers, damages vary mainly because of the receiving environment conditions. 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; it has however also resulted in high administration costs due to the need to regulate animal manure.

As was noted above, the fertilizer case is interesting also because it illustrates how the use of tax revenues can affect both the effectiveness and the legitimacy of a tax policy. The main impacts 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. Thus, the disadvantages of earmarking (see chapter 2) may be a relatively cheap price to pay for gaining some support, understanding and preferably partnership with the concerned industry. Moreover, the Austrian experience also illustrates that chemical taxes can provide signalling effects, i.e., just as much as they work via the price signal they also “work via their attention-seizing function,” (Speck et al., 2006, p. 221).

The above review also illustrates that the future implementation of economic instruments on chemical compounds may run the risk of being in noncompliance with legal obligations, not the least since most legal rules give priority to goal fulfilment rather than to flexibility in compliance strategies. The EC Nitrate Directive and the abandonment of the Dutch MINAS system is an apt illustration of this. In essence, the Nitrate Directive largely prioritizes risk reduction, goal fulfilment and full compliance with the environmental standards. Economic instruments, however, 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 (although *ex post* revisions are possible). 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.

3.3 The Nordic experience of pesticide taxation

Pesticides are a very heterogeneous group of products, as are the range of damages associated with the application of these products across different environmental media. Potential consequences of pesticide use in agriculture include, for instance, lack of aquatic and terrestrial biodiversity, contamination of groundwater, poisoning of agricultural workers and unwanted residues in food and water (Mourato et al., 2000). Exposure to pesticides through residues on food may leave people more susceptible to diseases from viruses and bacteria

Several studies on the effectiveness of pesticide taxes have been conducted, and a (largely Nordic) selection of these are summarized in Table 3.4. A large bunk of this literature is devoted to comparing and evaluating the efficiency of different tax designs as well as alternative regulatory approaches to reduce damage resulting from agricultural chemical use in different countries. Most *ex ante* analyses of the regulation of pesticides conclude that taxes can be an effective way of inducing reductions in use. However, the design of the tax (e.g., ad valorem or volume-based) may play a role in determining the effectiveness of the tax.

Since the early 1980s, the total quantity of pesticides in the Nordic countries has been reduced by more than 50 per cent. In some years pesticide sales have increased substantially (e.g., 1995 in Denmark), but these are often years preceding the introduction of a new (or increased) tax. These peaks are therefore probably explained by pre-tax hoarding behaviour (see also below). In order to evaluate to what extent the overall reduction can be attributed to the implementation of pesticide taxes we must turn to the experiences of the different tax schemes in the Nordic countries.

Before proceeding, though, it is important to note that the assessment of pesticide reduction policies involves a number of difficult methodological issues. These include, for instance, the facts that: (a) there exists many different types of pesticides (i.e., sub products); (b) the need for pesticides varies substantially depending on season and geographical region; and (c) new stronger pesticides are being introduced on the market (Skou Andersen et al., 2001). This last point is very important, since it indicates that volume-based reductions in pesticide use may provide an exaggerated picture of the environmental benefits of pesticide reductions. For this reason the Danes prefer to use the so-called ”frequency of treatment” as a parameter for analyzing the environmental impacts on pesticide use. The treatment frequency index expresses the average number of times an agricultural plot can be treated with the recommended dose, based on the quantities sold. According to Danish statistics, while the quantities of pesticides used in Denmark have been halved in 2003 in relation to the year 1990, the treatment frequency calculation shows a reduction of “only” about 35 per cent (Nielsen, 2005).

Table 3.4 Selected simulation studies of pesticide taxation schemes

Study and geographical area	Purpose and methodological approach	Main findings and implications
Shumway & Chessser (1994) (Texas, USA)	Analyzes the impacts of an ad valorem tax on pesticides on cropping patterns, pesticide demand and water quality. An econometric approach is used to study the impacts of a hypothetical ad valorem tax of 25 per cent on pesticide use.	There is a negative relationship between pesticide price and output quantity of almost all crops examined (rice, corn, grain sorghum, hay etc.), and the demand for pesticides tends to be price elastic.*
Falconer & Hodge (2001) (UK)	Evaluates four different input taxes in the context of reducing environmental damage from the use of pesticides (i.e., an ad valorem tax, a fixed levy per spray unit, a per kilo active ingredient levy and a levy based on the pesticide hazard score of each individual product).	The taxes were found to vary according to the goal to be achieved. Since pesticides is a heterogeneous group of chemicals with different environmental effects, single instruments are unlikely to solve the multi-dimensional environmental problems and thus differentially-applied taxes or restrictions should be considered.
Sunding & Zivin (2000) (California, USA)	Focuses on pesticide taxation and pre-harvest interval regulation** in the context of health risks among workers due to toxic substances in pesticides. The tax is designed to change marginal incentives to use toxic materials.	A marginal increase in the pesticide tax will reduce contamination and reduce the number of poisonings among workers, while the pre-harvest interval regulation will have an ambiguous effect on the number of poisonings.
Dubgaard (1987) (Denmark)	Analyzes the impacts of implementing pesticide taxes of DKK 100 and DKK 200 per standard dose*** in Denmark. He uses two different models, a damage threshold model and an econometric model.	The models produce very different results although they both indicate a reduction. An important reason for this ambiguous result about the size of the tax impact is that different own-price elasticities of pesticide demand are assumed in the models.
Rude (1992) and Jensen & Stryg (1996) (Denmark)	Rude provides a re-assessment of the Dubgaard study, while Jensen and Stryg (1996) extend the work by including the economic effects outside the agricultural sector,	Rude concludes that the taxes would have significant long-run impacts on pesticide doses. However, a tax levied on the wholesale price of pesticides would be difficult to administer. Jensen and Stryg are more pessimistic about the impact of the taxes.
Gren (1994, 1996) Sweden	Gren (1994) employs time-series data on pesticide use in Sweden to investigate the economic effects of pesticide taxes. Gren (1996) analyzes the impacts of a 50 per cent tax on fertilizer use and on tax revenues.	A pesticide tax would be a cost-effective policy, but its impact on farmers’ incomes would be higher than if a command-and-control policy was to be introduced instead. A 50 per cent tax on fertilizers would reduce the use of pesticides by about 45 per cent.

* It is worth noting, however, that this result contradicts the findings of many other studies, which indicate a more price inelastic pesticide demand. See, for instance, McIntosh and Williams (1992).

** Pre-harvest interval regulations impose a time interval between pesticide application and harvesting.

*** This unit – also referred to as “treatment frequency” – takes into account that new low-dosage agents, which make pesticides more effective and thus lead to similar impacts but in smaller quantities (compared to older types). The tax levels used here correspond roughly to an *ad valorem* tax of 60 per cent and 120 per cent, respectively, of the market price of pesticides in the period 1987–1989.

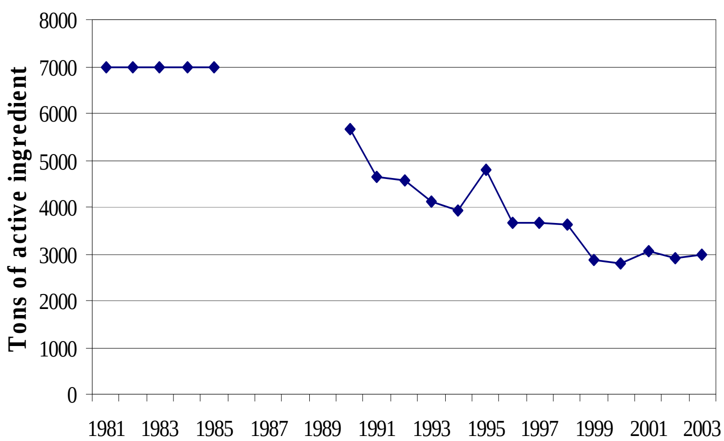
3.3.1 Denmark

Before 1996 a general value added tax of 3 per cent of the retail price (introduced in 1986) was levied on all pesticides used in Denmark. This tax constituted only a limited supplement to other regulations of pesticide use (e.g., information, approval, research etc.). According to Bager and Sogaard (1994) the cost of pesticides for Danish farmers in 1993 constituted only about 6 per cent of total operating costs. This implies that the 3 per cent tax is likely to have had only limited impacts on pesticide reduction.[9] In 1996, however, a more ambitious *ad valorem* tax on pesticides was introduced, and in 1998 the tax rates were increased.

The purpose of the new tax is to influence the use of pesticides by farmers and reduce total consumption. The tax is based on the maximum retail price and it is differentiated across different pesticides (e.g., insecticides are taxed more heavily than fungicides) (Nordic Council of Ministers, 2002); it is imposed on domestic manufacturers and on importers when the product is sold for use in agriculture. Exports are exempt from the Danish pesticide tax (Ibid.). The majority of the price increase has been passed on to farmers; however, a part of the price increase has been absorbed by the manufacturers. The price elasticity of pesticide demand has been estimated at -0.5 (ECOTEC, 2001).

At the time of the introduction of the new tax, it was estimated that the tax would induce a 5–10 per cent reduction in pesticide use. Figure 3.3 displays pesticide sales in Denmark. The decrease has been the largest for herbicides. However, since the use of pesticides is affected by many factors and the tax was introduced at a time when the level of consumption already was falling, it is difficult to determine the isolated tax impact.

Figure 3.3 Pesticide sales in Denmark, 1981–2003*



** The 1981–1985 data indicate the average sales over these five years.*

Sources: EEA (2003), Sjöberg (2004) and Nielsen (2005).

[9] In addition to the 3 per cent value added tax, a 20 per cent tax was levied on pesticides sold in quantities of less than 1 kg or 1 liter for household use (Skou Andersen et al., 2001).

The sales of pesticides have fallen consistently since the early 1980s, perhaps most significantly during the period when the low tax rate was in place. One exception to this is the rise in sales in 1995, which can be ascribed hoarding behaviour due to the announced tax change in 1996. Overall, though, this temporary peak tells us little about the tax's impact on consumption. In terms of the treatment frequency index we witness a reduction from 2.7 to just above 2 in 2003. This implies that the sale of pesticides in Denmark corresponds to spraying all the conventional farmland with two treatments with the recommended dose rate of pesticides sold in 2003. Still, the *ex post* evaluation is also complicated by the fact that any positive environmental effects on groundwater quality will not be observed until many years after the measures have been implemented.

In a report from the Danish Institute of Agricultural and Fisheries Economics (1998) the Danish pesticide tax is criticized for only providing incentives for substitution from expensive to cheap pesticides, while the environmental impact of such substitution is ambiguous. For instance, new and more expensive but less environmentally damaging pesticides will incur higher tax rates than older cheaper and more hazardous products (Nielsen, 2005). A fixed tax on the recommended normal dose has been proposed instead, but this approach is complicated by the fact that these normal doses differ across different crops.

The above also means that the *ad valorem* tax does not induce technological development that favours the use of less environmentally damaging pesticides. 60 per cent of the tax revenues in 1998 were however channelled back into the agricultural sector through different subsidy schemes, such as those to organic farming and extension services. The remaining 40 per cent was used for public research and pesticide monitoring programmes (Nielsen, 2005).

No *ex post* evaluation that focuses on the competitive impacts of the Danish pesticide tax has been identified. The tax is imposed on both domestic manufacturing and imports. This reduces the incentive to import cheaper products but also the costs of control and administration, because the number of companies registered at these levels is considerable smaller than at the retail level. Studies on the economic impacts of the tax have concluded that the use of pesticides can be reduced further without imposing large costs on farmers; it is suggested that the consumption of pesticides could be reduced by 30 to 50 per cent from current levels without any significant impact on competitiveness or the economy as a whole (ECOTEC, 2001).

3.3.2 Finland

In Finland the use of pesticides has fallen significantly during the 1990s, but much of the reduction in the quantities used can be ascribed to the introduction of new low-dose agents (OECD, 1996). For this reason the observed reduction in volume does not reflect a corresponding reduction in reduced environmental damages. In 1988, the Finnish government introduced a fee on pesticide use. The original aim of this fee was not primarily to induce reductions in the use of pesticides but rather to finance the control and registration costs associated with the use of pesticides (i.e., the fee is more of a cost-covering charge than an incentive tax).

The fee is designed as an added value fee levied on pesticide dealers (manufacturers and importers); in 1998 it was set at 3 per cent of the previous year's turnover and the current (2006) rate is 2.5 per cent (Skou Andersen et al., 2001; Speck et al., 2006). No *ex post* evaluations of the fee's impacts have been traced, but the impacts on pesticide use are likely to have been modest given the design of the fee and its relatively low level. The tax could have imposed some incentives for users as the increase in costs is passed on to the consumers, but it is also fair to assume that the signalling effects of the tax are likely to have been limited given its low rate and blurred motives. The fee was abandoned a few years ago.

3.3.3 Norway

Pesticide use in Norway has since 1988 been subject to a tax. Still, just as in the Finnish case, the original tax was not introduced primarily because of its impact on pesticide use, but rather to finance selected environmental projects (Berg, 1995). The tax was designed as a value added tax levied on wholesalers of pesticides. The rate was gradually increased during the late 1980s and 1990s. In 1998 it amounted to 15.5 per cent of the wholesale price.

During the 1990s a number of *ex post* evaluations of the Norwegian pesticide tax were conducted, but the results from these are somewhat mixed. In NOU (1992:203) it is claimed that the pesticide tax only had limited effects on pesticide use, primarily since it did not differentiate between the different pesticides' health and environmental impacts. In 1995 the Norwegian green tax commission evaluated the role of taxes in environmental policy; they concluded that the Norwegian pesticide tax may have reduced pesticide use by as much as 30 per cent (NOU 1996:210). However, a large part of this reduction is due to the transition towards more concentrated pesticides. The Ministry of Agriculture (1998a, 1998b) supports the conclusion drawn in NOU (1992:3), and argues that the impact of the pesticide tax has been modest. Still, Skou Andersen et al. (2001) note:

> The limited nature of the effect of pesticide taxes on actual consumption is contradicted, however, by the fact that a warning of a small increase in the environmental tax from 13 % in 1995 to 15.5 % in 1996 is thought to have encouraged hoarding. From 1994 to 1995, pesticide sales rose by 8 %. (p. 83)

This observation (and similar observations in the Danish case above), though, tells us very little about the farmers' costs of – and thus motivation for – reducing pesticide use. What it shows is essentially that the extra cost for Norwegian farmers of the tax increase exceeds the cost of storing pesticide products for one year; to what extent the tax raise is high enough to induce significant changes in pesticide *use* is another question. In any case, the Ministry of Agriculture (1998a, 1998b) came to the conclusion that a tax increase was needed. It also recommended that the new tax would better reflect the health and environmental impacts of pesticides although such a tax is typically more difficult to administer.

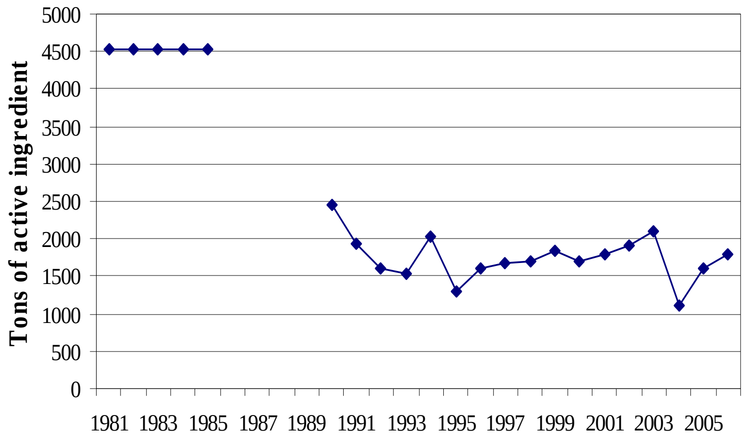
In 1999, the Norwegian tax was amended in line with these recommendations, and the tax was also changed from an *ad valorem* tax to a tax per normal dose. According to Pearce and Koundouri (2003), the current Norwegian tax on pesticides is one of very few taxes on agricultural pollution that (at least to some extent) is differentiated by toxicity. The present tax consists of a control (or inspection) tax and an environmental tax (Speck et al., 2006), where the former generates revenues covering the costs of the Norwegian Agricultural Inspection Services.

3.3.4 Sweden

In Sweden charges on pesticides were introduced in 1984 with the aim of reducing environmental risk and health risks associated with the use of pesticides. The charge is set on the basis of the weight of the active ingredient and is imposed on pesticide manufacturers and pesticide importers. Apart from the environmental charge, a price regulation tax was also introduced on pesticides in 1986. This tax has been used to finance export of agricultural products, but it was abolished in 1992. Instead, in 1994, the environmental charge was raised from SEK 8 per kg active ingredient to SEK 20 per kg. In 1995 the charge became a tax, as there is no longer any direct link between the level of the revenues raised and reinvestment of the revenues in environmental projects.

Figure 3.4 shows the development of pesticide sales over the time period 1981–2006. The total amount of pesticides used in Sweden has fallen since 1986 and in 1995 it had been reduced by 35 per cent compared to the average amount used between 1981 and 1985. Still, the greater part of the reduction was due to other factors than the tax; the pesticide tax has only had limited effects on the use due to the low tax rate and the lack of substitutes. SOU 1990:59 confirms this conclusion, and argues that the reductions in insecticide and fungicide uses have been less than 2 per cent as a result of the tax. It is even the case that after the tax raise in 1994, consumption of pesticides increased slightly in Sweden. An important explanation is that as Sweden joined the European Union in 1996, the incentives to intensify production levels in the agricultural sector were strengthened.

Figure 3.3: Pesticide sales in Sweden, 1981–2006*



* The 1981–1985 data indicate the average sales over these five years.

Sources: EEA (2003), Sjöberg (2004) and Swedish Chemicals Agency (2008b).

The Swedish Board of Agriculture (1992) concludes that in 1991 the environmental and price regulation taxes on pesticides together made up about 5–20 per cent of the price of pesticides. By comparing the actual use of pesticides in 1991 with the volume of pesticides that would have been used if there had been no tax, the Board concludes that the direct effect of the tax has been a 4–10 per cent reduction in pesticide use. The Board also argues that the same reductions could have been attained at lower costs through other policy means; advice, information and R&D efforts (which however largely are financed by the tax revenues). An evaluation made by the Swedish Environmental Protection Agency in 1997 reported that the charge/tax was set too low to have a significant impact on long-term user behaviour (SEPA, 1997b). In 2001 a government evaluation of the Swedish policy on pesticides was initiated; it concluded that the tax should be raised to increase its impact on pesticide use (SOU 2003:9), and this tax raise also formed part of the 2004 government budget proposition.

3.3.5 Summarizing comments

Most *ex ante* analyses of the regulation of pesticides conclude that taxes can be an effective way of inducing reductions in use. However, the design of the tax (e.g., *ad valorem* or volume-based) plays a role in determining the overall effectiveness of the tax policy. In practice the volume of pesticides used has decreased in the Nordic countries which have implemented taxes on pesticide use, but the *ex post* evaluations that have been made indicate that the taxes' impact on this development has been modest.

The most significant effect can – according to ECOTEC (2001) – be found in Denmark. A large part of the observed reductions are, though, due to a transition to low-dose agents that have the same impact even when smaller quantities are used. This implies therefore that the observed volume reductions do not reflect any corresponding reductions in health and environmental impacts. The Danish case also illustrates the inability of *ad valorem* taxes to approximate damage done and induce incentives to reduce environmental impacts rather than just use. The Norwegian taxes on pesticides show that there exist ways in which the tax system can be designed to achieve a closer proportionality to damage done. An optimal differentiation of taxes is however likely never to be achieved, especially since the damages caused by chemical use and disposal tend to vary a lot because of the receiving environment conditions.

3.4 The Nordic approaches to the taxation of hazardous chemicals

Environmentally motivated economic instruments for hazardous chemicals are relatively rare, and in this section we focus primarily on the regulation of organic solvents use in the Nordic countries. However, before proceeding we comment also briefly on the use of economic instruments for other so-called hazardous chemicals, including ozone-depleting chemicals and solvents used in the paint, printing and pharmaceutics industries.

Australia, Denmark and the USA tax ozone-depleting chemicals (ODC). There exists little evidence on the effectiveness of the Australian ODC tax, and Blackman and Harrington (1999) briefly remark that the Danish tax seems to have had significant impacts on use. In the USA the government decided in 1987 to use a tradable allowance system, in the sense that producers of ODCs were required to have adequate allowances. However, this system was only in use for a couple of years, and in 1989 a change to an excise tax on ozone-depleting chemicals took place. The U.S. ODC tax is differentiated, i.e., the specific rates for each chemical are set proportional to chemical-specific ozone-depleting factors. According to Hoerner (1995), the tax has been a policy success; total production of listed chemicals has declined substantially and the production of ozone-depleting chemicals with the highest potential depleting rate has been reduced by about 50 per cent since the tax was implemented. However, since the tax and a tradable permit system worked in parallel during the early 1990s it is difficult to determine which mechanism should be credited with the reduction in use. Stavins (2000) reports that different analysts tend to have different views on this particular issue. It is clear, however, that the pre-announced (and thus predictable) increases in the tax rate have played an important role in attaining the positive effects experienced, since they have facilitated the planning processes of producers and consumers of ODS.

The Danes have also introduced taxes on a product's PVC or phthalate content. The reason was a failed agreement with industry that did not meet the environmental targets. The rates introduced in 2000 have remained constant in nominal terms since then, and they equal DKK 2 per kg PVC and DKK 7 per kg phthalates. On this basis, rates are then calculated for a range of different products (Speck et al., 2006).

Chlorinated hydrocarbons are good solvents for degreasing and dry cleaning, since they dissolve fats, but many of these substances are hazardous or toxic. Moreover, they can form extremely toxic dioxins, such as chlorofluorocarbons (CFCs), if burned. The *ex ante* studies that are reviewed in Table 3.5 focus on the use of solvents in the paint, printing and pharmaceutics industries and the resulting emissions of VOC. The overview of the empirical experience of solvents taxes following in subsections 3.4.1–3.4.3 focuses on countries in which chlorinated solvents, used for metal degreasing and dry cleaning, are targeted with taxes.

Table 3.5: Selected simulation studies of solvents taxation schemes

Study and geographical area	Purpose and methodological approach	Main findings and implications
Santos et al. (1999) (Portugal)	A hypothetical (weight-based) tax on solvents on the production costs in three industries (printing and ink, pharmaceu-tical and paint industries) is analyzed.	The effects on the cost structure would be very significant for the ink and paint industries, but more modest for the other two industries.
Oosterhuis et al. (1997, 1998) (Europe)	Analyzes the feasibility of harmonized use of taxes on solvents to reduce VOC-emissions in the EU and in the Central and Eastern European Countries. The economic impacts on the paint, printing and pharmaceutics industries are assessed.	The economic impacts on the three industries would not be significant. A uniform tax throughout Europe would not be optimal or feasible and a differentiated tax therefore would be preferable.
Olsthoorn et al. (1996) (Europe)	Examines the feasibility of an EU-wide tax on emissions from organic solvents, and compare this with other regulatory policy instruments. Two different designs of the EU-wide tax are considered; a partial system in which only paint containing organic solvents is taxed, and a comprehensive system in which a tax on VOCs is combined with a refund system.	The latter system was found to be more attractive, since it was judged to be more effective in achieving environmental goals and lead to limited administration costs. An EU-wide tax would be more efficient and effective from an environmental perspective than other market-based instruments, such as subsidies and tradable permit markets, or other administrative instruments, such as eco-labelling and voluntary agreements.

The above studies show that taxation of solvents can be an efficient way of reducing the environmental impacts from these even though taxes are currently not common in this field. A limited number of countries have however implemented taxes on chlorinated solvents. Their good fat-dissolving properties mean that they are used as degreasing agents, for instance for metals, and as dry-cleaning fluids. Here we focus on the Nordic experiences of chlorinated solvents taxation in Norway and Denmark. We also discuss an interesting study on the Swedish policy on trichloroethylene (Slunge and Sterner, 2001), in which the authors argue – on empirical grounds – that the current Swedish prohibition has been ineffective (at least cost-wise) in achieving a phase-out of this chlorinated solvent and that a tax policy would have been more effective.

3.4.1 Denmark

Since 1996 Denmark employs taxes on the solvents tetrachlorethylene, dichloromethane and trichloroethylene. The purpose of these taxes has, among other things, been to reduce the use of these substances in the light of the ban on ozone-depleting CFCs. The tax rate is set at EUR 0.27 (DKK 2) per kilo (Speck et al., 2006). This corresponds to a consumer price increase of about 25 per cent. The tax on substances and products sold for exports is refunded due to concerns over competitiveness (RPA, 2002).

According to the Nordic Council of Ministers (2002), the consumption of these three solvents has dropped significantly since the taxes were introduced. This has been particularly evident for dichloromethane, and although the uses of trichloroethylene and tetrachlor-ethylene have also decreased profoundly (by 76 and 25 per cent, respectively) considerable use still occurs (RPA, 2002).

3.4.2 Norway

Since 2000, Norway taxes trichloroethylene and tetrachlorethylene. The purpose of the Norwegian solvents taxes is to promote the phasing-out of these chemicals and to reduce the environmental impacts. The tax rates (as of 2005) are about EUR 6.5 per kilo (just above 50 NOK) for both substances, which means that the market prices were raised about five times after the introduction of the tax. The tax on trichloroethylene is partly earmarked in the sense that half of the revenues are refunded on delivery of trichloroethylene-containing sludge for proper treatment (e.g., treatment plants or authorized recyclers). This deposit-refund feature of the tax scheme has, according to Sterner (2003), increased the legitimacy of the tax among companies. The Norwegian industry preferred the tax over a ban since the former permits a greater amount of flexibility in the compliance strategy.

There is evidence of a significant tax effect on solvents use. For instance, Figure 3.4 shows an index of trichloroethylene use in Norway over the time period 1986–2000. Based on these data, Sterner (2003) concludes that: "[…]" data for 2000 indicate a drastic decline in TCE [trichloroethylene] use of more than 80%. Even if there was some increase in 1999 (attributed to pre-tax hoarding), the fee still appears to have been effective." (p. 298). Eriksen (2001) (cited in Slunge and Sterner, 2001) confirms that the Norwegian taxes on trichloroethylene and tetrachlorethylene both have been very effective and have led to substantial decreases in purchases between 1999 and 2000 (82 and 90 per cent reductions, respectively).

3.4.3 Sweden

In 1991, the Swedish Parliament decided to prohibit the use of trichlorethylene (TCE), whose main use is metal degreasing. The new law was to become effective in January 1996. An important reason for the ban on TCE was the plan to abolish the use of ODCs (following the Montreal Protocol in 1987). Since ODCs also are used for solvents in metal degreasing, many users reverted to, for instance, TCE. However, given that TCE is highly toxic and cancerogenic, the Swedish government wanted to prevent (entirely) this type of substitution.

What makes the Swedish ban on TCE interesting for this report is the fact that it has been criticized for its effectiveness, and, some argue, a tax on TCE would have led to a swifter and more effective phase-out of the substance (Slunge and Sterner, 2001). There are two basic reasons for this conclusion; the first relates to the perceived legitimacy of the policy and the other to the structure of the TCE replacement costs, which largely determine the extent to which a tax can induce a phase-out.

First it is important to note that during the time between announcement (1991) and implementation of the ban (1996), some industrial users of TCE did little to prepare for a phase-out. This was partly due to the fact that the policy was so strong that many companies did not think the government really meant business. As they realized that the ban proposal was serious, TCE use began to decrease but at a relatively slow rate, and many companies spent a lot of effort to lobby against the ban and the protests grew stronger and stronger. Over the period 1993–1995, the use of TCE declined by about 20 per cent. Slunge and Sterner (2001) note:

> Many petitions and articles were written and a number of companies decided to fight the legislation, threatened to close down or leave the country, and have appealed to the courts. Leading points of contention are that the industries disapprove of the prohibition as a method, its timing, and a number of its consequences, [...]. (p. 9).

As a result of the opposition the Chemicals Inspectorate made possible the issuing of waivers for companies that could report difficulties in phasing out TCE; however, these companies also had to pay an exemption fee (SEK 150 per kg). This fee was challenged by the European Commission since it was considered to be “out of proportion” with the environmental damage. In 2000, the EC court ruled, however, that the TCE ban as such does not run counter to EU legislation on the free movement of goods. An important reason for this ruling was the reasonable possibilities of getting waivers, the same waivers that also made the ban less effective in phasing out TCE than the corresponding policies pursued in other European countries (including the Norwegian tax reviewed above).

Slunge and Sterner (2001) then make a rather strong case for the notion that a Swedish tax on TCE would have been a more cost-effective policy for phasing out TCE, and it would also have led to much less opposition among users (and hence to lower transaction costs). Based on a variety of sources, they present firm-specific estimates of the marginal costs of replacing TCE. These costs are of course uncertain, which would imply that a quantitative regulation (but not necessarily a ban) could be the best policy. Still, the authors argue that their cost estimates are accurate enough to conclude that a very large share of replaced quantity of TCE carries low marginal substitution costs while for a few firms the costs are very high. This implies that a tax of SEK 50 per kg (roughly equivalent the Norwegian one) would have been sufficient to phase out around 90 per cent of TCE use. A higher tax would have had only a very marginal effect on TCE use. Thus, the impacts of the tax on TCE use appear quite predictable ex ante, making the tax alternative (or a deposit refund system) an attractive policy in this case. With the benefit of hindsight, the authors argue, such a policy would probably have provoked less resistance and achieved a swifter – and less costly – reduction in TCE use than has the ban.

3.5 Conclusions and lessons for the future use of environmental taxes and fees

The taxation of chemical compounds in many European countries illustrates the importance of – and the practical problems involved in – designing the tax so as to target the environmental damages as closely as possible. The taxation of chemical use is often motivated by the desire to target downstream external costs in the form of harmful exposure to nature. These non-point source emissions are difficult and costly to control in those cases where environmental damages vary by location. Therefore most of the taxes implemented in practice are second-best instruments as they focus on the consumption or sales of the respective chemicals. Still, the Dutch tax on fertilizers and the Norwegian taxes on pesticides show that there exist ways in which the tax system can be designed to achieve a closer proportionality to damage done.

The analysis has also illustrated that the more successful policy makers are in designing an economic incentive scheme that can achieve a closer proportionality to damage done, the less opposition to the policy is (ceteris paribus) likely to emerge (at least from those directly affected by the policy). For instance, environmental taxation that targets damages rather than consumption of upstream products tends 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 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 R&D 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.

The problem of addressing environmental damages close at source in the use of economic incentives is well-illustrated by the use of *ad valorem* taxes, i.e., taxes designed as a fixed per centage of chemical prices. We have already noted that taxes on physical units may well fail to approximate differential environmental and health impacts, but this problem is typically even more accentuated for per centage-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) observe:

> 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 taxes 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. Thus, as far as possible, these types of policy designs should be avoided.

The experiences from the European countries covered in this study indicate that some kind of earmarking of tax revenues (or revenues from auctioning tradable allowances) can be effective in increasing the perceived legitimacy of the policy, although (as illustrated by the Norwegian case) it may not represent a sufficient condition for political acceptance. To some extent the main impacts of some of the 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 most types of chemical taxation are minor and (occasionally) of a transient nature. Thus, the disadvantage of ear-marking 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 a function of past releases. Water contamination following fertilizer use is a good example of this, but this extends towards many other chemicals used in, for instance, solvents and pesticides. If revenues are earmarked, they can also be used for clean-up programs and in this way reduce the impact of stock externalities. Still, earmarking can never be an excuse for the implementation of otherwise inefficient economic instruments.

Finally, economic instruments have both cost and signalling (aware-ness-raising) effects, and it is useful to combine any economic incentive chosen with targeted information. This means, thus, that economic and informative policy should be considered complements rather than substitutes in chemicals policy. The combination of economic instruments and information may be particularly important in the control of down-stream externalities, e.g., household use of hazardous chemicals where people's ability and willingness to search for and process information may be limited.

4. The scope for economic instruments in controlling chemicals use: three case studies

4.1 Introduction

In this chapter we discuss and evaluate the scope for using economic instruments to manage the risks associated with the production and use of specific chemical compounds and products. We identify three cases for which economic instruments possibly could successfully complement other regulations restricting use and reducing any negative impacts on both health and the natural environment. These cases include the use of: (a) substance nonylphenol (NP) and its ethoxylates (NPEs), which break down into NP (section 4.2); (b) different types of two-stroke oils causing, for instance, marine pollution (section 4.3); and (c) 1,2-ethanediol (ethylene glycol) which primarily is used in the cooling water system of automobiles to prevent it from freezing (section 4.4).

For each of these cases we first discuss and analyze the nature of the substance and its environmental impacts. We then analyze the current production and use of the substance and any related products, including the presence of potential substitutes. The focus in this chapter is on the Swedish situation; the Swedish Chemicals Agency's so-called Product Register comprises extensive information on the flow of chemical substances (including imports and exports) during the last 10–15 years. The data on the other Nordic countries – Denmark, Finland, and Norway – are less comprehensive, but occasionally we are able to present brief comparisons of the situation in the different countries using the SPIN database from the Nordic Council of Ministers. Finally, the scope for different types of economic instruments is assessed, including their effectiveness, economic efficiency and potential for policy acceptance.

A number of factors have influenced the choice of chemical compounds and products to be considered in this chapter. These include:

- Many studies on the potential use of economic instruments in chemicals policy have targeted relatively broad product groups and chemical categories, such as solvents (Entec UK Ltd., 2000) and pesticides (Hoevenagel et al., 1999) on an EU-wide basis. However, in the following we have chosen a more narrow scope and target (at least in two of the cases) specific substances. We believe our approach makes it easier to identify and design efficient environmental policies, not the least by combining economic and informative instruments, but this also requires a detailed analysis of each case to avoid substitution to other hazardous substances. Broad tax bases run the risk of giving rise to fiscal rather than environmental impacts, and may therefore also lack policy legitimacy.

- The cases analyzed complement each other in the sense that they highlight the important differences in controlling point- and non-point source pollutants. They concern, thus, both industrial and consumer use, and in the case of NPEs both uses are relevant from an environmental and health point-of-view (European Commission, 2002). This implies that different types of economic instruments (e.g., emissions versus product charges) should be considered.
- The chosen cases are of policy relevance. Nonyphenol is a persistent, bioaccumulative and toxic substance, and economic instruments can complement existing regulations. For instance, it could reduce the need for future regulatory action under the Water Framework Directive. The uses of two-stroke oils and 1,2-ethanediol have been highlighted in policy work (e.g., SEPA, 2004; Swedish Chemicals Agency, 2004) and media (Edman, 2008; Lindström, 1998), but so far many of the risks associated with these uses are mainly managed through informative policy instruments.
- Finally, in all of the cases substitutes exist, making the case for economic instruments of practical interest. However, a major challenge concerns the import of hazardous substances (e.g., NPEs) in different products.

In sum, the different cases do address different challenges and opportunities in the implementation of economic instruments, but they should not be considered the best options available for introducing incentive-based policies.

In each of the cases studied we have divided the analysis into four sub-sections. *First*, we begin by discussing the nature of the substance or product, and the types of environmental and health impacts it causes. We also briefly review the existing regulations that are in place to reduce the use of the substance or product, including any EU legislation. *Second*, we proceed by considering the production, use and trade of the substance/-product in Sweden and (in part) in the other Nordic countries. The availability of substitutes is also discussed. *Third*, the possibilities to implement different types of economic instruments are analyzed, and in the end one of these are selected for further assessment. *Fourth* and finally, the potential impacts of the selected policy instruments are assessed, including the likely impacts on use and compliance costs.

4.2 The case of Nonylphenolethoxylates (NPEs)

4.2.1 The nature of the substance, environmental impacts and existing regulations

At room temperature, nonylphenol (NP) is a pale yellow liquid. The greater part of all nonylphenol manufactured is used as an intermediate in the production of surfactants, above all nonylphenolethoxylate (NPE). From a global perspetive the main use of NPEs is in products for industrial and institutional cleaning. Some NPEs are used as detergents, i.e., they dissolve a small amount of dirt/grease in a great deal of water, while others are used as emulsifiers, i.e., they help to form stable systems of more fat in less water. The water-solubility of NPEs diminishes with rising temperature, and this can be utilised, for example, as a means of emulsifying grease from textiles. NPEs are also used as dispersing agents, e.g., in the production of latex dispersions and as a pulp dispersant in the paper industry.

NPEs are easily broken down in the environment and generate NP (or short-chained NPEs), which is a persistent, bioaccumulative and toxic substance that gives rise to hazardous emissions to the aquatic environment (e.g., Esfahani, 2008) and the work environment.[10] Like many other chemicals, NPEs have a rather complex life-cycle involving primary manufacturing, professional use and consumer use, and the emissions arise from both point sources and more diffuse sources. Upstream in the production chain local concentrations in surface waters of NP and NPEs may be high where waters are receiving inputs from industries that use either NP or NPEs (at least in countries where the existing regulations are lax). Furthermore, NP is strongly adsorbed to sludge in the wastewater treatment process, which may then be applied to agricultural land. High concentrations of NP may also occur in soils where sewage sludge is applied (European Commission, 2002), resulting thus in another exposure route, through food to humans. Downstream the use of cleaning and degreasing may cause harm to the aquatic environment.

While the use of NPEs remains a serious environmental problem at many industrial plants in countries worldwide, the European countries have come a long way in substantially resticting industrial use. NP/NPEs were included in the 1992 OSPAR Action Plan, and were therefore also added to the List of Chemicals for Priority Action in 1998 (OSPAR Commission, 2001), and in Europe a voluntary ban on the use of NPEs in domestic detergents has been agreed by all the major manufacturers of detergents. The uses of NP and their ethoxylates in Europe are today regulated through the so-called Marketing and Use Directive (amended

[10] There is also an ongoing debate as to whether NPs are so-called endocrine disruptors (RPA, 2002a). Endocrine disruptors are chemicals that interfere with the body’s endocrine system and produce adverse developmental, reproductive, neurological, and immune effects in both humans and wildlife.

2003/53/EC) and the Water Framework Directive (2000/86/EC) under which NPEs have been listed as priority hazardous substances. The former Directive restricts the marketing and use in Europe of products and product formulations that contain more than 0.1 per cent (by weight) of NPE or NP. These uses include, for instance, industrial and institutional cleaning textile and leather processing, emulsifier in agricultural teat dips, metal-working, manufacturing of pulp and paper, cosmetic products etc. An important exception is made for the use of NPEs in closed processes, where all the NP are eliminated by industrial cleaning processes before the process water is released into a recipient. The Marketing and Use Directive is expected to remove 80 per cent of the environmental exposure and 65 per cent of the use of NPE.

In many countries, including the Nordic ones, thus NPEs are being more or less voluntarily phased out, but use however continues across a range of different sectors. One reason is that the import of NPEs in certain products is difficult to monitor. The imports of NPEs via textiles into the European Union provide the single most important explanation why we still find NPEs in the sludge of wastewater treatment plants (e.g., Swedish Society for Nature Conservation, 2008). Much of the production of textiles is outsourced to low-cost countries in, for instance, Asia where the environmental and health legislation is weak. This implies that the import of textiles may contain dangerous substances whose use is restricted in the EU: Clothes are frequently washed, and chemicals in the fabric leach out and find their way into the household's wastewater (Swedish Society for Nature Conservation, 2008).[11]

Furthermore, in Sweden the use of NPEs in cleaning agents was reduced by 70–80 per cent already during the time period 1990–1995, but NPEs have been discovered in a number of investigations of water quality in lakes and water-courses in Sweden and are still found in sludge from sewage treatment plants in relatively high concentrations (e.g., Andersson, 2007). One reason is that a number of imported cleaning agents comprise NPEs in concentrations higher than the detection limit (0.01 per cent by weight). For instance, the content of NPEs in one American cleaning agent used in Sweden was as high as 6.9 per cent by weight (Andersson, 2007).

Owing to the widely varying sectors of use and associated life-cycle stages of concern, NPE provides a relevant and interesting case study for examining the scope for economic instruments in chemicals policy. In addition, the discusson has also illustrated that the challenges faced in inducing further reductions in the use of NPE differs significantly across various countries. In the following we will discuss the use of economic instruments for controlling NP and NPE use in general, but we also pay

[11] It should be noted that even textiles produced in the EU may contain NPEs, since the EU Directive (amended 2003/53/EC) only covers their handling in the textile industry's wet processes and not nay chemical residues that are left behind in the textiles (Swedish Society for Nature Conservation, 2008).

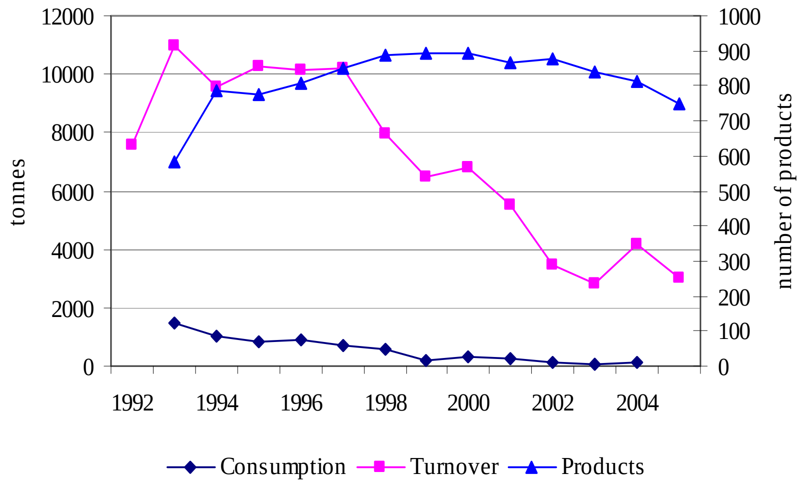
attention to Sweden and the Nordic countries in which releases from stationary sources are close to zero.

4.2.2 The production, use and trade of NPEs in Sweden

In 1997 four companies within the European Union produced NP with a total production of 73 500 tonnes. At this time Sweden was one of the producers, but since the turn of the century NP is not produced in the country anymore. Still, in Sweden large quantities of NPEs are still manufactured, of which the greater part are exported, and with the decline in NP production imports of this substance increased significantly. For instance, in 1994 total imports of NP amounted to about 20 tonnes but in 2004 the corresponding imports were about 2500 tonnes.

Akzo Nobel Surface Chemistry is the only producer of NPE in Sweden; in 2008 the level of production was below 7000 tonnes and it is gradually decreasing. Overall the environmental problems of NP and NPE releases at the production stage are likely to be limited. The production process at Akzo Nobel is sealed and no releases of either NP or NPE take place in the process with the exception of small amounts (less than 50 kg per year, corresponding to less than 10 per cent of the allowed discharge levels), which are washed out during the necessary equipment cleaning processes and controlled in the waste water system of the plant (Mårlind, 2009). The company is continuously working together with its (foreign) customers to find replacements for NPEs in all applications. Currently the production of NPEs represents less than seven per cent of the company's total production of surfactants.

Figure 4.1 Consumption and turnover of NPEs in Sweden, 1992–2005



Source: Swedish Chemicals Agency, Product Register.

Figure 4.1 confirms that the gross turnover (including exports) of NPEs in Sweden has decreased significantly since the mid–1990s but it is still significant. Direct consumption of NPEs is much lower in quantity terms and has also declined over time, implying that Sweden exports the major part of the NPEs produced within the country. Figure 4.1 also shows that the total number of products containing the substance equal around 800 and this figure has remained fairly stable during the last ten years.

Less and less NPEs are being used directly in products manufactured in Sweden for Swedish use (Table 4.1). For instance, as was noted above the use of NPEs in cleaning agents decreased substantially during the early 1990s, and this gradual phase-out has continued. Over the time period 1995–2006, the use of NPEs in cleaning agents decreased from almost 200 tonnes to less than 10 tonnes (Statistics Sweden, 2008). Nevertheless, use is still spread across a large number of sectors. Changes in the quantity of imported products are difficult to judge, because, prior to being made notifiable, some substances have presumably not always been reported to the Products Register. Specifically, in the product register of the Swedish Chemicals Agency the NPE-content of chemical engineering products is recorded, but there are no statistics for the corresponding ones in textile products. In a recent investigation the Swedish Society for Nature Conservation (2008) shows that NPEs are encountered in a majority of the t-shirts imported into Sweden, and the levels of NPEs were generally highest in t-shirts produced outside the EU. Overall in Sweden the import of products that contain NPE represents the biggest challenge in avoiding the negative environmental impacts of this substance.[12]

Table 4.1 Swedish import and production of products that contain NPEs (tonnes)

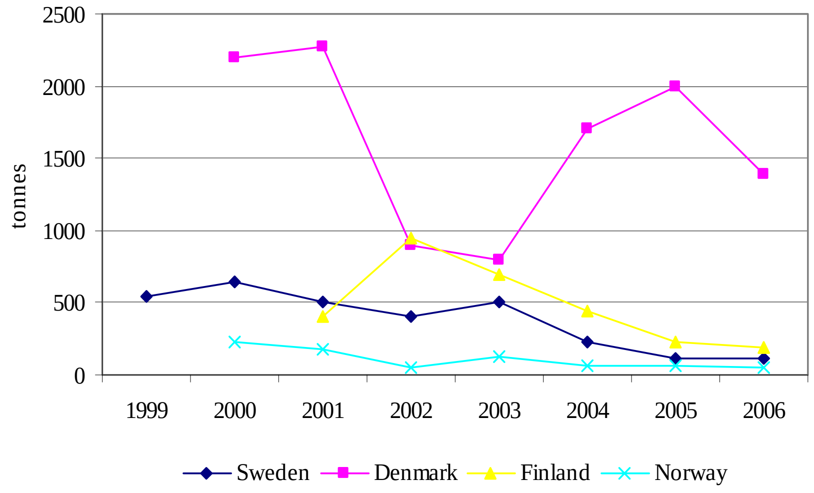
	1995		2000		2005	
Product categories	**Imports**	**Prod**	**Imports**	**Prod**	**Imports**	**Prod**
Binders (for paints, adhesives etc.)	13	370	11	98	8	67
Emulsifier, wetting agents, dispersing agents	44	723	24	20	9	9
Raw material for paint and plastic industry	33	45	72	12	15	<1
Paints	59	99	3	134	3	13
Degreasing and cleaning agents	26	133	13	22	4	3
Lubricants, agents for metal work	23	4	12	10	6	<1
Pesticide, biocide	5	-	14	<1	2	n.a.
Glues	14	-	n.a.	n.a.	<1	1
Impregnating agents, precipitant, diluents etc.	18	4	9	<1	<1	n.a.
Other types of products	n.a.	n.a.	11	5	3	1
Total	235	1378	169	302	51	95

Source: Swedish Chemicals Agency, Product Register.

[12] In the pre-registration for the new REACH legislation, which ended in December 2008, more than 250 legal entities in Europe have registered as producer, importer or a representative for foreign, non-EU, companies for production or import of NPEs to Europe (Mårlind, 2009).

Figure 4.2 is based on figures from the so-called SPIN database maintained by the Nordic Council of Ministers. It shows the amount of NPEs contained in products imported and manufactured (exports excluded) in the four Nordic countries, although care should be taken in making direct comparisons across the different countries due to various reporting strategies and coverage. Still, Figure 4.2 replicates the decrease in Swedish occurrence of NPEs as also shown in more detail in Table 4.1. Related decreases are reported for the other Nordic countries, although in Denmark total turnover of NPEs remain comparatively high.

Figure 4.2 Net turnover (exports excluded) of NPEs in the Nordic countries, 1999–2006



Source: SPIN database, Nordic Council of Ministers.

It should however be noted that the Danish government has been progressive in taking action to ban the production and use of NPEs. As a consequence, the Danish cosmetics and soap industry has removed all NPEs from their products. At present the main use of NPE in Denmark is as hardeners in, for instance, concrete and epoxy products, and in 2004 this constituted about 70 tonnes annually (Kjolholt et al., 2007).

The scope for reducing the negative impacts of NPE will differ significantly across different countries worldwide depending on the regulatory measures already undertaken, but can generally be attributed to a number of measures. Measures can be undertaken at IPPC regulated processes that use NPEs. RPA (2002a) mentions, for instance, the installation of so-called non-contact vacuum systems that reduce releases of NP. Furthermore, in some sectors there exist substitute chemicals or processing methods that are less environmentally damaging (e.g., RPA, 2002a). Such substitutes include alcohol ethoxylates (AEs), which increase in use in spite of a higher price. These alternatives are less toxic and degrade more quickly in the environment (e.g., Toxecology, 2002).

The sectors in which the cost of substitution is comparably low include the textile processing and paint production sectors (see also section 4.2.4). Some of the priority sectors in Europe – cleaning products, textile processing, and pulp and paper – already use significant quantities of alternative surfactants including various AEs. In other sectors, however, the development of viable substitutes is taking longer, and additional R&D efforts are needed. For instance, the use of alternative substances would be relatively costly in the case of NPE used to produce polymer dispersions, which is used in coatings (RPA, 2002a).

The cost of surfactants fluctuates with the price of the raw materials, thus implying that the costs of both NPE and AE fluctuate with the price of ethylene. In addition, the AE price is also sensitive to the price of detergent grade alcohols. Overall, during recent years the prices of the alternatives to NPE have been, on average, 20–40 per cent higher than NPE.

4.2.3 The use of economic instruments for NPEs

The above shows that a large number of policy and voluntary initiatives are already in place to limit the use of NPEs. These measures have achieved substantial reductions, but are not expected to remove the problems entirely. For this reason different types of economic instruments may have the potential to complement these existing regulations and voluntary agreements. Given that the use of NP and NPEs occurs both in point and non-point sources it is relevant to consider both emissions taxes and input taxes, although the choice between these will also depend on the types of measures already undertaken in specific countries.[13] We will also address the policy challenge in targeting imported NPEs in products such as cleaning agents and textiles should also be highlighted.

First, in the case of emission taxes (or fees) these could be applied to activities regulated under IPPC (RPA, 2002a), this since detailed information should be available on the identity of emitters and on the volumes being emitted. The charge could thus be levied on residual emissions following the application of IPPC-based controls. One must however question whether such an approach brings with it significant improvements compared to the current CAC regime; this will in part depend on the extent of variations in abatement costs across firms and on the existing incentives for developing substitutes. RPA (2002a) notes that a charge of about EUR 100 per kg NPEs would be required to provide a *minimal* incentive for emissions reductions. In the Swedish Akzo Nobel case, the economic impact of such a charge would be very modest as their

[13] RPA (2002a) also discusses the establishment of an input trading scheme for the use of NPEs. Such a regulatory approach could theoretically be efficient as there are expected to be less environmentally damaging substitutes in some of the applications of concern. For this reason there would exist incentives for intensive trade. However, an input trading scheme in the NPE case would also likely be more administratively costly than an input tax scheme.

residual emissions amount to only about 50 kg on an annual basis (see above), resulting in a compliance cost of EUR 5000 per year. For this particular producer, this is unlikely to induce additional measures towards reducing these emissions beyond those already undertaken. Based on the experiences of the Swedish producer Akzo Nobel, we anticipate that a charge-based approach to emissions occurring at the NPE production stage would not represent a major improvement compared to the current regulatory regime. However, in other countries, in which emissions from IPPC regulated facilities are still a concern, emissions charges should be considered.

Second, the aim of an input-based tax (or fee) on the use of NPEs and NP is to raise the relative price of NPEs versus substitute inputs of production. An input tax could thus be placed on all sectors that use NPs or NPEs as raw materials, and with an adjustment factor to account for differences between NP and NPE (based on the molecular weight of NP being 60 per cent of that of NPEs) (RPA, 2002a). As was noted above, the potentials and (marginal) costs of such substitution differs greatly across sectors, which strengthens the case for using this type of instrument. Still, this approach also represents a challenge in many respects.

One of the key factors affecting the potential effectiveness of input taxes is the ability to monitor the import of NPEs in products. Unless the tax is also levied on imports it will not fully address the environmental impacts and may put domestic producers at a competitive disadvantage. Today a tax on the NPE content of imported products is very difficult to achieve; although several tests of cleaning agents and textile products have been performed (e.g., Andersson, 2007; Swedish Society for Nature Conservation, 2008), the knowledge needed to achieve an effective tax differentiation across products (i.e., higher tax on products with a high content of NPEs) is still lacking. Clearly, in implementing input taxes one must address the problem of how closely these can approximate the damages done from NP and NPE releases. If the perception among producers and consumers is that such a close propor-tionality cannot be achieved, the policy may face tough resistance and result in significant transaction costs.

In principle an input charge could be placed on all sectors that use either NPs or NPEs as raw materials, including the sectors for which the Marketing and Use Directive applies. However, the charge should also cover sectors where few specific measures already exist, such as, for instance, electrical engineering applications, fuel and oil additives, photographic materials etc. This proposal would not entail charges thus on the NPE-content in imported products but this issue needs to be further investigated to come up with effective regulatory measures.[14] As was noted above, charges of about 20–40 per cent of the price of NPEs could be

[14] Clearly, informative policy instruments as well as targeted inspection activities may play important roles here.

needed to induce a switch away from these substances, and in the next sub-section we discuss compliance costs in greater detail.

4.2.4 Impact assessment

The existing regulations on the use of NP and NPEs in Europe have (ceteris paribus) achieved substantial reductions in environmental impacts from these substances. In a baseline scenario, however, the negative impacts on the aquatic environment are likely to persist in the foreseeable future, not the least due to the sometimes high NPE-content in imported products as well as the fact that the existing regulations do not cover all end user sectors. In this section we focus primarily on a general discussion of the impacts of economic instruments for controlling NPE use with only minor references to specific countries.

In our policy scenario we consider only the case of a charge on the use of NP and NPE as raw materials in different production processes. This would induce the substitution towards alternative substances (e.g., AEs); these may introduce new risks but overall the substitutes appear to be less hazardous than NPEs for most sectors. However, given that substantial measures already are undertaken in many sectors to reduce the dependence on NPEs, one would not expect substantial substitution behaviour. A major benefit of the input charge would however be to induce additional efforts to develop new technology and test new applications and surfactants.

As was noted above, the compliance costs would likely differ substantially across industries and uses, and there is not always a close link between the relative contributions to the environmental burden on the one hand and the volume used on the other. For instance, the costs to the industrial cleaning products sector would be comparatively low, yet would imply substantially reduced environmental risks (RPA, 2002b). Figure 4.3 shows estimates of the per-unit costs of avoiding the use of NPEs in different sectors in the United Kingdom. It should be noted that in Sweden many of the low-cost opportunities for NPE-reduction have already been exhausted.

Chemical intermediates and emulsion polymerisation are not included in the Marketing and Use Directive, and these represent sectors where compliance costs appear to be particularly high. Still, an input charge could have important dynamic incentive effects (i.e., induce search for new applications and materials) in these sectors. Moreover, the levying of a uniform charge of all NP and NPE use (with due weight given the different molecular weights of the two substances) does promote a cost-effective reduction of the uses of these. However, it remains to be investigated how these affect the environmental quality; most notably, charges in sectors with closed production sectors give rise to few direct benefits for the environment (other than reduced risks for spillage etc.) (e.g., Esfahani, 2008).

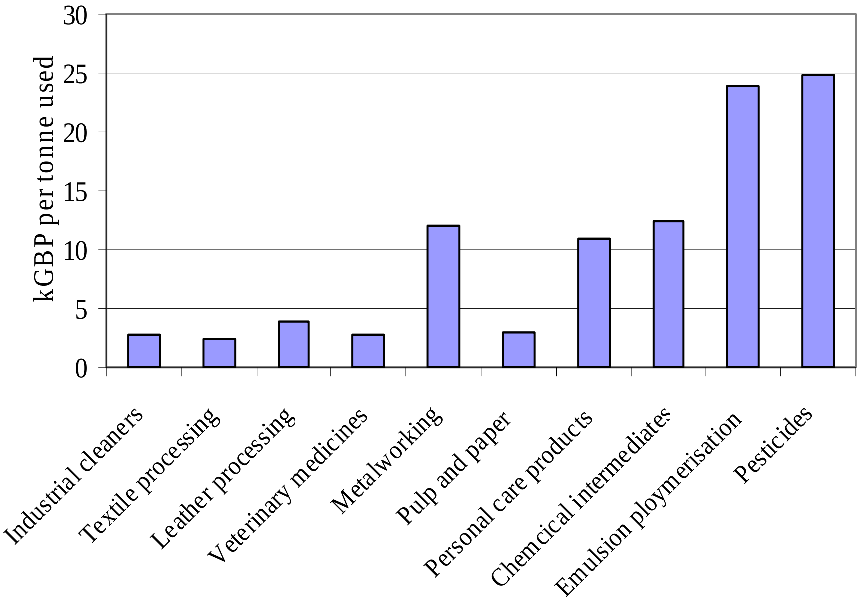


Figure 4.3 Estimated costs of restricted use of NPEs in the UK (kGBP per tonne)

Source: RPA (2002b).

Kjolholt et al. (2007) estimate that on average the cost of reducing NPE use in Denmark (based on subsitution to alcohol ethoxylates) would equal about DKK 30000 (GBP 3000) per tonne, which corresponds well to the lower estimates for the UK. However, the Danish estimate does not take into account the fact that the marginal cost of substitution in most uses gets increases the closer to 100 per cent substitution we wish to go.

The overall economic impacts on society as a whole are clearly uncertain, and call for additional research. The most significant negative impacts of NPE use, those resulting when products with high NPE-content are imported and used in the Nordic countries, would not be targeted with this policy. However, as has been noted above, a major benefit could be the additional research and development activities induced by the new policy, potentially with important spillover effects to other countries. In addition, a uniform tax or charge on NPE use would also target sectors (e.g., those that do not need to comply with the NPPC Directive) that have not faced significant policy measures towards reduced NPE in the past.

4.3 The case of two-stroke oils

4.3.1 The nature of the product, environmental impacts and existing regulations

Two-stroke oils are used as lubricants in small (two-stroke) engines by mixing the oil with the fuel; the oil is burned upon combustion of the air/fuel mixture. Due to their simple design and high power-to-weight ratios, two-stroke engines are suitable for outboard motors, chainsaws, snow mobiles, lawnmowers etc.

Two-stroke engines give rise to significantly higher emissions of different toxic substances than, for instance, four-stroke engines. An important reason for this is that 25–30 per cent of the fuel used, and an even higher share of the oil mixed in the petrol, is not combusted (Hammar and Drake, 2007). In addition, different exhaust purifiers can often not be used in these inefficient engines. Two-stroke oils typically contain about 50 per cent mineral oils, 30 per cent additives (specific content often unknown), and 20 per cent solvents (Lindström, 1998). This results in, for instance, substantial toxic emissions in the aquatic environment (e.g., fish life) due to the use of outboard motors (e.g., Balk et al., 2002). The emissions of carbon monoxide and hydrocarbons such as polycyclic aromatic hydrocarbons (PAHs) from two-stroke engines have been overlooked for many years. The emissions of PAHs are highly toxic and persistent compounds known to be carcinogenic to humans, bioaccumulative and poisonous to the marine environment (Landrum et al., 1987). SEPA (2002) reports that the extensive emissions of PAHs from boats are likely to contribute to the decreasing amounts of fish along the Swedish coast.[15] In Northern Sweden emissions of hydrocarbons from snowmobiles give rise to related negative health and environmental impacts.

The use of alkylate petrol in two-stroke engines can reduce the emissions of PAHs by 80 per cent (SEPA, 2002), and since 2002 these fuels have lower tax rates but are still slightly more expensive than, for instance, regular (95 octane) petrol (Riksdagens förvaltning, 2008). Similar tax rebates do not exist for two-stroke oils, however. In the early 1980s Germany, Portugal and Switzerland regulations introduced regulations mandating the use of environmentally friendly two-stroke oils (Theodori et al., 2004). Moreover, the eco-labels the Nordic Swan and the EU Flower (see also below) include two-stroke oil in the product group for lubricants. A recent EU Directive (2003/44/EC) amending the Recreational Craft Directive (1994/25/EC) puts new limits on exhaust gases, which may make it economically unattractive to produce two-stroke engines. This also means that lubricant manufacturers may not show com-

[15] According to US EPA (1998), using a 70 horsepower two-stroke engine for one hour emits the same amount of hydrocarbons as driving about 8000 kilometers using a modern automobile.

mercial interest in developing additional oil products that comply with, say, the performance criteria of the European Eco-label.

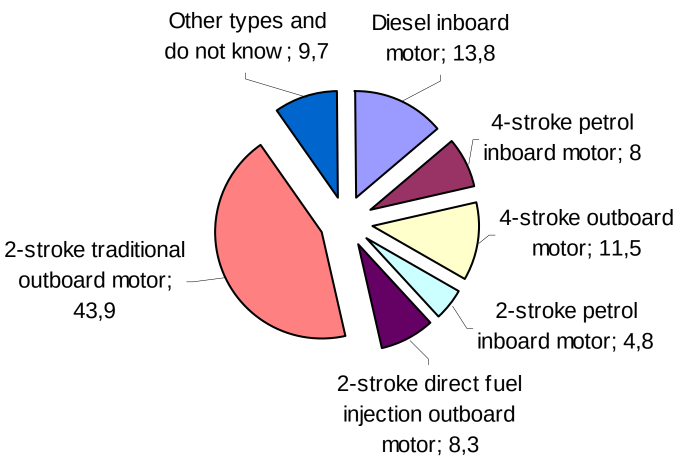
Nevertheless, the most important argument for additional policy intervention in this area is that the private users of both smaller boat engines and snow mobiles in the Nordic countries are likely to continue to use two-stroke in the future, not least because of cost reasons. The low capital turnover of two-stroke engines implies that many of the environmental problems caused by these – often in sensitive areas – will persist unless new or complementary regulations are put in place.

4.3.2 The use of two-stroke oils in Sweden

The demand for two-stroke oils is a derived demand, and it is useful to look in more detail at the nature of the different uses. In the following we will focus on personal use of two-stroke oils and in particular on applications in small boat engines and snowmobiles, respectively. These uses are prevalent in the Nordic countries, given the cold climate and the relatively long coast lines.

The total number of small private boats is not easily estimated, but the so-called Boat Life investigation from 2004 indicated that the correct figure lies in the range of 650,000 to 780,000 (excluding boats owned, by, for instance, charter companies, yacht clubs etc.). Among these at least about 500,000 have boat motors (Kågeson, 2000; Edman, 2008). Figure 4.4 shows how this total is distributed among the different types of boat motors.

Figure 4.4 Private boat motors in Sweden by category (per centage shares for 2004)



Source: Statistics Sweden (2004).

At least 60 per cent of all small boat engines are of the two-stroke type. Although this represents an improvement compared to the situation in the 1990s, when four-stroke and diesel engine motors accounted for only 6 and 9 per cent of the total (Burman and Johansson, 1997), respectively, the two-stroke engine still dominates the Swedish stock of private boats.

In part due to the above-mentioned EU Directive, which postulates strict emissions regulations that restrict the supply of two-stroke engines, total sales of two-stroke outboard motors have declined in both absolute and relative terms over the time period 2002–2007 (see Table 4.2). The sales of four-stroke engines have increased, and new two-stroke engines with fewer emissions generated are also penetrating the market. However, there are no restrictions in the case of existing boat motors, and given the very low capital turnover (3–4 per cent annually at a maximum) the environmental problems are still severe. Outboard motors are often maintained for decades (Lindström, 1998).[16] There exist today (at least ten different) environmentally friendly, biodegradable two-stroke oils for, for instance, outboard motors and chain saw engines (Hörner, 2002),[17] but these are still more expensive than the traditional ones. The regular price for traditional two-stroke oil is in the range SEK 70–90 per litre, while the "greener" alternatives are about SEK 20 more expensive per litre.

Table 4.2: Sales of outboard motors in Sweden (number of engines sold)*

	2002/03		2004/05		2006/07	
Product categories	**Number**	**%**	**Number**	**%**	**Number**	**%**
Regular two-stroke engines	8949	47	8764	44	3600	16
Low-emitting two-stroke engines	471	2	1003	5	1734	8
Four-stroke engines	9686	51	10234	51	16746	76
Total	19106	100	20001	100	22080	100

* Include sales from members of the Association of Swedish Marine Engine Manufacturers and Importers.
Source: Swedish Marine Industries Federation (2009).

Out of the existing two-stroke outboard motors (about 211,000 in 2004) over two thirds are run with the regular – cheaper – type of oil, while less than one fifth are fuelled with the biodegradable quality (Statistics Sweden, 2004). People state in surveys that an important reason for this bias against the less environmentally friendly alternative is the higher price for the environmentally benign alternative, but some also state that the motor may not work properly with this type of oil (Ibid.). In 2004 total consumption of two-stroke oils equalled about 2000 cubic meters per year, and out of this about 1000 cubic meters were emitted (in non-combusted forms) into the environment (SEPA, 2004). The current (2008) consumption is estimated at 2500 cubic meters (Ahlbom, 2009). This implies that the PAH emissions from private small boats is at least 50 tons on an annual basis (IVL, 2008).

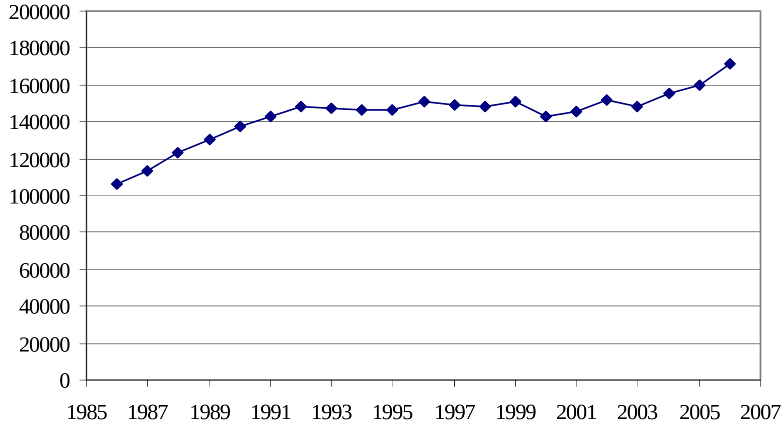
Figure 4.5 shows the total number of snowmobiles in use in Sweden over the time period 1986–2006. The use of snowmobiles has increased over time, and a majority of the current stock is run with two-stroke engines. Still, since 2001 the four-stroke engine has been launched also on

[16] Some Swedish municipalities (e.g., Allingsås) are however introducing bans for two-stroke engines for water quality reasons in selected waters and lakes (Blom, 2009).

[17] Rapidly biodegradable two-stroke outboard engine oils were first developed in the 1970s, but similar oils for other lubcricants only began to be developed in the early 1980s (Hörner, 2002).

this market segment, and as of 2005 their market share (out of total annual sales) amounted to about 30 per cent.[18] The annual turnover of snowmobiles in Sweden is about 5–6 per cent. This implies that about 25 per cent of the stock of snowmobiles in 2006 was 20 years or older (SEPA, 2007).

Figure 4.3: Snowmobiles in traffic in Sweden, 1986–2006



Source: SEPA (2007).

It may be noted that apart from private boat motors and snowmobiles, there existed in Sweden (2006) about 240,000 chain saws, 30,000 brush saws and 387,000 hedge trimmers with two-stroke motors (SEPA, 2007). Lawnmowers with two-stroke engines have however not been sold in Sweden during the last 30 years.

4.3.3 The use of economic instruments for two-stroke oils

In Europe the environmental problems associated with the use of two-stroke oils have been recognized (Hörner, 2002), and fairly recently producers of lubricants in the European Union are able to display the EU Eco-label (the Flower) on their products. This will help consumers to select the two-stroke oils that reduce harm to water and soil when used. The eligible lubricants will have to respect specific environmental criteria (Theodori et al., 2004). Still, the experiences from Sweden indicate that informative policy instruments have proved to be ineffective in this field, partly because of the extra cost of environmentally friendly oils but also since consumers are reluctant to switch to a product of “uncertain” quality (SEPA, 2004).[19] In Sweden no specific measures have been undertaken to manage the problem of two-stroke oil pollution, but a number of

[18] This information has been obtained from Snöfo, the association of the snow vehicle suppliers of Sweden.

[19] In addition, research within the so-called SHARP program (www.sharpprogram.se) shows that only about 10 per cent of Swedish households recognize the Flower as an eco-label when confronted with it (Söderholm, 2008).

measures have been suggested and are under way. One example is that exhaust purifiers are increasingly required even for two-stroke engines (Ibid.). Such a policy, however, introduces a new-source bias, and increases the incentives to prolong the lives of existing engines. A similar, contra productive, effect would emerge if a tax was levied on, say, the purchase of new two-stroke engines (see also SEPA, 2007).

In this sub-section we briefly discuss three alternative economic instruments that could be used to address the pollution problems arising from the use of two-stroke oils. The *first* option is a tax rebate on the purchase of environmentally friendly (biodegradable) oils; this would be similar to the existing rebate on alkylate petrol. However, a major problem with such an approach is that although it induces substitution away from the non-biodegradable oil, it also promotes increased overall use of two-stroke oils. This is far from uncontroversial. Two-stroke oils are complex products, and even the most environmentally friendly products currently marketed pose environmental risks (Theodori et al., 2004).

Second, SEPA (2004, 2007) discuss the idea of a refund premium for those who scrap their old two-stroke engines. Such a scrap premium is also endorsed by the Liberal party in Sweden (Folkpartiet, 2006). However, such a policy does not target the environmental damages from *consuming* the oils; it is best suited for cases in which improper disposal of the engines causes negative environmental damages. Moreover, many old engines that are collected may not be used and if so there is no environmental benefit of scrapping these. In sum, at the margin a scrap premium does not create any additional incentives to use less oil and/or switch to less-damaging oil qualities.

Third, in this report we instead analyze in more detail the suggestion of a differentiated tax (or fee) on two-stroke oils put forward by SEPA (2004) and Hammar and Drake (2007). The tax premium on regular two-stroke oils should be high enough to encourage the substitution of biodegradable oils for non-biodegradable ones. Such a policy would encourage flexibility and it would not discriminate between different technological solutions. Moreover, the number of consumers is very large and they are likely to face different switching costs, and since the number of suppliers is much more limited the tax (fee) could be targeted at these. This would keep the tax administration costs at a relatively low level.

Regular non-biodegradable two-stroke oils cost in the range of SEK 70–90 per litre, while the biodegradable alternatives typically are about SEK 20 more expensive. Also in this case earmarking should be seriously considered, and the tax revenues could preferably be used to clean-up contaminated harbour sites (Hammar and Drake, 2007). SEPA (2004) estimates that at a tax level of SEK 20 per litre the total tax revenues would initially equal about SEK 50 million but decrease over time at a pace determined by consumer switching behaviour. Such an earmarking strategy would likely increase the public's acceptance of the policy and also give rise to important signalling effects.

Thus, although information alone has proved to be a rather toothless policy instrument in this case, the combination of economic incentives and information could prove effective. Information provides an awareness-raising impact while economic instruments provide an explicit incentive to substitute greener for more environmentally damaging products.

4.3.4 Impact assessment

In discussing the impact of a differentiated tax on two-stroke oils we assume a reference scenario where no other national policy instruments are put in place. In addition, we anticipate that the EU Directive on recreational crafts will imply substantially reduced sales of new two-stroke engines for private boats. To a large extent this phase-out is well underway. Still, given the low annual capital turnover the aquatic pollution (e.g., PAHs) from two-stroke oil engines is likely to persist in the foreseeable future. In other words, in the baseline scenario the emissions from two-stroke oils would probably decline over time but at a comparatively low rate, thus motivating the introduction of more targeted policies.

In order to assess the impact of the differentiated tax on consumer choices it is important to estimate the own-price elasticity of two-stroke oil demand. Unfortunately the data are not available to estimate these elasticities empirically, and we need to rely on qualitative assessments instead. The own-price elasticity of demand is likely to be relatively low but far from insignificant. The purchases of two-stroke oils typically constitute a very low share of total household budgets, and even though motor oil is a complement to another good (e.g., private boats) that may represent a significant share of the budget the own-price elasticity of demand is likely to be low.

The availability of substitute biodegradable two-stroke oils implies, however, that the cross-price response following a differentiated tax may be significant. The quality differences between biodegradable and non-biodegradable are by all means insignificant for most users. Some, however, may perceive these differences to be important, not the least those who, for instance, compete in snowmobile and/or boat competitions. Still, any perceived quality differences should not represent major obstacles to product substitution in this case.

The overall economic impacts on private consumers of a differentiated tax are likely to be small, not the least given the low budget share for two-stroke oils and the availability of substitutes. The new policy does not require users to substitute any equipment per se, e.g., to abandon the two-stroke engine; rather it grants users substantial flexibility in choosing the most cost-effective compliance measures. Some professional users – e.g., Sami villages in Northern Sweden relying heavily on snow-mobiles in reindeer herding – may face notable total cost increases, but even for these professional users the overall economic impact is likely to be modest.

The suppliers of two-stroke oils will of course also be affected; they are provided with an incentive to switch supply towards the less environmentally damaging oil qualities. Swedish suppliers (e.g., Statoil, Total Sweden AB etc.) typically offer different types of two-stroke oils today, and a differentiated tax would mainly imply a switch between the different qualities while total sales are likely to be only modestly affected. As was noted above, the pure output effect of the tax is likely to be low. It should also be noted that even though there is an existing price difference between the different two-stroke products this does not necessarily translate into real production cost differences. For the greener products it may be easier for producers to target consumers with a high willingness-to-pay, thus resulting in higher prices.

The impact on the tax authorities is also limited given the fairly low number of suppliers in Sweden. In such cases the administrative costs of tax collection and enforcement have been estimated at about one per cent of total tax revenues (Swedish Chemicals Agency, 2008c). There is clearly a risk of cross-border trade in some parts of Sweden, this to take advantage of any price differences across countries. The administrative costs of taxing private imports could raise the financial burden of the tax authorities but ought not be prohibitive.

Finally, the overall benefits and costs to society as a whole are difficult to assess in detail. Still, given the existence of readily available substitutes to the most hazardous two-stroke oils – and the ensuing low compliance costs – in combination with the significant negative impacts, not the least on the aquatic environment, the net social benefits are likely to be positive.[20] Moreover, in Sweden the suggested policy represents a non-negligible step towards achieving the environmental quality objective "a non-toxic environment".

[20] It should be noted however that this is no more than an educated guess; we have not been able to identify any damage cost study of, for instance, the impact of PAHs on the natural environment in Sweden. The fact that some municipalities jave introduced bans for two-stroke engines in selected lakes, indicates hower that from a revealed prefernce perspective the valuation of the associated negative impacts appears to be fairly high.

4.4 The case of 1,2-ethanediol (ethylene glycol)

4.4.1 The nature of the product, health impacts and existing regulations

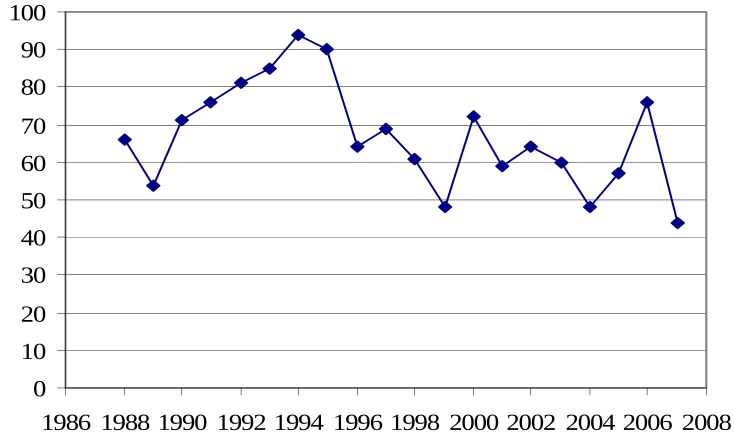
Ethanediol is a clear, colourless liquid with no odour. Worldwide, the largest area of use is in the production of polyester fibres. Its solvent properties also make it useful in the production of latex paints, surface cleaners, printing inks etc. In Sweden and the Nordic countries, though, the largest use of ethanediol is antifreeze in the cooling water systems of cars and as a deicing agent (see also section 4.4.2).[21] Ethanediol has been used as an antifreeze agent since 1925. The cooling liquid picks up heat from the engine without boiling at a temperature above 100 degrees C. Furthermore, aircrafts and runways are sprayed with deicing mixtures containing ethanediol to prevent the formation of ice.

Ethanediol is oxidized in living organisms to among other things oxalic acid, which is highly toxic. Near-accidental poisoning of children through unintentional oral ingestion occurs every year; about 60 cases of drinking exposure per year are reported annually in Sweden and even though this problem was brought to attention already 20 years ago the number of cases has not appeared to decrease over time (Figure 4.6). More than 90 per cent of the reported cases are deemed to be unintentional. Children like the sweet taste of the substance and less than 0.05 litres can cause life-threatening damages. Adults can die from just drinking 0.1 litres, and oral ingestion can also lead significant damages to the liver. The lethal dose for humans is about 1.5–2.0 ml per kg.

Oral ingestion of ethanediol also occurs in suicide cases as well as in the case of small animals, especially dogs and cats. These pets like the sweet taste of the glycol and just a small amount of licking from splashes of antifreeze may lead to serious damages and kidney failures. Cases of unintentional poisoning have also been identified among alcoholics, who drink, for instance, washer fluids, which may also contain ethanediol (Breider and Nilsson, 2003). The lethal dose for humans is about 1.5–2.0 ml per kg. The health accidents involved in ethanediol use were addressed already in the late 1980s and this resulted in a warning label on all glycols containing ethanediol, but this measure has proved to be ineffective (Swedish Chemicals Agency, 2004). In 2004 the Swedish Chemicals Agency recommended suppliers of glycols to apply child-proof seals and/or insert acrid substances to the liquid in order to remove the sweet (and thus tempting) taste.

[21] Globally only about 10-15 per cent of all ethanediol is used for de-icing and antifreeze etc.

Figure 4.6 Inquiries to the Swedish poisons information centre (GIC) concerning children younger than ten years having been exposed to ethanediol-based antifreeze



Source: Swedish Poisons Information Centre (GIC).

Figure 4.6 shows that these measures appear to have had only temporary effects on the number of reported cases of children exposure, given the relatively low number of reported cases in 2004.[22] One important reason for this is probably the difficulty in using information campaigns to promote permanent behavioural changes. Furthermore, the main suppliers of glycol have implemented the recommendations of child-proof seals. However, during recent years the number of small suppliers has increased significantly, including the imports of low-price glycol from other countries. These supplies are more difficult to target with information campaigns and they may not face strong incentives to follow the recommenddations anyway.

4.4.2 The use of ethanediol and its substitute in Sweden and the Nordic countries

Ethanediol is produced in Sweden, but the main part is imported as a component in chemical products. Propanediol is a substitute for ethanediol as a non-freezing solution for cars due to its less dangerous properties for human health.[23] Thus, overall the properties are similar to ethanediol but propanediol is less toxic to humans and animals. Figure 4.7 shows the total annual turnover of these two substances over the time period 1992–2005.

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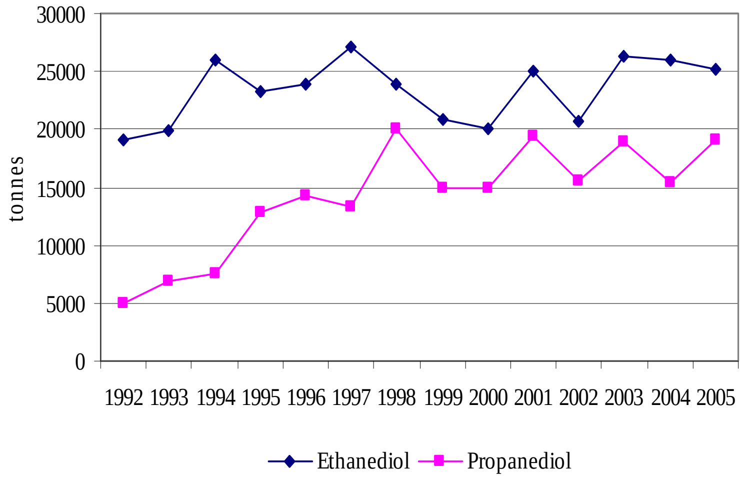
[22] According to the Swedish Plastics and Chemicals Federation a TV-commercial about lighter fluid can in part explain the decreased number of reported cases of glycol exposure between 2006 and 2007.

[23] In Sweden propanediol is also used as an industrial solvent in cleaning agents, paint and surfactants.

Figure 4.7 The total annual turnover of ethanediol and propanediol in Sweden (tonnes)



Source: Swedish Chemicals Agency, Product Register.

Ethanediol turnover has been rather constant over the years, while the turnover for propanediol has increased fourfold during the last ten years. The same pattern is not evident in the other Nordic countries. In Denmark, for instance, we have witnessed a substantial decline in propanediol use over the time period 2000–2006, from about 10,000 tonnes to less than 2000 tonnes. The consumption level in Norway is also around 1500 tonnes. At the same time the use of ethanediol is higher in Norway and Finland compared to Sweden (see Table 4.3).

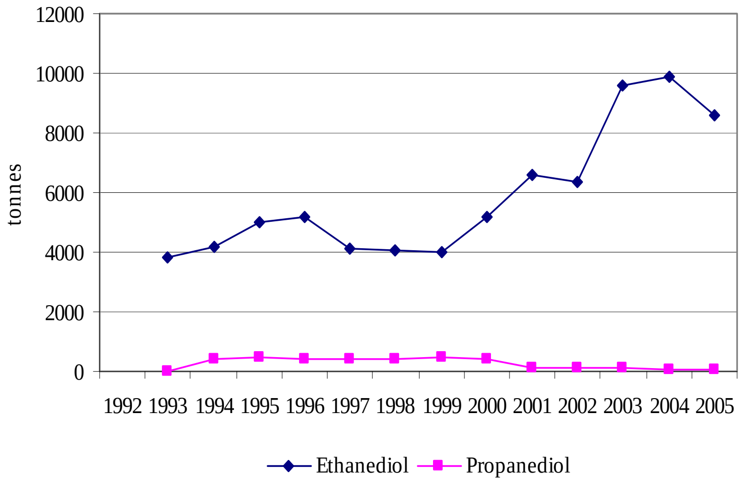
Table 4.3 Nordic estimated use of ethanediol and propanediol (tonnes), 2006

	Sweden	Finland	Norway	Denmark
Ethanediol	25100	51200	82100	12800
Propanediol	17300	n.a.	1340	1170

Source: Nordic Council of Ministers, SPIN database.

Still, these figures include use in the de-icing of aircrafts, and if we only investigate the case of ethanediol sold as antifreeze preparations to households a completely different picture emerges. Figure 4.8 provides an illustration of this for Sweden. In 2005, about 8600 tonnes of ethanediol were sold in *consumer available* glycol in motorcar antifreeze products, and this represents more than a doubling since 1993 (when the corresponding sales corresponded to less than 4000 tonnes). At the same time, the use of propanediol has remained low and stagnant during the same time period and in 2005 it amounted to only 60 tonnes (Statistics Sweden, 2006). Thus, there is currently no sign of substitution of propanediol for ethanediol in the case of glycols. Similar detailed data are not available for the other Nordic countries.

Figure 4.8 Glycol in consumer motorcar antifreeze products in Sweden, 1993–2005



Sources: Statistics Sweden (2006) and Swedish Chemicals Agency, Product Register.

One reason for the limited substitution is that propanediol poses a higher risk for corrosion (Swedish Chemicals Agency, 2004). For instance, in the case of ethanediol there is a five-year protection for corrosion, while in the propanediol case the same protection is only two years. Ethanediol glycol has a better property for transferring the heat and cooling the engine than propanediol. Newer engines are optimized for causing low emission levels; for this reason they have a high combustion temperature, something which biases use towards ethanediol. Still, the lower quality of propanediol is also debated (e.g., at web forums for car users), and the product is sold to consumers for essentially the same price as ethanediol (about SEK 60–70 per litre) (see Figure 4.9).

Figure 4.9 Glycol alternatives for car engines: an example

Ethanediol-based glycol	*Propanediol-based glycol*
Corrosion protection: 5 years	*Corrosion protection: 2 years*
Price: SEK 68 per litre	*Price: SEK 68 per litre*

Source: www.bildelar.nu.

It should also be noted that overall engine tests for a large number of vehicles have almost exclusively been done for ethanediol. Re-doing the test for propanediol is time-consuming and costly for the car producers, and they therefore often lack the incentives to perform these tests.

Table 4.4 shows the Swedish import and production of chemical products that contain ethanediol, and it confirms that most of the use is imported and ultimately used as a de-icing agent or in non-freezing solutions. The total number of products containing the substance amounted to 1044 in 2004, out of which 305 were consumer products. In the case of propanediol a similar picture emerges; in 2004 only about one tenth of the propanediol used in de-icing and antifreeze was produced in Sweden, while the remainder was imported.

Table 4.4 Import and production of products that contain ethanediol (tonnes)

	2001		2004	
Product categories	**Imports**	**Prod**	**Imports**	**Prod**
De-icing agent, non-freezing solutions	10597	1090	9794	2438
Car care products	329	-	137	-
Paints, including raw material	28	298	95	68
Cutting agents and lubricants	295	27	1063	37
Corrosion inhibitor	298	<0.1	n.a.	n.a.
Adhesives, including raw material to adhesive	22	183	37	223
Pesticide, biocide	55	134	22	0
Cleaning agents, including raw material	n.a.	n.a.	13	334
Raw material for plastic	n.a.	n.a.	66	0
Sealing compounds and binders	n.a.	n.a.	10	32
Other types of products	104	223	49	63
Total	11728	1955	11286	3195

Source: Swedish Chemicals Agency, Product Register.

The above pattern of substantial imports of glycol is important for the implementation of any economic instruments promoting product substitution in the Nordic countries. Such instruments would probably only have limited impacts on the manufacturers' incentives to improve the quality of their products. This conclusion is strengthened by the fact that the health problems associated with ethanediol use have so far only been highlighted in the Nordic countries, and they appear not to be a major concern in other EU countries.

4.4.3 Economic instruments for reduced ethanediol use

The recommendations to suppliers of glycols to apply child-proof seals and/or insert acrid substances to the liquid are essentially "technology standards" and one weakness of such a policy is that it provides very limited flexibility in terms of compliance measures. For this reason, and the past failures in affecting use on a more permanent basis, we believe it is worthwhile to consider incentive-based policy instruments instead.

The case of the negative health impacts of ethanediol use is a typical example of a downstream externality, which is difficult to internalize close to the source. However, an upstream tax on the use of ethanediol-containing glycol could well represent a decent second-best alternative.[24] The tax would be imposed on the purchase of these glycols, encouraging a switch to propanediol-based glycols. Given the often non-existing price difference between the two alternatives, the tax can probably be fairly low and still induce significant substitution behaviour (at least for those who have no strong preference for corrosion protection) (see also below). It is reasonable to believe that families with children would be particularly keen to switch; the tax provides both a direct incentive to do so but it can (if combined with information) also provide an important signalling effect. Previous research shows that households with children generally are more willing to pay premiums for eco-labelled products.

People without small children may be less enthusiastic about the switch but this does not matter much; in these cases the risk of severe accidents is very small. The legitimacy of the policy could be increased further if any tax revenues are recycled back to those suppliers who, for instance, use child-proof seals on their ethanediol-containing products. However, in this case the scope for such an ear-marking strategy would probably be limited, no the least since it may be difficult to identify and monitor the large number of diverse suppliers.

An important issue concerns whether only consumer (i.e., household) products containing ethanediol should be subject to the tax. We believe this should not be the case since this could induce households to purchase glycols through unconventional channels. The storing of glycols in, for instance, soda bottles could then increase, also increasing the risk that children or other people accidentally drink the substance. Targeting all products containing ethanediol also widens the tax base, and opens up for additional uses of the tax revenues. Still, significant uses of ethanediol occur in, for instance, motorcar repair shops and here the extent of the damages are likely to be very low. This will limit the cost-effectiveness of a fee on overall ethanediol use, and other economic instruments must also be considered. An additional drawback of the proposed product charge is that household members face no additional incentive to store bottles in safe and child-proof manner. However, it is hard to imagine a policy instrument that achieves this.

From experience we know that information campaigns alone have had little impacts in this area. Although people are generally aware of the problems with ethanediol they may well not perceive *their own* purchases as causing health problems; people tend to believe that they are more careful than the average person. This strengthens the case for the introduction of explicit economic incentives. Still, this does not imply that information should be rejected as a policy instrument for addressing the problems of ethanediol. One reason for this is that information campaigns

[24] Similar to the two-stroke oil case, by implementing a subsidy on the “good” product rather than a tax on the “bad” one could well achieve the same type of substitution behavior but would promote too much overall glycol consumption.

targeting child protection can sometimes have important effects on problem awareness and even consumer behaviour (e.g., Orbicon, 2007). Moreover, informative policy instruments would represent important complements to any economic incentive imposed, thus signalling to consumers why the tax has been put in place and that there is money to be earned by switching products.

4.4.4 Impact assessment

In our baseline scenario it is reasonable to assume that no additional policy instruments are put in place above the existing recommendations from the Swedish Chemicals Agency (see above). The reported cases of ethanediol poisoning is then likely to remain constant over time, and thus no significant positive health impacts would be likely to emerge (especially given the absence of a downward trend as illustrated in Figure 4.8).

With the imposition of a per litre tax on the purchase of ethanediol-based glycol, the consumption of this product would go down. Overall, however, the own-price elasticity of demand is likely to be low, not the least since glycols represent a small share of total household budgets. Nevertheless, given the availability of a now cheaper substitute the pure substitution effect may be significant, at least among those users who do not care much about the corrosion impacts of the different products. Still, as was noted above, the impact on consumer behaviour may be modest unless the tax is combined with information campaigns highlighting the health problems associated with ethanediol use.

The suppliers of glycol will be affected by the tax in the sense that their product mix will be altered. Still, as was noted above, we are unlikely to witness any strong incentives to change product content and search for new alternatives among (the mostly foreign) manufacturers. An important impact, however, may be that car manufacturers and suppliers in the Nordic countries will face much stronger incentives to test using propanediol in different types of engines and vehicles.

The choice of a product charge should make the matter rather simple for the tax authorities, but the total tax base is likely to be rather low. In Sweden a tax of, say, SEK 4–6 per litre implies a tax base of only SEK 40 million per year (and even less in tax revenues depending on the intensity of switching behaviour). This suggests tax administration costs of less than SEK 400,000 on an annual basis (one per cent of total tax revenues). However, if the tax (fee) is to be earmarked as well as complemented with information campaigns, the costs of administering the overall policy will be considerably higher.

The overall economic impacts on society as a whole are clearly uncertain, and call for additional research. One the hand one should note that overall people tend to have a fairly high willingness-to-pay to avoid negative health impacts on small children. A combination of a product tax and information playing the role as an attention-seizing function may well reveal such strong preferences. The suggested policy does however impose significant costs for the users of glycol, not the least in the form of a higher risk for corrosion. Some users are likely to have a high willingness-to-pay to avoid these impacts, but for others it is of less concern.

5. Conclusions and implications

In this report we have analyzed the potential for increased use of economic instruments – primarily taxes and fees – in chemicals policy. The report has discussed some theoretical aspects of such instruments, and reviewed the practical experiences of chemicals taxation in the Nordic and other European countries. A last chapter has discussed three proposals for extended use of economic instruments in Sweden and the Nordic neighbours.

Overall the report has shown that there is a clear scope for extended use of economic instruments in chemicals policy (see also Swedish Chemicals Agency, 2008c), but a number of challenges and implementation issues have also been highlighted. These include, not the least the trade-off between costly monitoring on the one hand and the achievement of a cost-effective allocation of abatement measures on the other. Many chemical substances give rise to significant non-point source emissions. These are typically very difficult and costly to control, not the least if the environmental damages caused by the substances are affected by geographical location and the receiving environment's condition. The targeted chemical may exist in many products as well as in a wide variety of end uses. It may thus be more efficient to simply tax the production or the use of chemical compounds upstream in the product chain.

Still, even though second-best incentive-based policy instruments may have an important place to fill in chemicals policy it also important to design instruments that as closely as possible can target the associated damages on the environment. 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.

In sum, an important condition for the successful use of economic instruments in chemicals policy is that damages done are targeted as closely as possible (without imposing very high administrative costs). In many cases instruments that address the use and/or production of chemical substances or products are reasonably cost-effective second-best measures;25 it is however important to avoid, say, environmental taxes that have more fiscal rather than incentive-based impacts.

[25] The case of taxing the lead-content of different types of ammunition is a good example of such a policy (see Swedish Chemicals Agency, 2008c).

We have also stressed that in considering the use of taxes and fees in chemicals policy it is useful to seriously consider: (a) earmarking of tax revenues as a way of gaining overall policy acceptance as well as for funding research and/or clean-up activities; and (b) the need to combine the use of economic and informative policy instruments. It is important that the policy instruments implemented provide both explicit economic incentives as well as awareness-raising effects, not the least in the case of downstream externalities (e.g., household use of chemicals). Still, earmarking of tax revenues can never be an excuse for the implementation of otherwise inefficient economic instruments.

In implementing new economic instruments it is also important to take into account the relationship between these and the already existing regulations in place. In the chemicals field it may typically be a good idea to combine economic and administrative policy instruments rather than to rely on a single type of policy. Given the hazardous nature of many chemicals combined with genuine uncertainty about the marginal abatement costs, it is often motivated to regulate quantities (e.g., emission limits). Still, different types of economic incentives may constitute important complements in improving the cost-effectiveness of the policy.

A key output of this report has been the identification of three cases of chemical use in which the use of economic instruments potentially could help reduce negative health and environmental impacts. We have discussed the nature of the chemicals use, the characteristics of the involved sectors and consumers, as well as potential designs of different economic instruments. The importance of policy design concerned both the economic effectiveness of the instruments as well as the legitimacy of the policy. The cases should not be look upon as first-best cases that deserve to be prioritized in future policy decisions; rather they have been chosen so as to illustrate different types of challenges in relying more extensively on economic instruments in chemicals policy. In each of the cases there are clear challenges in the implementation of the proposed instruments; overall these are linked to problems of implementing taxes and charges which closely targets the relevant environmental impacts.

In the NPE case, it is difficult given the existing scientific knowledge to identify an economic instrument that can address the NPE-content of imported products (textiles, cleaning agents etc.). This problem deserves additional research before efficient policy instruments can be put in place. An input tax based on the amount of NP and NPE used in the production of products is discussed instead; it could have only limited direct environmental impacts domestically but could nevertheless induce the development of new technology permitting cost-effective diffusion of new and less toxic surfactants in the long-run.

A differentiated taxation of two-stroke oils represents a rather promising regulatory measure towards addressing the water pollution and health impacts from the use of oils in two-stroke engines. This tax policy is reasonably easy to implement and given the existence of readily available substitutes the compliance process ought to be rather smooth. No significant economic impacts on the user collective (e.g., owners of private boats and snowmobiles) are expected. The challenge in this case probably lies in achieving significant consumer responses following the tax increase, i.e., increased demand for regular – non-biodegradable – two-stroke oils, given the low share of two-stroke oils in household budgets. For this reason it probably becomes essential to combine the explicit economic incentive with information campaigns about the impacts on, not the least, the marine environment.

The case of an output tax on the use of ethanediol-based glycol is also straightforward, given the availability of a reasonable substitute. However, in this case the health problems following oral ingestion of ethanediol are diffuse and differ widely across different consumer groups. Unlike the two-stroke oil case, in which the environmental impacts are fairly highly correlated with the consumption of the product in question, the health impacts of one litre of ethanediol-based glycol will be zero in many cases. This suggests that the implementation of a uniform per litre tax on this glycol only is a second-best policy instrument that does not promote a cost-effective reduction in oral ingestion incidents. Still, due to the limited impacts of previous information campaigns and bottle design recommendations, a tax could well represent a good first step towards a more permanent reduction of this problem. Also in this case, though, informative policy instruments should constitute an important complement to the economic incentives provided by a tax.

In sum, the above analyses show that the potential for increased use of economic instruments ought to be significant. At the same time, however, this does not necessarily imply that taxes, fees etc. ought to replace the existing policy instruments (e.g., information), but rather that the economic instruments complement these existing measures.

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Sammanfattning

(summary in Swedish)

Det övergripande syftet med rapporten är att analysera potentialen för ökad användning av ekonomiska styrmedel i kemikaliepolitiken. Mer specifikt tillhandahåller rapporten: (a) en konceptuell diskussion om den roll som ekonomiska styrmedel kan spela för att reducera de negativa effekterna av produktion samt användning av kemikalier; (b) en detaljerad översikt och analys av de europeiska erfarenheterna av skatter och avgifter för kemikaliepolitiska syften under de senaste decennierna; samt (c) en utvärdering av hur användningen av tre olika kemiska produkter och substanser kan regleras med hjälp av olika typer av ekonomiska styrmedel. De fall som analyseras är användningen av: (a) nonylfenol och dess etoxylater; (b) olika typer av tvåtaktsoljor; samt (b) etylenglykol.

I rapportens andra kapitel presenteras en översikt av de ekonomisk-teoretiska aspekterna på kemikaliekontroll och styrmedelsval. Analysen bygger på centrala lärdomar från den miljöekonomisk litteraturen, och behandlar i första hand vilka faktorer som påverkar valet mellan olioka styrmedel för att reducera de negativa effekterna av kemikalier. Användningen av ekonomiska styrmedel motiveras av en strävan att internalisera s.k. externaliteter, och utifrån ett effektivitetsperspektiv är det eftersträvansvärt att identifiera styrmedel som så nära som reflekterar de aktuella skadorna på hälsa och miljö. Många av dessa effekter är samtidigt diffusa och en träffsäker styrning kan därför bli dyr, inte minst i de fall där de miljöskador som substanserna ger upphov till varierar beroende på geografisk lokalisering. I sådana fall kan det vara mer effektivt att t.ex. beskatta produktionen eller användningen av den kemiska substansen.

Den teoretiska litteraturen om miljöskatter på kemikalier lyfter fram den trade-off som existerar mellan en dyrbar kontroll av utsläppen eller skadorna som sådana, och den billigare strategin att beskatta användningen av kemikalien. Litteraturen uppmärksammar också det faktum att förekomsten av osäkerhet om reduktionskostnader är viktig för valet mellan olika miljöpolitiska styrmedel. I sådana fall det vara effektivare att kontrollera den använda kvaniteten direkt snarare än att förlita sig på prisbaserade incitament; detta är speciellt aktuellt då de marginella skadekostnaderna öklar snabbt i takt med ökad användning medan det motsatta gäller för de marginella reduktionskostnaderna. Detta implicerar att i vissa fall är det mer effektivt med kvanitativa regleringar (t.ex. gränsvärden) eller med en kombination av sådana och ekonomiska styrmedel.

Det finns samtidigt också exempel på när ekonomiska styrmedel såsom skatter på användning ensamma kan fungera väl. I vissa fall kan också pantsystem för använda kemikalier vara ett intressant alternativ. De som använder kemikalien betalar då en avgift, som dock returneras då kemikalien återanvänds eller återförs på ett förutbestämt sätt. Denna typ av styrmedel medör en ökad kostnad för illegal dumping samtidigt som den ger ett incitament till återanvändning eller till annan miljövänlig återföring.

I rapporten uppmärksammas också ett antal politiska aspekter på implementeringen av ekonomiska styrmedel. Dessa inkluderar policylegitimitet och acceptans samt de offentlig-ekonomiska frågor som uppstår då skatteintäkter öronmärks for specifika syften. Det politiska motståndet mot miljöskatter kan ofta begränsas genom att återföra intäkterna från skatten till de som betalat den, antingen direkt eller indirekt genom att t.ex. subventionera aktiviteter (t.ex. FoU) i industrin. Även om denna strategi ofta vinner politiskt stöd – och på så sätt ökar acceptansen för den specifika skattepolitiken – kan den också ifrågasättas utifrån såväl ekonomiska som politiska grunder.

Tidigare forskning visar att de beteendeförändringar som följer av en prisökning kan variera beroende på om förändringen är ett resultat av en miljöskattehöjning eller en reguljär marknadseffekt. Skatter kan med andra ord signalera inte bara ekonomisk information utan även moraliska hänsynstaganden rörande individens ansvar att bidra till kollektiva nyttigheter. Då olika styrmedel i kemikaliepolitiken utformas och utvärderas är det centralt att ta hänsyn till hur sådana s.k. signaleffekter kan skapas och t.o.m. stärkas. Ekonomiska och informativa styrmedel ska därför också betraktas som komplement snarare än substitut.

I rapportens tredje kapitel analyserar vi erfarenheterna av ekonomiska styrmedel i kemikaliepolitiken i ett antal europeiska (och inte minst nordiska) länder. Två kategorier av kemikalieanvändningar analyseras: kemikalier som används i jordbruket (gödsel- och bekämpningsmedel) samt farliga kemikalier (t.ex. lösningsmedel). I praktiken beskattas vanligen användningen av olika kemiska produkter. Analysen belyser den trade-off som existerar mellan åstadkommandet av en kostnadseffektiv fördelning av reduktionsåtgärder å den ena sidan samt en dyr kontroll av utsläpp etc. å den andra sidan. Även om de flesta ekonomiska styrmedel som implementerats i praktiken är s.k. second-best lösningar finns det också exempel där man försökt åstadkomma en mer direkt proportionalitet mellan skatten och de uppkomna skadorna. Den tidigare holländska skatten på gödselbalansen samt den norska skatten på bekämpningsmedel utgör exempel på detta.

Analysen visar också ju närmara miljöskadan skatten kan implementeras detso mindre tenderar det politiska motståndet mot skatten att bli. En miljöbeskattning som på ett bra sätt kan approximera miljöskadorna snarare än konsumtionen av den aktuella produkten som sådan kommer att främja såväl kostnadseffektivitet som policyacceptans. Det är samtidigt så att det politiska intresset för sådana mer ”träffsäkra” styrmedel samt på hur de associerade transaktionskostnaderna kan reduceras är ofta svagt. De flesta regeringar uppmärksammar detta problem men de flesta väljer snabbt det enklare alternativet att rikta styrmedlen mot konsumtionen snarare än konsumtionens negativa effekter.

Erfarenheterna av kemikaliebeskattning i de europeiska länderna bekräftar att någon form av öronmärkning av skatteintäkterna kan öka legitimiteten för den förda politiken. Det är t.o.m. så att vissa av de mest signifikanta effekterna av vissa skatter kan itne hänföras till de incitament som skatten ger upphov till utan snarare till de sätt på vilka skatteintäkterna använts. Detta är speciellt tydligt då intäkterna återförts för att stdöja forkning och informationsinsatser kopplade till den berörda industrin. Även om en sådan öronmärkningstrategi har tydliga nackdelar ur ett statsfinansiellt perspektiv bör det noteras att de intäkter som genereras vid beskattning av kemikalier ofta är förhållandevis små samt temporära. Ett annat skäl för att återföra skatteintäkter till relaterad forskning etc. är att de miljöskador som många kemikalier ger upphov till är kumulativa och skatteintäkterna kan användas för t.ex. saneringsprogram. Samtidigt bör det påpekas att öronmärkning får aldrig blir en ursäkt för att införa annars ineffektiva ekonomiska styrmedel.

Rapporten behandlar slutligen tre fallstudier av kemikalieanvändning där införandet av ekonomiska styrmedel kan ha en potentiellt viktig roll för att reducera de associerade negativa effekterna på hälsa och miljö. I rapportens fjärde kapitel diskuteras – för respektive fall – kemikaliens eller produktens egenskaper samt nuvarande regleringar, dess användning ur ett svenskt och nordiskt perspektiv, samt den framtida potentialen för ökad användning av ekonomiska styrmedel. Kapitlet presenterar också översiktliga konsekvensanalyser av de styrmedel som identifieras. De tre fallen representerar inte skarpa förslag på styrmedel som bör prioriteras i framtiden; snarare har fallenm valut utifrån en önskan om att kunna illustrera viktiga utmaningar och möjligheter av ett starkare inslag av ekonomiska styrmedel i kemikaliepolitiken. I alla tre fallen existerar viktiga utmaningar, och på ett övergripande plan handlar utmaningarna om att idenitifera ekonomiska styrmedel som på ett effektivt sätt kan addressera de relevanta miljöeffekterna i respektive fall.

In fallet men nonylfenol (NP) och dess etoxylater (NPE) är det svårt att utifrån nuvarande forskningen identifiera ett ekonomiskt styrmedel som på ett effektivt sätt kan hantera NPE-innehållet iråmga importerade produkter (kläder, textilier, rengöringsmedel etc.). Denna användning utgör idag den viktigaste källan till förekomsten av NPE i Sverige. En skatt på användningen av NP och NPE i produktionen av produkter som innehåller dessa ämnen samt en skatt på NPE utsläpp diskuteras i kapitlet. Dessa styrmedel kan vara intressanta för länder där NPE användningen fortsatt är omfattande i inhemskt producerade produkter samt i större (lättidentifierade anläggningar). I princip skulle en sådan input-skatt kunna läggas på alla sektorer som använder NP eller NPE som råmaterial.

En differentierad beskattning av tvåtaktsoljor representerar ett förhållandevis lovande styrmedel för att hantera bl.a. de utsläpp i vatten som följer av sådan användning. En högre skatt på mindre miljövänliga oljor är dessutom mer effektiva alternativ än såväl en skattesubvention för de miljövänliga alternativen som en s.k. skrotpremie för tvåtaktsmotorer. Skattedifferentieringen är också lätt att införa och eftersom det finns miljövänliga men något dyrare substitut bör politiken inte medföra höga merkostnader (t.ex. för privatbåtsägare). Den stora utmaningen handlar främst om att åstadkomma betydande beteendeförändringar hos konsumentkollektivet, dvs. ökad efterfrågan på de miljövänligare (biotillgängliga) tåtaktsoljorna, givet den låga budgetandel som inköpen representerar för de flesta användare. Det blir därför viktigt att kombinera ekonomiska styrmedel med information om inte minst de negativa miljöeffekterna på den marina miljön.

Den tredje fallstudien berör en skatt på användningen av etandiol-baserad glykol, s.k. etyleknglykol. Som substitut til denna hälsofarliga produkt finns främst s.k. propandil-baserad glykol. Ett problem i detta fall är att de hälsoeffekter som kan uppstå vid förtäring är diffusa och endast svagt kopplade till försäljningsvolymen. Detta skiljer sig från t.ex. fallet med tvåtaksoljor, där motsvarande koppling är starkare. Detta innebär att en skatt på användningen av etylenglykol kommer inte att åstadkomma en kostnadseffektiv reduktion av förtäringsindicenter (bl.a. kopplade till barn). Samtdigt har tidigare informationskampanjer samt rekommendationer rörande förpackningarnas design visat sig vara ineffektiva och en skatt skulle kunna utgöra ett steg i riktning mot en mer permanent lösning på problemen med etylenglykol. Även i detta fall bör dock informativa styrmedel komplettera de ekonomiska.

Sammanfattningsvis visar denna rapport att ett viktigt villkor för införandet av effektiva ekonomiska styrmedel på kemikalieområdet är att så långt som möjligt åstadkomma en stark proportionalitet mellan t.ex. skatten och den aktuella skadan på miljön. Identifieringen av åtgärder som kan reducera de administrativa kostnaderna av sådan förehållandevis träffsäkra styrmedel bör prioriteras. Samtidgt visar också analysen att ekonomiska styrmedel som riktar in sig på användingen av olika kemiska produkter ocsksä kan vara effektiva (t.ex. tvåtaktsoljor), men det är viktigt att undvika skatter som är för generella och därför främst får fiskala snarare än beteendeförändrande effekter. Införandet av ekonomiska styrmedel i kemikaliepolitiken bör också föregås av en detaljerad analys av tillgången till substitutsubstanser och produkter samt de miljöeffekter som kan hänföras till dessa substitut.