Fiscal incentives to advance sound management of chemicals and sustainable chemistry

Review Paper for the Global Chemicals Outlook II

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ABSTRACT

Fiscal incentives are governmental policies that change the relative price of a given activity or input, either encouraging or discouraging its use. They can be created through the removal of existing price distortions that generate perverse incentives for overuse, or through the implementation of new market-based instruments such as taxes, charges, deposit-refund systems, subsidies and tradable permits.

This paper discusses the use of market-based instruments within the broader array of policy instruments for chemical management and analyses factors that facilitate or impede their deployment in different institutional contexts. We also discuss the main challenges in using market-based instruments in the particular context of chemicals, and outline key policy options.

The two main arguments in favour of market-based instruments is that they can be more cost-effective and better at promoting innovation than bans, use restrictions and technology standards. By allowing firms with different substitution costs to reduce their use of harmful chemicals at different levels of intensity and time scales, price-type instruments can incentivise a cost-effective reduction in the use of the targeted chemical. Moreover, by increasing the cost of using a specific chemical, market-based instruments can provide strong incentives to innovate in the search for cheaper alternatives. However, there are many situations where the use of market-based instruments for chemicals management is less appropriate, including when the health or environmental costs from exposure to a hazardous chemical are very high, when effects are location specific and when threshold effects are likely. Market-based instruments should be seen as complementing rather than replacing bans and use restrictions in chemicals management.

In practice, we find a limited but increasing use of market-based instruments for managing hazardous chemicals. Examples in the agricultural sector include taxes on pesticides and inorganic fertilisers. Similarly, taxes have been used to phase out the use of chlorinated solvents in industry. Lately, taxes have been used to encourage substitution of phthalates and brominated flame retardants in products. Refund systems are increasingly used for products containing hazardous chemicals such as batteries, electronic equipment and vehicles. Although the use of market-based instruments is mainly found in high-income countries, a number of low- and middle-income countries are using such instruments for hazardous waste management.

Good knowledge about context-specific factors, such as price elasticities, market structure, availability of substitutes and exposure characteristics for the targeted hazardous chemical, is essential for the design of market-based instruments. Importantly, because market-based instruments can be deployed at different stages of the life cycle of a given chemical, information on the relation between the use of the chemical and its damage function in all those stages is needed. However, in many cases, there is a lack of data, and assessments based on existing data are often surrounded by considerable uncertainties. Hence, there is a need for careful data collection and monitoring and evaluation of the performance of different policy instruments for chemicals management. Flexibility to adjust tax levels after observing market reactions is also necessary.

Promising policy options for scaling up the use of market-based instruments in chemicals management include:

- Expanding the use of risk-based taxation of hazardous chemicals. Learn from the recent implementation of risk-based taxation of pesticides in Denmark, Norway, France and Mexico.
- Evaluating and addressing the effects of subsidies and other policies generating perverse incentives for increased use of hazardous chemicals in agriculture and other sectors.
- Using charges to speed up the phasing out of substances of very high concern.

- Evaluating the use of market-based instruments for groups of chemicals, such as taxes on flame retardants and phthalates.
- Using legal requirements on extended producer responsibility, environmental liability and access to information to incentivise sound chemicals management in line with the polluter pays principle.

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1 INTRODUCTION

This report takes stock of the extent of and lessons-learned from using fiscal incentives for chemicals management. It discusses the effectiveness, benefits and challenges of market-based instruments within the broader array of possible policy instruments in the particular context of chemicals management.

Fiscal incentives are governmental policies that change the relative price of a given activity or input, either encouraging or discouraging its use. They can be created through the removal of existing price distortions that generate perverse incentives for overuse, or through the implementation of new market-based instruments such as taxes, charges, deposit-refund systems, subsidies and tradable permits.

Market-based instruments form part of a larger menu of policy instruments that policy makers can use in order to address various types of environmental problems (Table 1).

Table 1. A menu of policy instruments

USING MARKETS	CREATING MARKETS	REGULATION	ENGAGING THE PUBLIC
Subsidy reduction Environmental taxes and charges User charges Deposit-refund systems Targeted subsidies	Property rights and decentralisation Tradable permits and rights International offset systems	Standards Bans Permits and quotas Zoning Liability	Public participation Information disclosure

Source: Sterner and Coria (2012)

In chemicals management, use restrictions such as standards, bans and permits in combination with information disclosure have traditionally been the most common policy instruments. The two main arguments in favour of market-based instruments is that they can be more cost-effective and better at promoting innovation than bans, use restrictions and technology standards (Sterner and Coria, 2012; Stavins, 2001). These command and control policies typically allow for very little flexibility in the means of achieving specific targets. As a result, all firms need to meet the same target, irrespective of how costly the change is. However, the cost of complying with a ban or use restriction often differs between companies, for example due to differences in production processes and sunk costs from technology investments.

By making the use of a certain chemical more costly, market-based instruments – in contrast to standards and permits – provide a continuous incentive for substitution with less hazardous alternatives. Companies have an incentive to substitute the targeted chemical as long as their marginal cost of substitution is lower than the cost of using the targeted chemical. By allowing firms with different substitution costs to reduce use at different time scales, market-based instruments can incentivise a cost-effective reduction in the use of the targeted chemical. Moreover, by increasing the cost of using a specific chemical, the introduction of taxes and charges can also spur innovation and the search for new alternatives to the relatively more expensive input. Ideally, market-based instruments should target groups of chemicals in order to avoid undesirable substitution where companies shift to using a close substitute with similar properties as the targeted chemical.

While market-based instruments have some merits, there are many situations where they are less appropriate, including when the health and environmental costs from exposure to a hazardous chemical is very high, where effects are location-specific and where threshold effects, i.e. an abrupt spike in the damage function after a given threshold, are likely. In such situations, bans and use restrictions are more appropriate (Weitzman, 1974; Baumol and Oates, 1988). In practice, a large number of context-specific factors – such as information constraints, administrative costs, distributional effects and political economy pressures – determine which policy instruments are most effective and feasible to implement, and thus, policy instrument design needs to be context and problem specific.

1.1 Objectives and scope

The purpose of this review paper is to capture current knowledge concerning the extent of using market-based instruments to advance the sound management of chemicals and sustainable chemistry, as well as the lessons learned. It also, to the extent possible, assesses the effectiveness of market-based policy instruments within the broader array of possible policy measures.

Questions addressed include:

- To what extent have market-based instruments been used to stimulate sound management of chemicals in low, middle, and high-income countries?
- Which specific market-based instruments have been introduced, and for what specific purpose? Were these measures stand-alone or coupled with other policy measures?
- What evidence exists that the identified market-based instruments achieved (or did not achieve) their intended objectives? What are important lessons learned?
- What is the promise of market-based instruments in achieving the sound management of chemicals in the long term, and in what contexts are they likely to be effective?
- What are the lessons learned and insights relevant for consideration at the global level? Are some aspects of particular relevance for low and middle-income countries?

1.2 Methodology

While there are several reviews of the use of market-based instruments for environmental management in general (see e.g. Harrington et al., 2004; Stavins, 2001; European Environmental Agency, 2006; Sterner and Coria, 2012; OECD, 2001), there are relatively few studies reviewing the use of market-based instruments for chemicals management (see e.g. Söderholm, 2009; Swedish Chemicals Agency, 2011; Swedish Chemicals Agency, 2013; Swedish Chemicals Agency, 2007; Macauley et al., 1992).

The present study reviews existing evidence on the use of market-based instruments for chemicals management, based on the scientific literature as well as grey literature and the OECD database on policy instruments for the environment[2]. A review of the scientific literature was conducted as follows. First, a literature search was performed in the Econlit database using a set of 32 keyword combinations (see Appendix 1). Then, 324 hits in the time period 2007–2017 were evaluated for their relevance using pre-defined quality criteria. As we found a relatively limited number of relevant articles in Econlit, additional literature was identified by including literature citing the original hits and searching additional scientific databases. Grey literature was retrieved through web searches and personal contacts of the authors.

The search in the OECD database was conducted using a selection of relevant instruments from the environmental domains 'land contamination', 'ozone layer protection' and 'waste management' for 107 countries (OECD countries and additional countries). The results are summarised in Appendix 2.

Despite our efforts, this review should not be seen as an exhaustive review of all cases of market-based instruments used for chemicals management. It rather illustrates some of the applications as

[2] http://www2.oecd.org/ecoinst/queries/

well as considerations policy makers face in policy instrument design. It should also be noted that the review of the effectiveness of market-based instruments in chemicals management is hampered by an unfortunate lack of formal evaluations of effectiveness in the existing literature. The review focuses on market-based instruments targeting chemicals manufactured for commercial use and does not cover instruments targeting chemicals created as by-products during production or use, such as carbon dioxide and nitrogen dioxide. Moreover, it does not cover petrochemicals. An additional caveat is that the review focuses on market-based instruments that generate incentives by affecting the relative price of chemicals and chemical products. Consequently, other instruments that may create incentives for behavioural change, such as nudges, information campaigns or labelling systems, are not covered.

2 INSTRUMENT TYPES AND KEY PRINCIPLES OF POLICY INSTRUMENT DESIGN

2.1 Different types of market-based instruments for chemicals management

A wide range of market-based instruments are used in chemicals management (Table 2). These instruments stimulate behavioural change by providing price signals to relevant agents such as chemicals producers and manufacturers, downstream users, consumers and waste management agents.

Table 2. Typology of market-based instruments and examples of applications in chemicals management

Policy instrument	Description	Example of application
Tax	By increasing the price of using a chemical, a tax incentivises decreased use. Taxes are levied by the state, with proceeds going to the general budget. The level should reflect the damages caused by the production, use and disposal of the chemical, which in the absence of the tax would not be reflected in the market price of the input or final product.	Pesticides, inorganic fertilisers, chlorinated solvents, batteries
Charge / Fee	Similar to a tax but revenues are typically earmarked. The level of a fee should reflect the cost of providing a specific service – such as processing hazardous waste.	Hazardous waste, pesticide or chemical containers, tyres, batteries
Subsidy	A subsidy is the mirror image of a tax. It can provide incentives to increase the use of alternative chemicals that are less hazardous. In particular, authorities may want to subsidise learning and technology development.	Subsidies for organic farming, lead paint removal.
Subsidy removal	In many cases, subsidies are introduced to deal with distributional concerns, yet may result in unsound practices from a health or environmental perspective. Hence, subsidy removal is considered a policy instrument in its own right.	Removal of subsidies for the use of chemical fertilisers or pesticides.
Deposit-Refund	A surcharge is paid when purchasing potentially	Pesticide or chemical containers, batteries, tyres

	polluting products. A refund is received when returning the product to an approved centre for recycling or disposal.	
Tradable permits	An overall level of 'allowable' pollution is established and allocated among firms in the form of permits. These permits can be traded on a market at market prices	Lead in petrol (trade among refineries), ozone-depleting substances (trade among producers and importers)

Source: Authors, based on Stavins (2001), Sterner and Coria (2012) and the OECD database on Policy Instruments for the Environment (OECD, n.d.).

2.2 Regulating prices or quantities?

Policy instrument choice in the area of chemicals is surrounded by uncertainties in terms of both the damage costs from chemical production and consumption and the abatement or substitution costs. In a seminal paper, Weitzman (1974) shows that the assumptions made under uncertainty about the marginal damage costs and marginal abatement costs are fundamental to the choice of policy instrument. When there are reasons to assume that the marginal damage cost rises sharply with use, for example due to threshold effects, and the marginal cost of abatement is comparatively low, then a quantitative restriction is generally more efficient than a price-type instrument. When there are reasons to assume that the marginal damage cost is not rising sharply with use but the marginal abatement cost is high, a tax or other price-type instrument is usually more efficient than a quantitative instrument.

Chemicals policy has traditionally focused on reducing the risk from chemicals where the damage costs are known to be high, such as lead and PCB. Hence, quantitative restrictions through bans and permits have been the most common policy instruments in chemicals management. There is growing concern about the cumulative effect or combination effects from exposure to multiple chemicals, as the production of synthetic chemicals has grown very rapidly in recent decades (Bernhardt et al., 2017). A focus on addressing not only substances of very high concern but also broader groups of chemicals could motivate increased use of market-based instruments in chemicals management (Sadler, 2000).

In practice, the relative effectiveness of a ban or a tax also depends on administrative costs related to monitoring and enforcement and its implementation feasibility (see Section 2.4 below).

2.3 How high should a chemical tax or fee be?

According to economic theory, a tax should be designed so that the environmental and health costs caused by the production, use and disposal of the chemical are reflected in the price of the chemical or the products containing the chemical. In line with the polluter-pays principle, the tax thereby creates an incentive for economic actors to internalise the full cost into their decisions. The level of a fee should reflect the cost of providing a specific service, such as processing of hazardous waste.

In practice, several factors make it difficult to accurately estimate the health and environmental costs of the production, use and disposal of chemicals. First, understanding the degree to which chemicals are toxic, persist, bioaccumulate and have endocrine-disrupting properties is crucial for estimating the damage caused by their use. However, data on the hazard and exposure of specific chemicals are not always available and data on combination effects when chemicals interact in specific environments are even more scant. Second, the damage costs also differ with different uses. In some cases, the main damage stems from point sources, such as industrial plants for production or recycling. In other cases, the damage costs arise from diffuse sources when millions of consumers use products containing chemicals. There may also be large spatial variations in damage costs from for example the use of fertilisers and pesticides (Söderholm, 2009).

An alternative to trying to estimate what tax level would internalise the external cost from the production, use and disposal of a chemical is to design a policy instrument that achieves an objective at the lowest possible cost to society (for example phasing out the use of lead in paint or the use of chlorinated solvents in metal degreasing operations).

2.4 Where in the lifecycle of chemicals should market-based instruments be applied?

Health and environmental impacts may arise from chemical exposure during different stages of the life cycle of a chemical – i.e. from its actual design, from its use as input into the production of goods and services, as part of a finished good or service and as a result of its final disposal (Figure 1). Accordingly, market-based instruments can be designed to encourage or discourage the use of a given chemical at all or any of those stages.

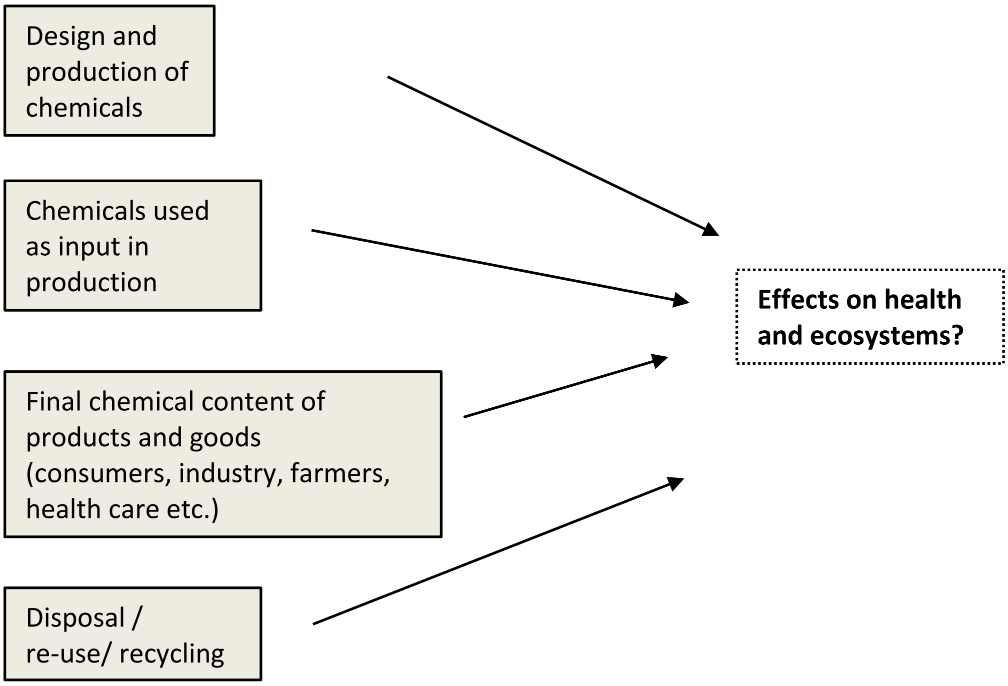

Figure 1. Chemical exposure during different stages of the life cycle of chemicals

From an economic efficiency point-of-view, it is desirable to target policy instruments to specific environmental or health damages as closely as possible. Formally, an optimal (green) tax should be set such that the marginal damage is equal to the marginal benefit of using the chemical in question. However, as explained in Figure 1, the damage to nature or human health can happen at different stages in the life cycle of the chemical. Importantly, because market-based instruments can be deployed at different stages of the life cycle of a given chemical (during the design phase, as input in production of other goods and services, as part of a final consumption good or as after-use regulation), information on the relation between the given chemical and its damage function in all those stages is needed. The risk of unduly restricting the use of chemicals that do not cause negative environment and health effects can thereby be reduced.

However, balancing the benefits of a targeted approach against its transaction costs is a key dilemma in policy instrument design (Vatn, 1998). For example, it might be very hard or impossible to acquire information on the production of the chemical itself, so typically social planners move up on the life cycle, for example taxing the use of the chemical as input in the production of other goods and services. Still, in some cases information on the production of those other goods and services is hidden or private, so the regulator is forced to deploy taxes and similar instruments on the final disposal of the chemical. This is for example the case for non-point pollution, where the regulator can only observe final pollution levels in the aggregate.

The technical complexity of the design of policy instruments varies across the different stages of a chemical's lifecycle. Typically, the number of economic actors increases further down the chemical life cycle. The damage costs from non-point source pollution from millions of users of chemical products can be difficult to assess as they may vary with how, where and by whom the products are used. It can therefore be complicated to develop differentiated taxes based on specific damage estimates for different uses.

Alternatively, an input tax can lower the overall use of a specific chemical and also can be easier to administrate as the number and diversity of producers is far more limited than at later stages. However, an input tax may risk unduly restricting less harmful applications of the targeted chemical (Macauley et al., 1992).

While taxing actors early in the chemical life cycle can in some cases be reasonably cost-effective second-best measures (Söderholm, 2009), regulatory design needs to carefully consider the technical and political complications associated with the distribution of regulatory costs and benefits resulting from targeting actors at different stages of the chemical life cycle (Coria, 2018)

2.5 Taxes that reduce environmental and health effects or generate fiscal revenue?

The effects of a tax on the use of a chemical – and the associated health and environmental effects – depend on the marginal abatement (or substitution) cost, i.e. on how difficult economic actors find it to reduce the demand of the targeted chemical(s). When taxes on chemicals are used, companies are assumed to compare the marginal costs of reducing chemical use with the tax rate and reduce their use until their marginal cost of reduction equals the tax rate.

Knowledge about the marginal abatement or substitution cost is also crucial for estimating how much fiscal revenue will be generated from a tax. If demand for the polluting good (e.g. a good involving substances of very high concern during production or use) is inelastic to price increases, as in Figure 2A, then a tax will have a limited effect on the demand for the good. The reason for this is that abatement or substitution is costly or difficult for other reasons. For example, some studies have found that the demand for certain pesticides is inelastic to price increases, at least in the short run (Finger et al., 2017). This does not necessarily mean that a tax is not an adequate policy instrument. Because the demand curve is steep, other instruments would also encounter difficulties in achieving significant substitution. In this situation, a tax can provide a continuous incentive for innovation while generating fiscal revenue. The revenue from the tax can also in principle be used to subsidise the development and use of alternatives to the targeted chemicals.

If, on the other hand, demand is elastic, as in Figure 2B, then a tax will lead to a big reduction in the use of the targeted chemical and therefore also to a rapidly diminishing tax base. Since substitution in this case is easy, tax revenues are likely to be small and decreasing.

Figure 2C depicts a third situation where the demand curve is a combination of A and B and has a convex shape. In such a case, a tax will reduce the demand among economic actors with low substitution costs, and, as demand becomes more inelastic, will continue to generate fiscal revenue from actors with higher substitution costs. Petrol taxes fall into this category (Sterner and Coria, 2012).

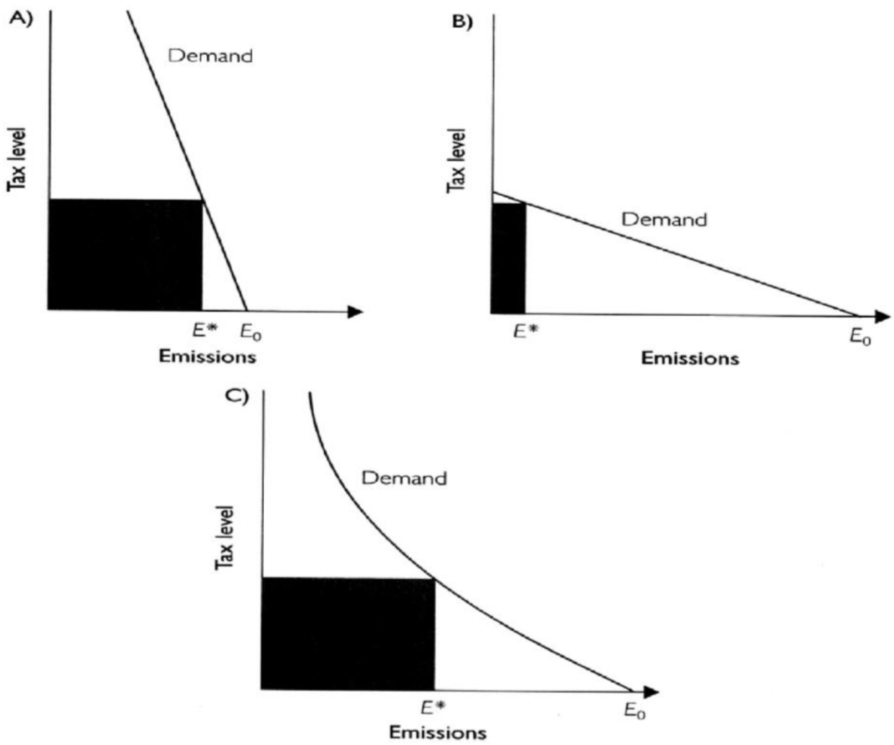

Figure 2. Environmental taxes and demand elasticity
Source: Sterner and Coria (2012)

Hence, there may be a trade-off between using a tax as an instrument to phase out the use of specific chemicals and using it as an instrument to generate fiscal revenue. In order to predict the effect of a chemical tax on the demand of a chemical, regulators need information on companies' cost of reducing or substituting the targeted chemical. However, this information is normally in the hands of the companies and is not easily accessible by the regulator. Studying the price elasticity of demand using market data can be one way for regulators to obtain the information needed. If such data is not accessible, regulators may have to resort to observing market reactions to a specific tax level and then adjust the tax level in order to reach the objectives set.

2.6 Fiscal incentives and innovation?

By increasing the cost of using specific chemicals, taxes and charges can also spur innovation and the search for new alternatives to the relatively more expensive input. Taxation changes relative prices and the rate of return of investments in alternative technology. While taxation may lead to an internalisation of environmental and health costs and induce some innovation, there is an additional reason for governments to actively encourage innovation. Due to imperfect information and knowledge spill-overs, innovators usually do not reap the full benefits of their efforts, which lowers the incentives to invest in innovation (Jaffe et al., 2005).

By providing tax credits for expenses related to research and development or by providing favourable treatment of capital or labour expenses, governments can generate additional incentives for research and innovation. These measures normally form part of broader policy packages encouraging innovation, such as investments in research and education or more targeted innovation programmes (OECD, 2010).

3 EXPERIENCES FROM USING MARKET-BASED INSTRUMENTS IN PRACTICE

3.1 Limited but increasing use of fiscal incentives for managing hazardous chemicals

Based on a search for market-based instruments for chemicals management in the OECD Policy Instruments for the Environment database[3], we find that the most frequent use of these instruments is for hazardous waste management (Appendix 2). Taxes, fees and charges and to some extent also deposit-refund systems are frequently applied for products such as tyres, batteries, accumulators, electrical and electronic products, vehicles and other aspects of hazardous waste management. A few countries also use charges or deposit-refund systems for containers made for pesticides and other chemicals. The use of market-based instruments for other aspects of chemicals management is less frequent. Taxes and charges are imposed on pesticides, fertilisers, ozone-depleting substances and chlorinated solvents. Tradable permit systems are found for ozone-depleting substances and chlorinated solvents. See Section 3.3 regarding the use of subsidies in relation to chemicals management.

While it is likely that some uses of market-based instruments for chemicals management are not included in the table in Appendix 2 and that some of the instruments included are no longer active, the information indicates that compared with other policy areas, the use of market-based instruments for chemicals management is relatively limited. In combination with our literature review, we find that market-based instruments are mainly used in high-income countries. However, a number of low- and middle-income countries have also begun to use market-based instruments, particularly in relation to hazardous waste management and plastic bags.

The cases presented below illustrate some of the applications of market-based instruments as well as some common considerations policy makers face when designing policy instruments.

3.2 Bans vs taxes in practice: the case of regulating chlorinated solvents

In practice, the effectiveness of policy instruments depends on a number of implementation characteristics. We illustrate this by discussing the regulation of chlorinated solvents where different countries have used different policy instruments to control emissions and phase out the use of the substances. This allows for a comparison of the pros and cons and the relative effectiveness of different policy instruments.

In 1991, the Swedish government announced that a total ban on the production and use of trichloroethylene (TCE) would come into force in 1996. However, after strong protests from many industries using TCE, the government allowed companies to continue using the substance also after 1996 if they could prove that replacing TCE was very expensive, that they had actively (but unsuccessfully) searched for an alternative to TCE and that continued use of TCE did not lead to unacceptable exposure. Using data from 65 companies on the marginal costs of reducing the use of TCE, Slunge and Sterner (2001) compared the effect of the Swedish TCE ban with a hypothetical scenario where instead a tax of 50 SEK per kg[4] (about 5 EUR per kg) had been introduced. Figure 3 shows the marginal cost of reducing TCE for each of the 65 companies. The marginal cost was very low for most companies, but for a number of companies it was estimated to be over 100 SEK per kg. One year after the ban had been imposed, the 65 surveyed companies had reduced their use of TCE by 68 %. Assuming that companies with a marginal cost of replacing TCE lower than the presumed tax

[3] The database covers economic policies in both OECD and other countries (107 in total), but the data is more complete for the OECD countries.
[4] This tax level was proposed in a Swedish government investigation on market-based instruments for environmental management in 1990 (Miljöavgiftsutredningen, 1990).

rate of 50 SEK per kg would have replaced TCE in the hypothetical scenario, such a tax would have led to a reduction in TCE use by 88 %.

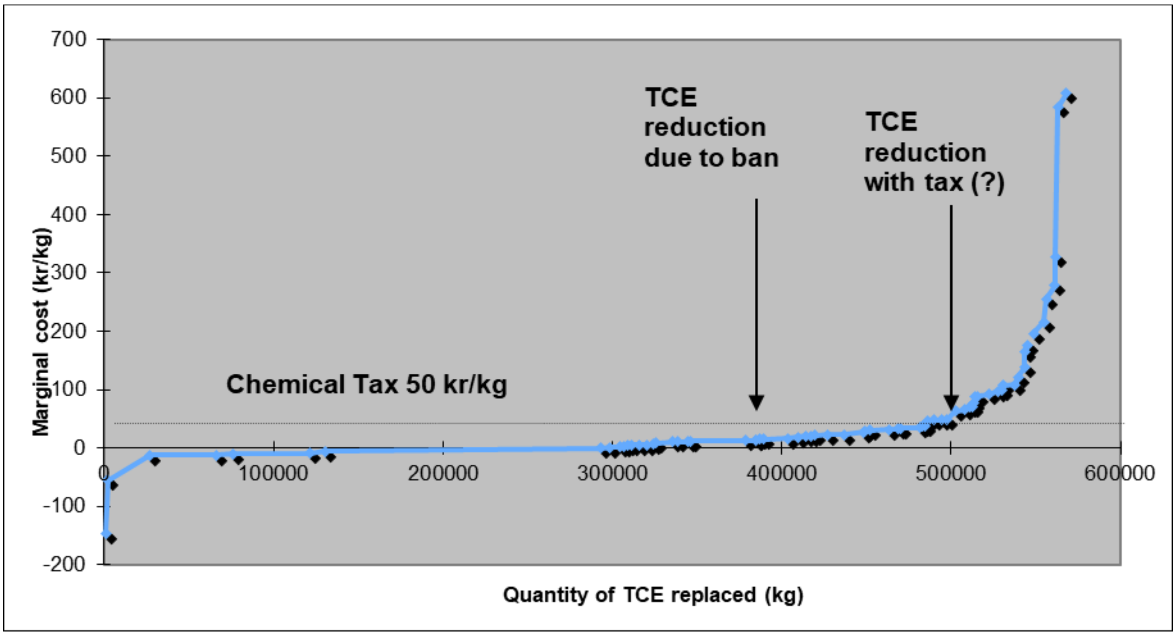

Figure 3. A tax versus a ban for regulating the use of trichloroethylene (TCE)

Notes: Reported marginal costs of reducing the use of TCE (or replacing the use altogether) in metal degreasing for 65 industrial companies in Sweden. Assumptions: 15-year equipment life, 4 % real interest rate.
Source: Slunge and Sterner (2001)

In year 2000, Norway introduced a TCE tax of 50 NOK per kilo (roughly the same as 50 SEK or 5 EUR)[5]. Already in the early 1990s, Germany introduced best-available-technology requirements, implying tough emission standards for equipment using chlorinated solvents for metal degreasing. Figure 3 plots the phase-out rates in the three countries compared with the rest of Europe. The use of TCE has decreased more rapidly in Sweden, Norway and Germany – the countries that actively tried to reduce the use and exposure of TCE – compared with the less-stricter policies of the majority of other European countries, where emissions have declined only very gradually.

[5] In Norway there was also a 50 NOK per kg tax on perchloroethylene (PER), which is a close substitute to TCE. In Sweden, PER was neither banned nor taxed.

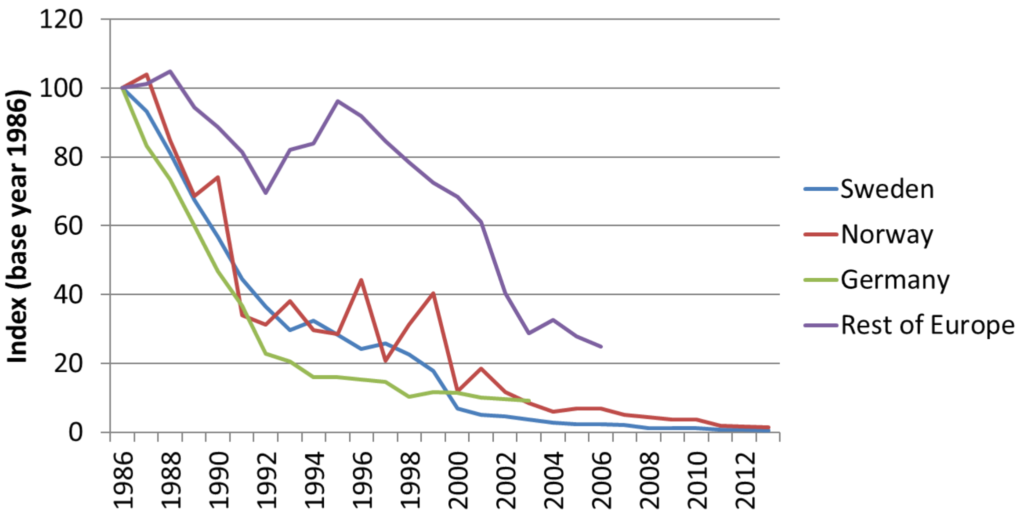

Figure 4. Rates of reduction of trichloroethylene (TCE) in Europe

Source: Author, based on Sterner (2004) and the SPIN database on the use of chemical substances in the Nordic countries. Data on sold quantities of TCE in Europe is not publicly available after 2006.

Comparing the use in Sweden and Norway, we find that a ban is not necessarily more effective than market-based instruments in quickly reducing use and emissions. The difficulties that Swedish authorities encountered in enforcing the ban led to a continued use of TCE in Sweden many years after the ban entered into force in 1996. The Norwegian tax is likely to have been the instrument that was easiest to administer. The tough German regulations of emissions may have been the most effective in addressing exposure to TCE in the short run (Slunge and Sterner, 2001).

3.3 Market-based instruments during different stages of the lifecycle of chemicals

Market-based instruments have been applied at different stages of the lifecycle of chemicals and products. For example, US producers and manufacturers of chemicals were involved in tradable permit schemes intended to incentivise refineries to phase out lead in petrol 1982–1987 (see Box 1) and in the phasing out of ozone depleting substances in the early 1990s (Harrington et al., 2004).

The Norwegian tax on trichloroethylene and perchloroethylene introduced in 2000 is an example where the inputs to the metal manufacturing industry and dry-cleaning facilities have been targeted (Slunge and Sterner, 2001). The taxes on phthalates, PVC and flame-retardants introduced in Denmark and Sweden are example of taxes targeting consumer products but where it is the agent importing or selling the product who pays the tax (see Box 2). Taxes on plastic bags used in Ireland, South Africa, the United Kingdom, the United States and several other countries are examples of taxes paid by the consumers (Box 3).

Box 1. The use of tradable permits and taxes to phase out lead in petrol

Lead in petrol has been phased out in most countries in the world. The initial driving force was the need to protect the catalytic converters introduced in newer cars, which were destroyed by only small amounts of lead (Sterner and Coria, 2012). However, with growing awareness of the health effects of lead exposure, the main driving force for the phasing-out of lead changed. Policy instruments used to support the phase-out varied. The United States, the first country in the world to begin the phasing out of lead, did so by using a tradable permit system (Newell and Rogers, 2003), while Sweden and other European countries used a system with a higher tax on leaded than unleaded petrol (Hammar et al., 2004). The tradable permit system offered a flexible system for refineries to invest in the required technology for producing unleaded petrol, thereby reducing the risk that the legislation would be delayed by being challenged in court, a common procedure in the US (Sterner and Coria, 2012). The system reduced the phase-out time of lead in petrol by at least five years, from 1979 to 1988, and is estimated to have been highly cost effective, saving hundreds of millions of dollar compared with the use of a command-and-control policy (Newell and Rogers, 2003). In Sweden, leaded petrol was phased out from 1987 to 1994 by gradually decreasing the lead content allowed and gradually increasing the tax difference between leaded and unleaded petrol (Hammar et al., 2004). Initially, the tax difference was not enough to compensate for the investment cost and difference in production costs of leaded and unleaded petrol, but along with a further increased tax on leaded petrol and the compulsory introduction of catalytic converters in new cars from 1988, lead in petrol was eventually phased out.

Box 2. Chemical taxes on consumer products in Denmark and Sweden

Following a growing concern about the risk of cumulative exposure to hazardous chemicals in consumer goods, Denmark introduced a tax on products containing phthalates and PVC in 2000. The tax rate was approximately 0.3 euro cents per kilogram of PVC and 0.9 euro cents per kilogram of phthalates, with some variation depending on the product. The effects of the tax are uncertain. One early assessment points to a 15 % decrease in the use of phthalates from 2002 to 2004 (Government of Denmark, 2006). It cannot be ruled out that technological advancements would have reduced the use of phthalates and PVC anyway, especially since the tax rate is considered to have been relatively low (Government of Sweden, 2015). The effect of the tax levelled off as the tax level has not changed or been inflation-adjusted since it was introduced and as European regulations of phthalates have been introduced. The Danish government has decided to abolish the tax bon 1 January 2019. The Danish tax is no longer considered to have any significant effect on health or the environment since many of the phthalates are now regulated within the European Union. Moreover, the tax has been criticised for not differentiating between phthalates, which gives the industry no incentives to change to less hazardous phthalates (Stringer, 2017a).

Sweden introduced a tax on certain chemicals in electrical and electronic products in 2017. The aim was to incentivise substitution of hazardous flame retardants to less hazardous alternatives. Producers and importers of these products pay an excise duty of around 0.8 EUR per kilogram for kitchen appliances and 12 EUR per kilogram for other electronic products. There is a maximum charge of 32 EUR per item. If producers and importers can prove that the electronic products do not contain additive compounds of bromine, chlorine or phosphorus, they can get a 50 % tax deduction. If they can also show that the products do not contain reactive added bromine or chlorine compounds, they are entitled to a 75 % tax deduction. Since the tax recently came into force, it has not yet been evaluated. However, it has been criticised by industry for not being based on a comprehensive risk assessment and for being administratively burdensome. In addition, the tax is also imposed on phosphorus flame retardants, i.e. the alternative to the more hazardous bromine and chlorine compounds, which has raised concern that the phasing out of bromine and chlorine compounds may be delayed (Stringer, 2017b). Moreover, as retailers not registered in Sweden are not subject to the tax, Swedish consumers may switch to buying their electronic products from companies abroad (Rendahl, 2017). The tax also has a fiscal incentive, yet due to its complex design a considerable part of the revenues may be lost in administrative costs.

Box 3. Different effects of charges on plastic bags in South Africa, Ireland and UK

Charges have been used in several countries as an instrument to reduce the demand for plastic bags. The primary motive has been to reduce plastic littering. The effects of plastic bag charges are mixed. A charge of 15 euro cent per shopping bag introduced in Ireland in 2002 led to a 90 % reduction in use and an associated gain in the form of reduced littering (Convery et al., 2007). Similarly, the charge of around 0.05 EUR (0.46 rand) on plastic carrier bags introduced in South Africa in 2003 led to an estimated 90 % reduction in demand. However, after pressure from plastic-bag manufacturers, the charge was lowered after only three months, and the demand for plastic bags then increased again (Dikgang et al., 2012). In addition to the effect of the Irish charge on demand, the extensive stakeholder consultations and information campaigns conducted in the country contributed to the effectiveness of the policy instrument. In South Africa, the revised charge was too low to affect demand.

Since October 2015, large retailers in England have been required by law to charge 0.06 EUR (0.05 GBP) for single-use plastic carrier bags. The seven main retailers issued around

83 per cent fewer bags (over 6 billion bags fewer) in 2016-2017 compared to the calendar year 2014. This is equivalent to each person in the population using around 25 bags in 2016-2017, compared to around 140 bags a year before the charge (UK DEFRA 2018).

3.4 Taxing inputs or emissions – experiences from taxation of agrochemicals

Pesticides[6] and chemical fertilisers play a critical role in increasing agricultural productivity. However, their use can have severe negative health and environmental effects. With few exceptions, agrochemical pollution originates from non-point sources, from multiple agents, that are very difficult (or very expensive) to monitor. Its consequences show high variability in time and space. Hence, controlling agricultural non-point source pollution poses many challenges to the design of policy instruments (Shortle et al., 2012).

The most frequently used policy design features are: i) proxy of emissions, like soil erosion estimations as a proxy for fertiliser exports into water bodies, ii) specific indicators of farm performance, such as polluting inputs (fertilisers or pesticides), or other inputs and practices correlated with pollution flow, and iii) ambient concentrations, i.e. aggregates measured at a relevant scale (e.g. hydrographic basin).

Instruments based on a proxy of emissions can be expected to be the most efficient, given that they are able to target only those farmers whose marginal damage is higher. Moreover, for taxes and subsidies based on the use of certain inputs or management practices to be efficient, they need to reflect the impacts on health and ecosystems that these inputs and practices cause. Ambient concentration-based instruments are the most feasible and simple to monitor, but they cannot distinguish between those farmers and locations that generate pollution and other pollution sources.

Table 3 summarises the use of taxes in relation to pesticides and chemical fertilisers. Notably, all applications are in high-income countries with the exception of the pesticide tax in Mexico introduced in 2013. Fertiliser taxes have in most cases been based on the weight of nitrogen and phosphorus in the products. Sweden has also taxed cadmium based on amounts in gram in fertiliser. However, in the Netherlands, the fertiliser tax targeted expected emissions with a tax on soil phosphorous and nitrogen surplus, based on thresholds and a block rate structure. Austria and Finland removed their fertiliser taxes arguing that they had a negative impact on agricultural sector competitiveness after they joined the EU in 1994. Sweden kept these types of taxes until 2010 when they were abolished to improve the competitiveness of the Swedish agricultural sector. Since the price elasticity of the demand for fertilisers is generally low, high taxes are needed to induce a substantial reduction in demand. Fertiliser taxes have also been criticised for their low effectiveness in actually reducing water pollution. Regulatory limits on nitrogen fertilisers may be more effective (OECD, 2012).

Also pesticide taxes have commonly been based on per unit taxes for all pesticides, leading to the same tax rate on products differing in relative toxicity. In some cases, taxes have been charged as a percentage of the price of the pesticide. However, this has led to a situation where major users of pesticides pay a lower tax per unit of pesticide than smaller users (OECD, 2012).

[6] Such as herbicides, insecticides, fungicides and rodenticides.

Table 3. Summary of taxes on fertilisers and pesticides

<table>
<tr><th colspan="6">Inputs / Practices</th><th>Emissions proxy</th></tr>
<tr><td rowspan="6">Fertilisers</td><td></td><td>N</td><td>P</td><td>Cd</td><td>K</td><td rowspan="6">Tax by N and P surplus

The Netherlands (HOL):
• N surplus> 40 kg / ha 2.30 EUR
• N surplus 0 – 40 kg / ha 1.15 EUR
• P surplus> 10 kg / ha 9 EUR/ha</td></tr>
<tr><td>DEN</td><td>0.67 EUR / kg. (not agriculture)</td><td>0.1679 EUR/g P added in animal food surplus</td><td></td><td></td></tr>
<tr><td>FLA</td><td>0.00111 EUR /kg</td><td>0.00111 EUR / kg</td><td></td><td></td></tr>
<tr><td>SUE</td><td>1.8 SEK / kg.</td><td></td><td>30 SEK / g Cd over the 5 g. of Cd/tonne P</td><td></td></tr>
<tr><td>AUS</td><td>0.47 EUR / kg.</td><td>0.25 EUR / Kg.</td><td></td><td>0.13 EUR / kg</td></tr>
<tr><td>FIN</td><td>0.44 EUR / kg.</td><td>0.44 EUR / kg</td><td></td><td></td></tr>
<tr><td rowspan="10">Pesticides</td><td>DEN</td><td colspan="4">3% al 35% depending on type</td><td rowspan="10">Specific tax per site computed by the potential damage

Norway (NOR):
• Complex formula</td></tr>
<tr><td>FIN</td><td colspan="4">840 EUR + 3.5% final price (without VAT)</td></tr>
<tr><td>SWE</td><td colspan="4">30 – 34 SEK/ kg of the active substance</td></tr>
<tr><td>UK</td><td colspan="4">Registry: 5000 EUR
General rate: 5719 EUR</td></tr>
<tr><td>FRA</td><td colspan="4">381 to 1067 EUR / tonne depending on the category</td></tr>
<tr><td>ITA</td><td colspan="4">Over the final price (0.5% domestic and 1% imported)</td></tr>
<tr><td>BEL</td><td colspan="4">0.395 EUR / kg of the active substance</td></tr>
<tr><td>BC</td><td colspan="4">1.20 CAD per litre of pesticide</td></tr>
<tr><td>WAS</td><td colspan="4">0.7% of wholesale price</td></tr>
<tr><td>MEX</td><td colspan="4">3% to 4.5% depending on toxicity level</td></tr>
</table>

Notes: SWE = Sweden; DEN = Denmark; BEL = Belgium; AUS = Austria; FIN = Finland; NOR = Norway; HOL = The Netherlands; WAS = Washington, USA; FRA = France; ITA = Italy; FLA = Flanders, Belgium; BC = British Columbia, Canada; MEX = Mexico; P = Phosphorous (P_2O_5); N = Nitrogen; Cd = Cadmium; K = K_2O.
***Bold** indicates the currently active ones*
Source: Own elaboration based on OECD (2012). REDUCTIONS (2014). DEC (2016)

Several countries, including Denmark, Norway, France and Mexico, have recently begun to use risk-differentiated taxation of pesticides (Plant Protection Products) to incentivise farmers to use less hazardous products. In Norway, a new taxation scheme for pesticides was introduced in 1999. Instead of the former ad valorem tax, pesticides were separated into different risk categories with higher taxation levels for the higher risk categories. The tax is estimated using two rates: a base rate and an additional rate. For the base rate, pesticides are classified using seven categories and the tax baseline was initially set at 25 NOK/hectare (around 3 EUR), which is then multiplied by a factor associated with the category of each pesticide (from 0.5 to 150). The additional rate is estimated according to the maximum recommended dose for the main crop in the field.

Figure 5 indicates that a shift towards using more of the relatively less hazardous pesticides has taken place after Norway introduced the differentiated pesticide taxation (Kjäll, 2012). However, the tax has only led to a slight reduction in overall pesticide use. Later assessments have shown reductions in both violations of maximum allowed water nutrient levels and the number of detected residues, but it is not clear whether the tax caused this reduction (Böcker & Finger, 2016).

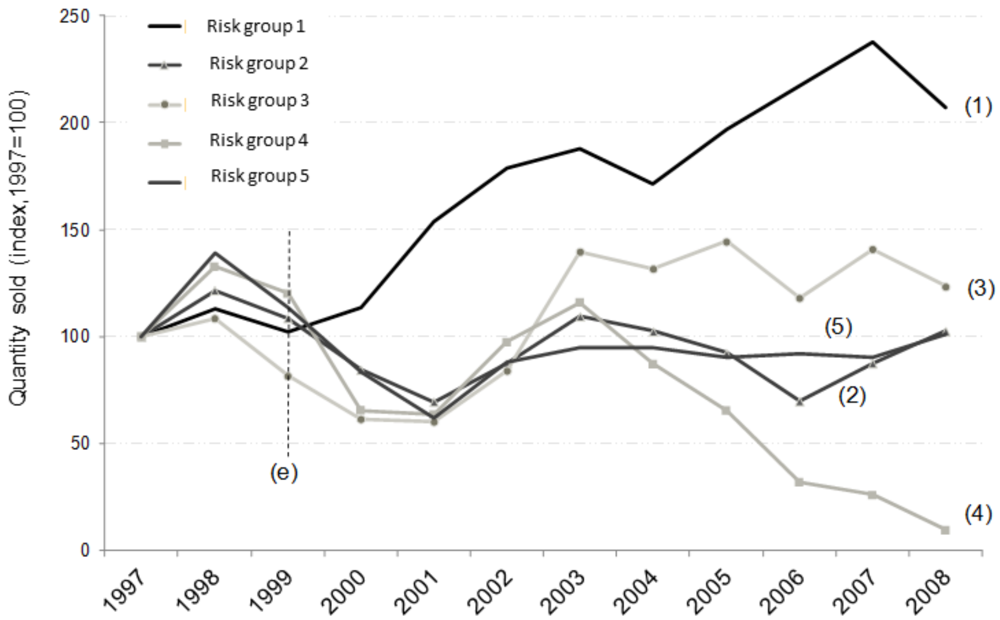

Figure 5. Effects of differentiated taxation on sold quantities of pesticides in Norway

Notes: A new taxation scheme for pesticides (Plant Protection Products) was introduced in 1999 (e). Pesticides were divided into 7 categories based on their health and environmental risks, with higher taxation of products in higher risk categories. Classes 6 and 7 concern non-professional use and are not shown in the figure. The figure indicates that a shift towards using more of the relatively less hazardous pesticides (Class 1) has taken place. The tax rate was adjusted in 2000 and 2005.
Source: Kjäll (2012)

A difficulty with classifying pesticides into distinct risk categories is that pesticides with similar levels of environmental and health risks may differ substantially in the tax rate applied. If one pesticide is in the top of its risk category and another is in the bottom of the next higher risk category, then the difference in the tax rate between the two pesticides may be greater than their difference in risk. In 2013, Denmark adopted an alternative pesticide taxation where the tax level of each approved pesticide is calculated based on its human health risks and environmental characteristics. Instead of distinct risk categories, the tax level is based on an environmental load index ranging from 0 to 40. The heterogeneity in tax levels is high, ranging from €25.5/ha to €0.57/ha. The new pesticide taxation scheme was projected to play a major role in achieving the Danish government objective of reducing the total amount of pesticides applied by 40 % from 2013 to 2015 (Böcker and Finger, 2016). Preliminary evaluations indicate that this objective has been fulfilled (Ørum et al., 2017).

Experiences from France show that the use of risk-based taxation of pesticides is not always effective. In 2008, a tax on diffuse pollution from agriculture was levied on the sale of pesticides, with the rate varying according to substance toxicity. However, tax rates of 5–6 % of the sale price of most pesticides were too low to reduce the demand for pesticides and to reach the goal of a 50 % reduction by 2018. Instead, the use of pesticides increased by 25 % from 2008 to 2014 (OECD, 2017).

In summary, there is evidence that risk-based taxation – which links taxation more closely to external effects – can be effective in reducing the environmental and health effects from pesticides. In contrast, non-differentiated taxation, e.g. ad-valorem or per unit taxes, of pesticides can have unintended consequences as quantity reductions can be achieved through substitution with more toxic products (Finger et al., 2017). Closer proportionality of taxes in relation to environmental and health risks may also increase the likelihood that a tax will be perceived as fair. Thus, risk-based taxation may not only enhance the economic desirability of taxes but may also increase their political legitimacy (Söderholm and Christiernsson, 2008). However, risk-based taxation is administratively more burdensome than per unit taxation for both regulators and industry. For example, more than 3000 pesticides are registered for use in the United Kingdom; in Norway, the number is below 200 (OECD, 2012).

The creation of a permit market is potentially a powerful policy instrument to control agricultural pollution. In a review of worldwide water quality markets, Greenhalgh and Selman (2012) found that most water quality markets seek to control *point source pollution*. Sometimes these markets allow point source polluters to offset their emissions by buying pollution credits from the agricultural sector. In these cases, farmers can certify their production process and generate pollution credits that can be sold to point source polluting parties. The case of Lake Taupo in New Zealand is the only case where a water quality market was created to reduce *non-point source pollution* from agricultural activities, one that is still in operation (see Box 4).

Box 4. Tradable emission quotas for regulating nitrogen pollution in Lake Taupo, New Zealand

Lake Taupo in New Zealand is the only case of what may be considered a pure non-point pollution source market. Since 2010, the Waikato regional government has established a limit for environmental nitrogen levels, aiming at a 20 % reduction by 2020. An Environmental Protection Fund has also been created. It uses public funds to gradually acquire (and retire) emission rights and finance environmental protection initiatives. Explicit rights have been assigned to producers based on their performance in previous years to allow them to generate a certain pollution load. Thus, those who need to increase their nitrogen emissions beyond their allotted quota need to buy quotas from other producers.

Emissions are calculated according to the Nitrogen Management Plan presented by each producer, and rights are assigned based on average nitrogen losses from 2000 to 2005. This has been a point of contention among stakeholders, because forest landowners who wish to convert their land to productive activities first have to purchase permits, while those who are already in the system receive them as a matter of course. From 2009 to June 2014, 23 transfers of emission rights to the Environmental Protection Fund had been carried out, for a total of 151 066 kg of nitrogen. This is a considerable number of transactions, which indicates a successful scheme. By 2015, the stated goal of reducing nitrogen discharges by 170 tonnes or 20 % of manageable nitrogen emissions had been met three years ahead of time. However, it is not possible to evaluate the instrument's capacity to improve water quality, because water permanence periods in Lake Taupo and nutrient permanence periods in underground aquifers are much longer than the time span of this instrument's implementation. (OECD, 2015)

Other examples of instruments targeting agricultural non-point source pollution, including manure production permits in the Netherlands and pasture irrigation districts in the San Jose Valley, California, have been working in the same direction. Manure production permits in the Netherlands, an indirect way of accounting for emissions, were used 1994–1997, allowing for the transfer of manure production permits instead of emissions. Manure permits were repealed in 1998 when a new regulatory approach, the Mineral Accounting System (MINAS), was introduced in the Netherlands.

Using market-based instruments to address non-point source pollution based on expected emissions can in principle be efficient, given that the damage depends not only on the inputs used, but also on the location, crop and time of application. However, their design is more complex since it requires sophisticated modelling of agricultural run-off, and such models have to be well understood by farmers so they know how their actions impact expected emissions. Instruments that require actual emission measurements may become more popular in the future, thanks to technological innovations that will reduce transaction costs and increase farmers' trust, along with fertiliser plans and registries.

3.5 Subsidies and subsidy removals

A subsidy is the mirror image of a tax. It can provide incentives to increase the use of alternatives to hazardous chemicals. Subsidies can take many different forms, including direct payments, tax reductions or exemptions and favourable loans. Subsidies do not fulfil the *polluter pays principle* and involve an opportunity cost of public funds. Moreover, once introduced, reducing or removing subsidies is often difficult since they encourage the formation of lobbies made up of beneficiaries striving to protect the subsidies. They can also lead to rebound effects, as a result of attracting more suppliers to the industry. However, subsidies tend to be popular and may be feasible from a practical standpoint when it is difficult to implement other policy instruments (Sterner and Coria, 2012).

The OECD Policy Instruments for the Environment database contains many entries for 'environmentally-motivated subsidies' that can have an indirect effect on the use of chemicals. Examples include tax exemptions for pollution or hazardous waste control, subsidies for energy savings and subsidies for clean-ups of contaminated sites (OECD, n.d.). In the US, several states subsidise the removal of lead paint from private properties. In for example Massachusetts, a tax credit is available for property owners who have paid for deleading. This subsidy is estimated to cost about $2.5 million annually in forgone tax revenue[7].

Input subsidies universally applied to farmers to promote better practices or decrease polluting inputs are not a common practice. The Republic of Korea is one of few cases where organic fertilisers are subsidised. Moreover, Denmark and Norway offer tax exonerations to farmers able to prove they fulfil specific fertiliser and pesticide management requirements. For example, farmers in Norway producing close to polluted river streams can apply for a subsidy to apply lower doses of P, depending on soil type and status.

Many countries provide substantial agrochemical subsidies in order to promote agricultural production and increase food security. However, these subsidies can have severe negative environmental effects and imply a high fiscal burden. These 'perverse subsidies' are also common in many other policy fields why 'subsidy removal' can be classified as a policy instrument in itself (Sterner and Coria, 2012).

The nature of the environmental effects of agrochemical subsidies depends on site-specific agri-environmental conditions and how the subsidy programme is designed. This makes a general

[7] http://children.massbudget.org/tax-break-removal-lead-paint

environmental assessment of the benefits of agricultural subsidy removal difficult. The fertiliser subsidy programme in India is briefly described in Box 5.

Box 5. The fertiliser subsidy programme in India

To incentivise agricultural production, the central government in India subsidises the use of chemical fertilisers. This subsidy has played an important role in increasing grain production in India. However, the cost for the government of keeping fertiliser prices below the market price has increased drastically since the introduction of the subsidy programme in the late 1970s. In 2015, the cost of fertiliser subsidies was estimated to reach approximately 12 billion USD in India.

There is only limited evidence on the environmental effects of the programme. There is evidence that the programme has led to an imbalanced use of nutrients by farmers by keeping the price of urea (an inexpensive form of nitrogen fertiliser) at a very low level. The excessive and imbalanced use of nutrients has contributed to soil degradation as well as water pollution (Gulati and Banerjee, 2015). A key challenge in reforming the programme is that the subsidies provide important benefits to many farmers with low income. (Praveen *et al.* 2017). In 2018, the Indian government reformed the subsidy program to prevent overuse of fertilizers and to reduce costs[8].

3.6 Charges and refunds used to finance hazardous waste management

Waste is classified as hazardous if it contains hazardous substances above a certain concentration. Hazardous substances, containers for hazardous substances as well as products containing hazardous substances (e.g. batteries, accumulators and waste electrical and electronic equipment) are considered as hazardous waste (Inglezakis and Moustakas, 2015)[9].

Market-based instruments have become a common strategy to encourage and finance proper hazardous waste management, mostly in developed countries but also with increasing frequency in developing countries. The market-based instruments used mainly consist of taxes or fees on the disposal of hazardous waste in landfills together with taxes or fees producers or consumers have to pay for products containing hazardous substances. In some cases, taxes or fees on a product are combined with a refund to the consumer upon disposal of the product. High-income countries have systems for managing most products containing hazardous substances while in low- and middle-income countries, hazardous waste management is, to a large extent, carried out by actors in the informal sector, who only collect products that contain materials with economic value.

Charges and deposit refund systems are frequently applied in the management of hazardous wastes, such as batteries, end of life vehicles and waste electrical and electronic equipment (WEEE). These instruments can both incentivise a reduced use of for example batteries containing hazardous chemicals and finance systems for collection and processing of hazardous waste.

Extended producer responsibility (EPR)

Extended producer responsibility (EPR) refers to the practice of holding the producer responsible for the entire life cycle of a product (Gupt and Sahay, 2015; Turner and Nugent, 2016; Wang et al., 2017) – a concept first launched in Europe in the early 1990s (Lifset et al., 2013). In addition to financing waste management, the idea with EPR is also to give the producer an incentive to reduce the waste generated and to improve the product design to facilitate easier recycling and resource recovery. Most common is to make the producer responsible for taking back their end-of-life products, either by themselves or by paying an authorised organisation. In some cases, for example in Switzerland and California, the producer charges the consumer an advance recycling fee (ARF), which is then paid to an organisation that handles the collection and recycling of the waste (Box 4**)** (Wang et al., 2017).

[8] https://www.livemint.com/Politics/GHChHFrP7cRA1yjRemycTK/DBT-scheme-for-fertilizer-subsidies-gets-cabinet-nod.html

[9] See for example the European Union directives 2000/532/EC, 2008/98/EC.

Deposit-refund or refund systems can also be applied to incentivise consumers to return their end-of-life products. In the European Union, the EPR system is recommended for hazardous waste such as batteries/accumulators, end-of-life vehicles (ELVs) and WEEE in directives 2006/66/EC, 2000/53/EC and 2012/19/EU, respectively (Inglezakis and Moustakas, 2015). The design of the EPR system is up to each member state, but the directives provide that a consumer must be able to dispose of a product at no charge or for a refund.

Sweden was the first country to introduce an EPR system (Lifset et al., 2013) for the recycling of aluminium cans. Several different collection systems, such as curb-side recycling stations, were tested by the producers, but it was difficult to establish a high collection rate (Franklin, 1997). In 1982, the government threatened to ban aluminium cans unless a 75 percent recycling rate was achieved by 1985. Finally, the problem was solved by introducing a deposit-refund system for aluminium cans. Today, Sweden has adopted the EPR system for handling the waste of many product types including products containing hazardous compounds such as batteries, WEEE, ELVs and scrap tyres. However, these EPR systems for hazardous waste do not include a consumer refund upon product disposal but instead rely entirely on consumers' awareness of the importance of proper disposal. Consumer behaviour is also strongly linked to the availability and accessibility of collection sites (Favot and Grassetti, 2017; Lim-Wavde et al., 2017), implying that an information-based incentive for disposal works better in countries with a developed waste management system. In Sweden, the producers finance waste collection by paying a fee to organisations set up by the government for each product category (Bragadóttir et al., 2014). Recycling is then organised within the municipal waste management system. The Swedish recovery and recycling system resembles the system in most European countries, with take-backs through specialised organisations being common for example in Germany and Portugal, while take-back schemes mainly involve reuse centres in France and Belgium and scrap dealers in Greece (Salhofer et al., 2016; Wang et al., 2017; Niza et al., 2014). In Switzerland, non-governmental producer responsibility organisations charge the consumer an advance recycling fee for managing the recovery and recycling of hazardous waste such as WEEE (Box 6). The effectiveness of the European systems is generally high but varies. For example in 2012, Sweden had the highest collection rate of WEEE at 17.5 kg/cap/yr., which can be compared with other European countries such as Switzerland, Germany and Italy at 16 kg/cap/yr., 8.5 kg/cap/yr. and 3.8 kg/cap/yr., respectively (Baldé, 2015). Effective collection infrastructures and campaigns to inform people about the importance of correct disposal of hazardous waste are essential to high collection rates (Salhofer et al., 2016; Niza et al., 2014; Favot and Grassetti, 2017).

Box 6. Advance recycling fee for waste electrical and electronic equipment in Switzerland

The cost of management of waste electrical and electronic equipment (WEEE) in Switzerland is ultimately charged to the consumer through an advance recycling fee (ARF) added to the product price (Wang et al., 2017; Khetriwal et al., 2009). The system was established already in 1998 by the *Ordinance on the Return, the Take-Back, and the Disposal of Electrical and Electronic Equipment*. The legislation defines clear responsibility for all stakeholders with the manufacturers and importers taking responsibility for their end-of-life products through two non-governmental producer responsibility organisations, SENS and SWICO.

At 16 kg/cap/yr., Switzerland has one of the highest WEEE collection rates in Europe. In Germany and Italy, the corresponding figures were 8.5 kg/cap/yr. and 3.8 kg/cap/yr., respectively, in 2012 (Baldé, 2015). The Swiss system is effective due to well-designed legislation with clear responsibility for all stakeholders, but also because of the country's long tradition of waste separation and the high public trust in the producer responsibility organisations (Khetriwal et al., 2009; Wang et al., 2017). The ARF for laptops is currently US$6.4 and for mobile phones US$0.1 (SWICO, 2018a), which is higher than average in Europe (SWICO, 2018b). A drawback of the advance recycling fee system is that by placing the cost on the consumer, it provides only limited incentives for the

Canada has a long tradition of stewardship organisations for hazardous waste collection (Turner and Nugent, 2016). Producers finance these organisations by paying a fee for each product that they produce. The Canadian stewardship organisations collect and recycle a range of products including batteries and WEEE and have a collection efficiency at the European level. In the United States, hazardous waste collection and recycling are not as developed as in Europe and Canada, although this varies across states (Seeberger et al., 2016; Turner and Nugent, 2016). In addition, only the most hazardous batteries containing heavy metals are classified as hazardous waste in the US (except in California), while all battery types are considered as hazardous waste in Europe and Canada (Sun et al., 2015). Moreover, the US lacks federal regulation of WEEE, but around half of the states have implemented their own rules for WEEE management (Seeberger et al., 2016). California was the first US state to introduce regulations for WEEE management by introducing the *Electronic Waste Recycling Act* in 2003 (Nixon and Saphores, 2007). The system is similar to the one used in Switzerland and is based on an advance recycling fee charged to the consumer. The Californian system was one of the most stringent WEEE recycling regulations at the time, but it has now become outdated with problems such as a too limited product range covered, low participation rates, and high management costs. Several other states, such as New York, now have introduced more stringent regulations (Loung, 2014).

Many low- and middle-income countries have severe problems with environmental pollution in connection to hazardous waste management. Dismantling and recycling are mainly performed by an uncontrolled informal sector using crude methods that pollute the environment and affect the health of the workers. These countries also struggle with the additional challenge of illegally receiving large amounts of hazardous waste from high-income countries, a consequence of much higher costs for waste recycling and waste disposal in landfills in these countries (Sigman and Stafford, 2011; Liu et al., 2017). To stop the trade of hazardous waste from high- to low-income countries, the international *Basel Convention on the Control of the Transboundary Movements of Hazardous Wastes and Their Disposal* came into force in 1992, with amendments made in 1995 and 1997 (Khan, 2016). Despite international agreements, the trade of hazardous waste continues and more than half of the WEEE generated in high-income countries is illegally exported (Agbor, 2016; Kellenberg, 2015; Bakhiyi et al., 2018). Moreover, half of the WEEE in India originates from OECD countries (Pathak et al., 2017). In the last decade, several low- and middle-income countries have introduced legislation for hazardous waste management by learning from the EPR system implemented in many high-income countries.

Both China and India have introduced EPR systems for WEEE management while Bangladesh and Pakistan still lack formal regulations (Islam et al., 2016; Iqbal et al., 2015; Gu et al., 2017b; Awasthi and Li, 2017). In China, the economic policy instrument is based on a government-controlled WEEE recycling fund introduced in 2012, providing financial support to formal recyclers to enable them to compete with the informal sector, which makes a larger profit by not following environmental regulations (Box 7) (Zeng et al., 2017; Gu et al., 2017b). The subsidies to formal recyclers have been effective and by 2015, 109 licenced enterprises were available all over China and the informal sector covered by the subsidy had been largely reduced (Zeng et al., 2017). In India, the *E-waste Management and Handling Rules* were introduced in 2011, using the EPR system to require the producers to set up a system for WEEE collection and only dispose of the collected WEEE to recycling units certified by the Pollution Control Board (Awasthi and Li, 2017). Although the rules resulted in the registration of 178 formal recycling units, 95 % of the WEEE in India was still recycled by the informal sector. A difference between the Chinese and Indian systems is that the recycling fund in China is controlled by the government. This fund likely helped China to establish a more functioning infrastructure for waste management compared with India. However, the e-waste rules in India were updated in 2016 with clearer responsibilities and a demand for the producers to implement a deposit refund system to promote collection (Pathak et al., 2017). It is too early to conclude whether this new regulation will succeed in improving Indian WEEE management.

Box 7. The waste electrical and electronic equipment recycling fund in China

Several areas in China have been severely polluted due to crude methods for WEEE recycling performed by an uncontrolled informal sector. To help create a regulated formal sector for safe WEEE recycling, the *WEEE Processing Fund Collection and Subsidy Management Approach* was introduced in 2012. Producers and importers pay a charge based on annual sales and product type. Revenues are placed in a fund that is owned by the government and used to support formal dismantling companies (Gu et al., 2017b). The fund system effectively reduced the informal sector for the original five products groups and established a formal sector consisting of more than 100 licenced enterprises (Zeng et al., 2017). However, the fund is financially imbalanced as subsidies to the formal enterprises are 5–10 times higher than the fee charged to the producers and importers of electrical and electronic products (Gu et al., 2017b; Zeng et al., 2017).

A limitation of the EPR system for WEEE management in China is that the fund system provides no incentive for environmental design of products. However, as the fund is controlled by the government, it could be designed to benefit producers and importers that design or sell less hazardous products that are easier to recycle (Gu et al., 2017b).

Deposit-refund and refund systems

Deposit- refund and refund systems are market-based instruments that give consumers an incentive for correct disposal of their hazardous waste. In traditional deposit-refund systems, consumers pay a deposit on top of the price of a new product and then receive a refund when turning in the end-of-life or consumed product. This system is for example widely used in the beverage industry as an incentive for consumers to return empty cans and bottles. In hazardous waste management, deposit-refund schemes have mainly been used for ELVs and lead acid batteries. The Nordic countries Denmark, Norway and Iceland all have deposit-refund schemes in place for ELVs (Bragadóttir et al., 2014), but it is difficult to conclude the effectiveness of these schemes as the recovery rate of ELVs is high in all European countries (in 2014, the average European recovery rate was 91 %; Eurostat, 2016). From 1975 to 2007, Sweden also had a deposit-refund system for ELVs, and compared with the UK, which lacks such a scheme, the Swedish system was highly effective during the investigated period from 1990 to 2005 (Manomaivibool, 2008). However, removal of the Swedish deposit-refund system in 2007 and replacement of it with producer responsibility regulations did not affect the ELV recovery rate (Eurostat, 2016).

Lead acid batteries commonly used in motor vehicles are collected within deposit-refund schemes in most states in the US (Walls, 2011). The customer pays a deposit of approximately $10 for a new battery and the fee is paid back to the customer if they return a spent battery. These programmes have increased the recovery rate of lead acid batteries in the US from 86 to 97 % (Walls, 2011). In addition, Norway has introduced a deposit-refund system for trichloroethylene, tetrachloroethylene, hydrofluorocarbons and perfluorocarbons where a refund is paid upon the returning of waste containing these chemicals (Bragadóttir et al., 2014). In 2011, the tax on trichloroethylene and tetrachloroethylene was approximately US$10/kg pure compound and about 40 % of the tax was returned when disposing waste containing either of the two substances.

Refund systems are commonly used in hazardous waste management in low- and middle-income countries for hazardous waste with significant economic value. Waste is considered more valuable in these countries than in high-income countries (Hu and Wen, 2015), due to the availability of cheap labour and crude recycling methods with low human and environmental protection. To encourage consumers to dispose of their hazardous waste to the controlled formal sector, a refund is paid to the consumer, often along with subsidies to formal enterprises to enable them to perform the more costly dismantling and recycling in line with regulations for human and environmental protection.

To promote recovery and recycling of ELVs within the formal sector in China, the government introduced a policy in 1995 that pays a scrapping compensation of approximately US$160 to consumers that dispose of their ELVs within the formal sector (Hu and Wen, 2015). In addition, command and control are used in the form of occasional controls and information campaigns. However, these instruments have failed to reduce the informal recycling of ELVs as the informal sector still pays ten times more for a scrap vehicle. The policies have been ineffective due to a lack of advertisement and because they have not addressed the mechanisms that enable the informal sector to pay so much more for an ELV.

Battery collection and recycling is poorly developed in both China and India (Sun et al., 2015; Sun et al., 2017; Gu et al., 2017a; Gupt, 2014). China uses the command and control principle for lead acid batteries by only allowing recycling by certified enterprises, while no regulation is available to encourage collection and recycling of household batteries (Sun et al., 2015; Sun et al., 2017). Despite the fact that the regulation requires recycling of lead acid batteries by formal recyclers, 30–40 % of the spent batteries of this type are still handled by the informal sector (Sun et al., 2017). Combing regulation with economic policy instruments, as is done for WEEE, has been suggested as a strategy to reduce the informal sector handling of lead acid batteries. Similar to China, Indian battery management rules introduced in 2011 only cover lead acid batteries (Gupt, 2014). The system resembles the e-waste rules in India and relies on the EPR principle. Similar to the e-waste rules, it has succeeded in establishing a formal sector for recycling, but has failed to significantly shift recycling from the informal to the formal sector (Box 8).

Box 8. Refund system for lead acid batteries in India

India has a well-established informal refund system for lead acid batteries, driven by the high price of lead (Gupt, 2014). Consumers return spent lead acid batteries to retailers, who pay a refund to the consumer. The lead is then extracted from the batteries in informal smelters, which often operate without the precautions needed to avoid exposure of the toxic lead to the workers or to the environment. To handle the problem, the Indian government established the *Batteries Management and Handling Rules* in 2001, requiring the retailers to sell their collected batteries back to the manufacturers or to registered smelters, who follow the required rules to avoid lead leakage and human exposure. In addition, the manufacturers are obliged to collect 90 % of their sold new batteries. In 2010, 353 smelters were registered in India but due to shortage of supply many were either not operating or operating at reduced capacity. In contrast, the banned informal sector experienced no shortage of supply. A survey in 2010 revealed that retailers bought back 69 % of their sold batteries and that they in turn sold these both to the manufacturers and to the illegal informal scrap dealers. Retailers had an incentive to sell to the informal scrap dealers since they made more frequent visits to the retailers, who often had limited storage capacity for spent batteries. Another important incentive was that the informal sector paid a slightly higher price for the batteries. Thus, the legislation has failed in shifting the lead acid battery recycling from the informal to the formal sector due to inability of the regulator to monitor the market along with insufficient coordination in the formal sector, giving the retailers an incentive to sell to the informal sector. A tax on new batteries, combined with a subsidy to promote recycling within the formal

There may be a scope for increasing the use of refund systems also in high-income countries, for example to increase the collection of hazardous waste products stored in homes. A survey in Finland reveals that consumers store unused mobile phones because they do not consider them as end-of life products, just end-of-use products, but that a small refund of less than US$6 would provide a large enough incentive for them to turn in their mobile phones to the collection system (Yla-Mella et al., 2015).

3.7 Fiscal revenue generation from market based-instruments

Fiscal revenue generated from environmentally related taxation constituted an average of around 2 % of gross domestic product (GDP) and around 5 % of total tax revenues in OECD countries in 2008. More than 95 % of these revenues stemmed from taxation of energy products, motor vehicles and transports. The revenue generation from other environmental taxes, such as taxes on agrochemicals, batteries and waste, was very limited (OECD, 2010). The present study finds no evidence that this general picture has changed since 2008.

Earmarking the revenue generated from chemical taxes is sometimes suggested in order to increase the political acceptance when introducing new taxes or financing chemical management activities (Söderholm, 2009). However, public financial management principles caution against earmarking since it makes the tax system less flexible. Moreover, since the level of revenues from a particular tax is unlikely to match the level of spending in a particular policy area, earmarking can result in both under- and over-funding (OECD, 2010).

In practice, a variety of earmarking and compensatory measures are common. For example, the revised pesticide taxation in Denmark in 2013 was accompanied with a reduced property tax on agricultural land to compensate farmers, and tax revenues were used to support organic farming and for administrative services. Revenues from pesticide taxation in Denmark were estimated to amount to about 88 million EUR in 2013 (or 0.23 % of the state budget). As provided by the Danish financial management legislation, these tax revenues first flow into the state treasury and are then returned for agricultural and environmental purposes (Böcker and Finger, 2016).

3.8 Institutional and political economy aspects

The introduction of market-based instruments for chemicals management is often met with resistance by interest groups that will face higher taxes or no longer benefit from a subsidy as a result of the instrument in question. For example, the Swedish tax on fertilisers was removed following protests from farmers that the tax would hurt their ability to compete with other European farmers. However, protests from interest groups are also common when other policy instruments to regulate chemicals are introduced. Certain amounts of policy consensus and institutional enforcement capacity are required to implement policy instruments that increase social welfare. This is a generally valid assertion, regardless of policy instrument choice.

Instruments should preferably be designed together with relevant stakeholders and then be gradually implemented. Consultation and monitoring of the reactions of stakeholders to a new incentive is important in order to avoid undesired side effects and to ensure that the incentives operate at the right level. How information about a policy change is communicated and how revenues are used is often critical to successful implementation.

It is important to have an adaptive capacity to re-design an instrument during the initial stages. Several schemes, mostly regarding fertilisers, have failed due to strong lobbying against them by the respective sector. It is key that, for example, farmers understand the relationship between their actions, the state of the environment and the policy goals to which the introduction of a tax or other market-based instruments should contribute. This implies a need to work together with relevant stakeholders to build agreements and share information during instrument design and implementation.

If taxes on emission proxies or inputs are introduced, it is important to help farmers to implement better practices and input substitutes. Many times, polluting inputs are essential for production, given soil conditions, and farmers only marginally decrease their use following price increases. Promoting cleaner alternatives is hence an important complement to the introduction of taxes.

Our review shows that the use of taxes on chemicals is rare in low- and middle-income countries. A pertinent question is whether this is due to a lack of specific institutional capacities only available in high-income countries. In general, the capacity to generate fiscal revenue is restrained in low-income countries, which tend to have a large informal economy and a relatively large share of agriculture in total output. This makes income taxes play a relatively less important role for revenue generation. Instead indirect taxes, such as foreign trade taxes, excise taxes on e.g. alcohol, tobacco and fuel and value-added taxes, play a relatively more important role, compared with in high-income countries (Di John, 2006; Slunge and Sterner, 2009). In principle, taxation of specific or groups of chemicals should therefore not be more difficult than taxing income in many low-income countries. However, a limited general capacity to assess when and how to use market-based instruments and to implement such instruments in practice may contribute to the low use of them. More research on institutional aspects of the use of policy instruments for chemicals management is warranted.

Our review also indicates that the effectiveness of market-based instruments is linked to other institutional factors. In the area of hazardous waste management, effectiveness is associated with the level of awareness among citizens about the importance of correct disposal and the accessibility of collection and recycling stations (Lim-Wavde et al., 2017; Favot and Grassetti, 2017). For example, the hazardous waste management system in Canada is more efficient than in many states in the US, due to a well-developed system for waste collection along with a long tradition of the public separating different waste streams (Turner and Nugent, 2016; Sun et al., 2015; Seeberger et al., 2016). In addition, the Nordic countries have more efficient systems for hazardous waste management than some other European countries due to higher levels of consumer awareness and extensive waste management systems already being in place (Bragadóttir et al., 2014).

4 FUTURE OUTLOOK AND RECOMMENDATIONS

In this section, we discuss lessons learned and policy options for improved chemicals management by increased use of market-based instruments in chemicals management.

4.1 Lessons learned

This review shows that market-based instruments are used in chemicals management mainly in high-income countries. However, several low- and middle-income countries have also begun to use market-based instruments in hazardous waste management. As chemical policy has targeted the prevention of health and environmental damage from a limited number of highly hazardous chemicals, bans and use restrictions have been the most frequently used policy instruments. Price-type instruments give the regulator less certainty about effects on the quantity used of a specific substance, and are therefore less appropriate for addressing chemicals of very high concern. Nevertheless, in some cases, taxes and other market-based instruments have provided important complements to bans and use restrictions, also in relation to very hazardous substances such as lead.

The main benefits of market-based instruments are that they can stimulate cost-effective substitution and spur innovation. In many cases, it can be beneficial to combine market-based instruments with restrictions on hazardous chemical exposure. Introducing a tax or charge that creates incentives for substitution and innovation can also make it easier to implement tougher use restrictions or even bans at a later stage. Hence, market-based instruments should be seen as complements rather than substitutes to bans and use restrictions in chemicals management. Transparency and access to information on the use of chemicals and their effects are common pre-requisites for effective design and implementation of market-based instruments.

There are growing concerns about the cumulative effect and combination effects of low-dose exposure to multiple chemicals. Even though the damage cost of each chemical when assessed one by one is not very high, there is evidence that the cumulative effect may be considerable. This could motivate a broader use of market-based instruments in chemicals management (Sadler, 2000).

The repeated calls to speed up innovation in order create chemicals that are ‘safe by design’ may also generate an interest in an increased use of market-based instruments. By changing the relative prices in favour of less hazardous chemicals, chemical taxation or other market-based instruments can change the rate of return of investments in favour of alternative technology. In addition, governments can generate additional incentives for research and innovation for example by providing tax credits for expenses related to research and development or by providing favourable treatment of capital or labour expenses. This would also address the market failure related to imperfect information and knowledge spill-overs, which tend to result in a sub-optimal level of innovation (Jaffe et al., 2005).

Our review also points to important knowledge gaps regarding the use of market-based instruments for chemicals management. There is a lack of good evaluations of the effectiveness of various instruments. This is unfortunate, since the performance of policy instruments for chemicals management depends on context-specific factors such as price elasticities, market structure, the availability of substitutes and on the exposure characteristics of the targeted chemicals. Thus, it would be important to establish a policy learning process with systematic monitoring and evaluation of the effectiveness of policy instruments for chemicals management in different sectors and contexts. Such a policy learning process should also encompass and identify institutional and political economy factors, that are critical during instrument design and implementation.

4.2 Policy options

- *Expand the use of risk-based taxation of hazardous chemicals.* Our review indicates that risk-based taxation – which links taxation more closely to specific environment and health risks – can be effective in reducing the environmental and health effects of pesticides. It is likely that instruments that require actual emission measurements will become more popular in the future, thanks to information technology that can help reduce monitoring costs. However, more knowledge about exposure-damage relationships is needed in order to properly design such instruments. Experimentation and learning about the effectiveness of such taxation schemes is warranted.

- *Evaluate and address the effects of subsidies and other policies generating perverse incentives for increased use of hazardous chemicals in agriculture and other sectors*. Reducing subsidies for fossil fuels has been identified as a cost-effective instrument to reduce greenhouse gas emissions. Similarly, addressing subsidies that incentivise an increased use of agro- and other hazardous chemicals is most likely key to improved chemicals management. However, more research on the extent of such subsidies as well as their effects on environmental resources, income distribution and health is needed. As subsidies have a tendency to create strong vested interests, subsidy removals or reductions should be carefully designed.

- *Expand the use of risk-based taxation of hazardous chemicals.* Our review indicates that risk-based taxation – which links taxation more closely to specific environment and health risks – can be effective in reducing the environmental and health effects of pesticides. It is likely that instruments that require actual emission measurements will become more popular in the future, thanks to information technology that can help reduce monitoring costs. However, more knowledge about exposure-damage relationships is needed in order to properly design such instruments. Experimentation and learning about the effectiveness of such taxation schemes is warranted.

- *Use charges to speed up the phase out of substances of very high concern.* Chemicals classified as especially problematic from an environment and/or health perspective are placed on the so-called Candidate List within the European chemicals regulation REACH or corresponding lists in other jurisdictions. However, even though the candidate list sends a strong signal that the listed substances should be phased out, only a subset of the chemicals are banned and it can take considerable time before the substances on the list are phased out. Even though the externalities from the use of these substances are difficult to quantify precisely, inclusion on the Candidate List indicates that the cost is considerable. Even a small charge may incentivise substitution and the development of alternatives that eventually will make a total phase-out of the Candidate List substances easier.

- *Evaluate experiences regarding taxation of chemicals in groups.* Denmark and Sweden have implemented taxation of phthalates in PVC and flame retardants in electronic products. While there are some indications of a lack of precision and high administrative costs associated with these taxes, there is also a lack of formal evaluations. These initiatives may provide important lessons for future policy instrument design and should be studied further.

- *Incentivise sound chemicals management by making producers bear the full environmental and health costs during the entire product life cycle.* The establishment of systems with extended producer responsibility has in many countries succeeded in shifting the cost of waste management from authorities to producers. Implementing legal obligations for firms to prevent and remedy environmental damages – for example requirements to be insured or to build up funds for clean-up or waste management – is another example of an institutional reform that may generate incentives in line with the polluter-pays principle. These types of institutional reforms provide important complements to market-based instruments and other policy instruments in generating incentives for sound chemicals management.

- *Expand requirements on transparency and access to information related to chemicals*. Access to information is key to successful policy instrument design and implementation. Requirements on information disclosure can also in itself incentivise sound chemicals management among producers, retailers and other agents.

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Appendix 1. Literature search in Econlit

Keywords combinations selected (number of hits)

1. chemic* AND tax* (NOT “taxa”) (35)
2. chemic* AND fee (2)
3. chemic* AND subsid* (32)
4. “chemical pollution” AND tax* (1)
5. “chemical pollution” AND fee (0)
6. “chemical pollution” AND subsid* (0)
7. Pestic* AND tax* (13)
8. Pestic* AND fee (0)
9. Pestic* AND subsid* (14)
10. Fertiliz* AND tax* (NOT “taxa”) (24)
11. Fertiliz* AND fee (0)
12. Fertiliz* AND subsid* (106)
13. Fertiliz* AND tax* (NOT “taxa”) (8)
14. Fertiliz* AND fee (0)
15. Fertiliz* AND subsid* (12)
16. Lead AND tax* and “heavy metals” (0)
17. Lead AND fee and “heavy metals” (0)
18. Lead AND subsid* and “heavy metals” (0)
19. Cadmium AND tax* (0)
20. Cadmium AND fee (0)
21. Cadmium AND subsid* (0)
22. Mercury AND tax* (0)
23. Mercury AND fee (0)
24. Mercury AND subsid* (0)
25. batter* AND tax* (36)
26. batter* AND fee (8)
27. batter* AND subsid* (25)
28. batter* AND refund (2)
29. plastic bag* AND tax* (3)
30. plastic bag * AND fee (2)
31. plastic bag * AND subsid* (1)
32. plastic bag * AND refund (0)

Appendix 2: Information on the use of market-based instruments for chemicals management in the OECD Policy Instruments for the Environment database

Chemical/product	**Number of countries**	**Countries with the specific market-based instrument**			
		Tax	**Fee/Charge**	**Deposit-refund systems**	**Tradable permit systems**
Pesticides	7	Denmark, Italy, Norway, Sweden, US	Bulgaria, Canada		
Fertilisers	1		Bulgaria		
Ozone depleting substances	10	Australia, Czech Republic, Denmark, Spain, US	FYR of Macedonia, Latvia, Montenegro, Serbia		Canada[1], US[2]
Chlorinated solvents	4	Denmark, Norway, US			Canada[3]
Polyvinylcloride and phthalates	1	Denmark			
Pesticide or chemical containers	4		Canada, Korea	Poland, US	
Tyres[a]	15	Canada, Denmark, Finland, Hungary, Slovenia, South Africa, US	Bulgaria, Canada, Croatia, FYR of Macedonia, Latvia, Lithuania, Malta, Poland, Portugal, US	Canada, Denmark, US	
Batteries/accumulators[a]	17	Canada, Denmark, Hungary, Iceland, Liechtenstein, Sweden, US	Austria, Bulgaria, Denmark, FYR of Macedonia, Italy, Korea, Lithuania, Poland, Portugal, Switzerland	Canada, Denmark, Lithuania, Mexico, Poland, US	
Electrical and electronic products[a]	13	Canada, Denmark, Hungary, Slovenia	Canada, China, FYR of Macedonia, Korea, Liechtenstein,		

Chemical/product	Number of countries	Countries with the specific market-based instrument			
		Tax	**Fee/Charge**	**Deposit-refund systems**	**Tradable permit systems**
			Lithuania, Malta, Poland, Portugal, Switzerland		
Vehicles[a]	10	Slovenia, Russia	Bulgaria, Czech Republic, Croatia, Finland, Switzerland	Denmark, Finland, Norway, Sweden	
Hazardous waste (generation/disposal)	15	Belgium, Brazil, Czech Republic, Estonia, Hungary, Iceland, Poland, Portugal, Spain, US	Croatia, Denmark, Germany, Montenegro, Serbia		

Notes: Data retrieved from the OECD Policy Instruments for the Environment (PINE, http://www2.oecd.org/ecoinst/queries/) database. The database has been developed in co-operation with experts at the European Environment Agency and other government agencies. It covers economic policies in OECD and other countries (107 countries in total), but the data is more complete for the OECD countries. The table contains data from the environmental domains land contamination, ozone layer protection and waste management. It only includes currently active policies although it is unclear how frequently the database evaluates whether the policies are still active. Differences between the content of the table and information in the text in this report, which is also based on available literature, are therefore possible. Countries consisting of multiple states are listed if the policy is present in at least one of the states.

[a] Economic policy targeting either the new or the discarded product.

[1] HCFCs allowance system introduced in 1990.

[2] Tradable permit system for ozone-depleting substances introduced in 1989.

[3] Allowance system for TCE (Trichloroethylene) and PERC (Tetrachloroethylene) introduced in 2001.