Instrument Mixes for Environmental Policy

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QUELLES COMBINAISONS D'INSTRUMENTS?

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Foreword

What are the impacts on environmental effectiveness and economic efficiency of using an "instrument mix", rather than a single instrument, to address a given environmental problem? Which are the main arguments for using such instrument mixes, and are the instrument mixes currently in use actually well designed in response to these arguments? These are the main questions addressed in this report, which is based on a series of in-depth case studies analysing instrument mixes applied in OECD member countries to address household waste, non-point sources of water pollution in agriculture, residential energy efficiency, regional air pollution and emissions to air of mercury.

There are *a number of good arguments for using a mix of instruments to address a specific environmental problem: First and foremost, many environmental problems are of a "multi-aspect" nature – in addition to the* total *amounts of releases of a certain pollutant, it can, for example, also matter* where *emissions take place,* when *they occur,* how *a polluting product is applied, etc. Secondly, certain instruments can* mutually underpin *each other – as when a labelling scheme enhances the responsiveness of firms and households to an environmentally related tax, while the existence of the tax helps draw attention to the labelling scheme. Often, a mix of instruments is required in order to address* non-environmental "failures" *in the markets in which environmental policy instruments operate, such as lacking information, ill-defined property rights, market power, etc. Sometimes such mixes can also limit* compliance-cost uncertainty, *enhance enforcement possibilities and reduce administrative costs.*

On the other hand, when applying several instruments in a mix, there is a danger that one instrument will unnecessarily hamper the flexibility to find low-cost solutions to a problem that another instrument could have offered if it had been used on its own. In other cases, some of the instruments in a mix are simply redundant, contributing only to increase total administrative costs.

*Many of the instrument mixes analysed in this project respond well to some of the arguments above for using instruments in combination. However, in a number of cases, the use of overlapping instruments reduce the economic efficiency of the mix, while in other cases, the lack of some instruments can jeopardise environmental effectiveness and/or economic efficiency. Such problems can, in part, be due to insufficient analyses of the impacts of a given instrument mix, both prior to its implementation (*ex ante*) and once it has been in place for some time (*ex post*).*

On this background, the report offers a number of recommendations on how environmental instrument mixes should be assessed, and on how their design could be improved.

Previous drafts of this report – or parts of it – have benefited from comments from several OECD bodies. The case studies on household waste have been discussed by the Working Group on Waste Prevention and Recycling. The case studies on non-point sources of water pollution in agriculture have been discussed by the Joint Working Party on Agriculture and Environment. The Working Party on National Environment Policies has discussed the entire report and agreed to its declassification.

The report was written by Nils Axel Braathen and Ysé Serret of OECD's Environment Directorate. Jakob Maag of COWI A/S, Denmark, prepared a draft of the case study on emissions to air of mercury.

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Acronyms

AOO	Dutch Waste Management Council
BAT	Best Available Technology
BATEA	Best Available Technology Economically Achievable
BMP	Best Management Practices
BOD	Biological Oxygen Demand
CAFO	Confined Animal Feeding Operations
CAP	EU Common Agricultural Policy
CBP	Chesapeake Bay Program
CCME	Canadian Council of Ministers of the Environment
CEAP	US Conservation Effects Assessment Project
CEC	Council of the European Communities
CEU	Council of the European Union
CH_4	Methane
CMHC	Canada Mortgage and Housing Corporation
CO_2	Carbon dioxide
CREP	US Conservation Resource Enhancement Program
CRP	US Conservation Reserve Program
CSA	Canadian Standards Association
DCR	Virginia's Department for Conservation and Recreation
DEFRA	UK Department for Environment, Food and Rural Affairs
DEQ	Virginia's Department of Environmental Quality
DfE	Design-for-Environment
DLA	US Defense Logistics Agency
EAE	Efficiency and Alternative Energy
ECOS	US Environmental Council of the States
EEC	UK Energy Efficiency Commitments
EFR	Ecological Fiscal Reforms
EGH	Canadian EnerGuide for Houses
EGs	Emission Guidelines
EPA	US Environmental Protection Agency
EPR	Extended Producer Responsibility
EQIP	US Environmental Quality Incentives Program
EST	UK Energy Saving Trust
ETS	EU Emission Trading Scheme
EU	European Union
FDA	US Food and Drug Administration
FFDCA	US Federal Food, Drug, and Cosmetic Act
FIFRA	US Federal Insecticide, Fungicide, and Rodenticide Act

FSA	US Farm Service Agency
GAEC	Good Agricultural and Environmental Condition
GDP	Gross Domestic Product
GHG	Greenhouse Gas
HAPs	Hazardous Air Pollutants
Hg	Mercury
IPPC	EU Integrated Pollution Prevention and Control Directive
LATS	UK Landfill Allowance Trading Scheme
LCPD	EU Large Combustion Plants Directive
LESA	UK Landlords' Energy Saving Allowance
LRTAP	UNECE Convention on Long-range Transboundary Air Pollution
MACT	Maximum Achievable Control Technology
MAFF	UK Ministry for Agriculture, Food and Fisheries
MINAS	Dutch Minerals Accounting System
MNECH	Canadian Model National Energy Code for Houses
MoU	Memorandum of Understanding
MTAS	Dutch Manure Transfer Agreement System
MWC	Municipal Waste Combustor
N	Nitrogen
NECD	EU National Emissions Ceilings Directive
NESHAPs	US National Emission Standards for Hazardous Air Pollutants
NEWMOA	US Northeast Waste Management Officials' Association
NH_3	Ammonia
NMP	Nutrient Management Plans
NO_x	Nitrogen oxides
NPEP	US National Partnership for Environmental Priorities Program
NRC	National Research Council Canada
NRCS	Natural Resources Conservation Service of the USDA
NRTEE	Canada National Round Table on the Environment and the Economy
NSA	Nitrate Sensitive Areas
NSPSs	US New Source Performance Standards
NVMSRP	US National Vehicle Mercury Switch Removal Program
NVZ	Nitrate Vulnerable Zones
OEE	Natural Resources Canada's Office of Energy Efficiency
OERD	Canadian Office of Energy Research and Development
OFGEM	UK Office of Gas and Electricity Markets
ORD	US EPA Office of Research and Development
OSPAR	Convention for the Protection of the Marine Environment of the North-East Atlantic
P	Phosphorous
P_2O_5	Phosphate
PERD	Canadian Programme for Energy Research and Development
POPs	Persistent organic pollutants
PRN	UK Packaging Waste Recovery Notes
PRO	Producer Responsibility Organisation
RCRA	US Resource Conservation and Recovery Act
RRAP	Canadian Residential Rehabilitation Assistance Programme

SFT	Norwegian Pollution Authority
SNUR	US Significant New Use Rule
SO_2	Sulphur dioxide
TMDL	US Total Maximum Daily Load
TRI	US Toxics Release Inventory
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environment Programme
USDA	US Department of Agriculture
UWR	US Universal Waste Rule
VOCs	Volatile Organic Compounds
VROM	Dutch Ministry of Housing, Spatial Planning and the Environment
WCA	UK Waste Collection Authorities
WDA	UK Waste Disposal Authorities
WEEE	EU Waste Electrical and Electronic Equipment Directive
WHO	World Health Organization
WPZ	UK Water Protection Zones
WRAP	UK Waste and Resources Action Programme
WTO	World Trade Organization

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Executive Summary

Main objectives of the report

What are the impacts on environmental effectiveness and economic efficiency of using an "instrument mix", rather than a single instrument, to address a given environmental problem? Which are the main arguments for using such instrument mixes, and are the instrument mixes currently in use actually well designed in response to these arguments? These are the main questions addressed in this report, which is based on a series of in-depth case studies analysing instrument mixes applied in OECD member countries used to address household waste, non-point sources of water pollution in agriculture, residential energy efficiency, regional air pollution and emissions to air of mercury.

The main objectives of the project have been to derive further insights on – and, where appropriate, to elaborate policy recommendations concerning:

- How should member countries assess the environmental effectiveness and economic efficiency of a given instrument mix?
- What are the *additional* impacts (in terms of environmental effectiveness and economic efficiency) that appear because a given instrument is *used in combination* with one or more other instruments?; and
- Which types of instrument mixes are likely to provide high environmental effectiveness and economic efficiency?

Reasons for using a mix of instruments

There *are* a number of good arguments for using a mix of instruments to address a specific environmental problem: First and foremost, many environmental problems are of a "multi-aspect" nature – in addition to the *total* amounts of releases of a certain pollutant, it can, for example, also matter *where* emissions take place, *when* they occur, *how* a polluting product is applied, etc. Secondly, certain instruments can *mutually underpin* each other – as when a labelling scheme enhances the responsiveness of firms and households to an environmentally related tax, while the existence of the tax help draw attention to the labelling scheme. Often, a mix of instruments is required in order to address non-environmental "failures" in the markets in which environmental policy instruments operate, such as lacking information, ill-defined property rights, market power, etc. Sometimes such mixes can also limit compliance-cost uncertainty, enhance enforcement possibilities and reduce administrative costs.

Reasons for restricting the number of instruments in a "mix"

When applying several policy instruments in a mix, there is a danger that one instrument will unnecessarily hamper the flexibility to find low-cost solutions to a problem that another instrument could have offered if it had been used on its own. In other cases, some of the instruments in a mix are simply redundant, contributing only to increase total administrative costs.

Detailed country case studies were analysed

This report is based on a number of in-depth case studies of instrument mixes applied in selected OECD countries to address a number of different environmental problems. Regarding *household waste*, the instrument mixes applied in the Netherlands and in the United Kingdom were analysed; in the latter case, focused on the instrument mixes applied in England and in Wales, alongside the nationwide instruments.

Instrument mixes dealing with *non-point sources of water pollution in agriculture* were analysed in four countries: Denmark, the Netherlands, the United Kingdom and the Chesapeake Bay area in the United States. For Denmark, separate studies were made on instrument mixes addressing nitrogen run-off, phosphorous run-off and pesticides use. Mixes addressing both pesticide use and nutrient run-off were also analysed regarding the United Kingdom, while the case studies of the Netherlands and the Chesapeake Bay area only focused on nutrients run-off. As regards the Chesapeake Bay, both Federal measures and State-level measures in Virginia and Maryland were analysed.

As for *residential energy efficiency*, the instrument mixes applied in the United Kingdom and in Canada were analysed – with attention given to both Federal and Provincial instruments in the latter case. The focus in both cases was on the thermal energy efficiency of buildings and on the energy efficiency of household appliances.

The case studies *regarding regional air* pollution focused on the instrument mixes used to limit SO_2 and NO_x emissions in Sweden and in Canada. The Canadian case study addressed instruments applied at the Federal level and in the Provinces of Ontario and Alberta.

Instrument mixes addressing *emissions to air of mercury* in Norway, Sweden and in the United States were also analysed. In the latter case, the focus was on instruments applied at the Federal level, and on instruments in use in the States of Massachusetts and Michigan.

Many of the cases analysed respond well to the arguments for using instruments in combination, *e.g.* by reflecting the *multi-aspect character* of many of the environmental issues addressed. Many instruments are also applied to reduce *information problems* and other "failures" in the markets in which environmental policy operate, and in several cases one instrument enhances the functioning of other instruments applied. However, in a number of situations, the use of *overlapping* instruments reduces the economic efficiency of the mix, while in other cases, the *lack of some instruments* can jeopardise environmental effectiveness and/or economic efficiency.

Better ex ante *and* ex post *assessments are needed*

On this background, the report offers a number a number of recommendations on how environmental instrument mixes should be assessed, and on how their design could be improved.

First and foremost, policy-makers should consider carefully whether the benefits of further environmental improvements and the costs to society of achieving these improvements balance reasonably well. This implies the need to assess the *targets* set for environmental policy. If the sum of *all* the benefits of additional improvements can be expected to exceed the related costs, the improvements ought to be pursued. If the costs clearly outweigh the sum of all relevant benefits – even if the most efficient policy instruments were to be applied in order to reach the targets – a reconsideration of present priorities would be appropriate.

Policy-makers should also make in-depth *ex ante* assessment of any new *instrument* being considered for implementation – to assess what each instrument would contribute as regards environmental effectiveness and economic efficiency. They should also regularly undertake careful *ex post* reviews of the instruments they apply to achieve a given target – and eliminate or modify redundant instruments, *i.a.* to reduce administrative costs.

Careful review of current or potential instrument use, as well as the design of improved instrument mixes, clearly requires that policy-makers have a good understanding of the environmental issue at hand, covering all the aspects of a "multi-aspect problem", and of the interrelations between this environmental issue and other related environmental and non-environmental policy issues.

Instruments should address environmental problems as broadly as possible

Both from the perspectives of environmental effectiveness and economic efficiency, policy-makers should apply instruments that address a given environmental problem *as broadly as possible*. Further, one should (to the extent possible) provide *similar incentives at the margin* to all producers that contribute to the problem in question. Economic instruments can "automatically" provide equal marginal abatement incentives, but various regulatory instruments can (at least partially) also do the same job.

For "multi-aspect" environmental problems, policy-makers should supplement instruments that address *total amounts* of pollution with instruments that address *the way* a certain product is used, *when* it is used, *where* it is used, etc. In many cases, regulatory instruments, information instruments, training, etc., can be better suited to address these dimensions than *e.g.* a tax or a trading system.

Instrument mixes should provide flexibility

In order to exploit possibilities for various instruments to mutually underpin each other (so that the application of one instrument enhances the effectiveness and efficiency of another), instruments that *provide as much flexibility as possible* to the targeted groups should be used. Economic instruments will generally provide such flexibility – but *some types* of regulatory instruments can also do so.

Scope for mutually enhancing instruments should be exploited…

Various labelling schemes and other information-instruments can enhance the environmental effectiveness of taxes, fees and charges – especially if they convey information on *private benefits*. Hence – with the general caveat of making sure that benefits exceed costs – such instrument combinations could usefully be applied.

… but otherwise, overlapping instruments ought to be avoided

Except for situations where mutual reinforcement between instruments is likely, or when the instruments address different "aspects" of a given problem, policy-makers should generally *avoid introducing overlapping instruments* – as such overlaps will tend to reduce the desired flexibility and create unnecessary administrative costs.

Avoid "over-burdening" environmental policy instruments with non-environmental tasks

It is advisable to address social concerns primarily with non-environmental policy instruments (*e.g.* the social security system or the tax system), rather than to modify environmental policy instruments. This will *inter alia* make it easier to provide an incentive at the margin for low-income households to behave in environmentally benign ways.

To the extent policy-makers want to address any negative impacts on the competitiveness of certain sectors stemming from environmental policies, it is again important to provide an incentive at the margin to abate emissions. Such incentives can, for example, be provided through (both grandfathered and auctioned) emission trading systems and through environmentally related taxes – possibly with recycling of revenues back to the sectors of concern.

It can also often be preferable to primarily address *non-environmental market-failures* (market power, incomplete property rights, split incentives, etc.) with non-environmental instruments, such as competition policy instruments, improvements to patenting systems, deregulation of the housing markets, etc.

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Chapter 1

Introduction and Basic Concepts

This chapter introduces the main objectives of this report. The report, based on a series of in-depth case studies, examines the impacts on environmental effectiveness and economic efficiency of using an "instrument mix", rather than a single instrument, to address a given environmental problem. The case studies review instrument mixes applied in OECD member countries to address household waste, non-point sources of water pollution in agriculture, residential energy efficiency, regional air pollution and emissions to air of mercury. The report examines how member countries assess the environmental effectiveness and economic efficiency of a given instrument or instrument mix. It analyses the additional impacts on environmental effectiveness and economic efficiency caused by the combination of instruments, and asks which types of instrument mixes are likely to provide the best results.

1.1. Background

In 2003, the OECD Working Party on National Environmental Policy (WPNEP) launched a project to examine the environmental effectiveness and economic efficiency of "mixes of instruments" used in environmental policy. WPNEP has long been interested in these questions at the level of *individual* policy instruments; this project looked at essentially the same questions from the perspective of *combinations* of instruments.

The main objectives of the project have been to derive further insights – and, where appropriate, elaborate policy recommendations – on:

- How should member countries assess the environmental effectiveness and economic efficiency of a given instrument mix?
- What are the *additional* impacts (in terms of environmental effectiveness and economic efficiency) that appear because a given instrument is *used in combination* with one or more other instruments?; and
- Which types of instrument mixes are likely to provide high environmental effectiveness and economic efficiency?

To support the work, detailed case studies were carried out in five areas of environmental policy:

- municipal solid waste management, with an emphasis on household waste;
- non-point sources of water pollution, with an emphasis on nitrogen and phosphorous run-off and pesticide use in agriculture;
- regional air pollution, with an emphasis on contributors to acidification (sulphur dioxide and nitrogen oxides);
- residential energy efficiency, with an emphasis on dwellings and on household appliances; and
- emissions to air of mercury.

After a discussion of some background concepts in this chapter, the report presents an overview of the specific instrument mixes currently in use in the five environmental policy areas that were examined in the case studies (Chapters 2-6). This is followed by an analysis of the economic efficiency and environmental effectiveness of the various instruments mixes (Chapter 7). Some conclusions and policy recommendations follow at the end of the report (Chapter 8).

1.2. Some basic concepts

Any individual policy instrument should only be introduced if there is a reasonable expectation that this instrument will add to the total welfare of society. The same is true for *combinations* of policy instruments. In practical terms for the environmental policy domain, this means that: *i)* the marginal cost of implementing a given instrument mix should be less than its marginal benefit ("cost-benefit" criterion); *ii)* the marginal cost of

applying a given instrument mix should be as low as possible ("cost-effectiveness" criterion); and *iii)* the marginal environmental benefit associated with a given instrument mix should be a high as possible ("environmental effectiveness" criterion). All of the following discussion of instrument mixes is concerned, in one way or another, with these three criteria.

As one example, administrative costs associated with a particular combination of policy instruments are sometimes overlooked in the decision-making process. Including them in the original discussion will give a clearer picture of the net benefits likely to result from the mix, as well as a better understanding of the environmental results that are likely to follow from implementing that mix.

Any instrument mix also risks being environmentally ineffective if the "correct" price signals are not transmitted to decision-makers. In the household waste area, despite long-standing policies to address (*inter alia*) total amounts of household waste in the Netherlands – and more recent measures with the same objective in the United Kingdom – total household waste is today as high in the Netherlands as it is in United Kingdom, and the total amounts of household waste have *increased* over time in both countries. Part of the reason is that none of the households in the United Kingdom, and only a limited number of households in the Netherlands, face waste collection charges that vary with the amount of waste they generate.

On the other hand, it is difficult in a report of this scale to fully explore *all* dimensions of the welfare improvement problem. It has therefore been necessary to focus here on only some of those dimensions. Again in the interest of practicality, much of the following discussion emphasises the "co-ordination" aspect of the problem – an aspect that is most fundamentally concerned with cost-effectiveness, but which also has interrelations with both environmental effectiveness and economic efficiency.

1.3. How many instruments should be in the instrument mix?

A rather obvious first requirement for developing an environmentally effective and economically efficient instrument mix is to have a good understanding of the environmental problem that is being addressed. A key question at the outset is whether the environmental issue at hand is a "single-aspect" or a "multi-aspect" one. The former is an environmental problem with only one relevant dimension or characteristic – for example, the total amount of a certain type of emission. To fall into this category, it should be of no importance *where* the emissions take place, *when* they take place, etc. Emissions of ozone-depleting chemicals and greenhouse gases are (at least close to) examples of such issues.

Most environmental problems, however, fall into the "multi-aspect" category. For example, we are usually *not only* interested in the *total* amount of SO_2 emissions or in the total amount of noise created at an airport. It is also relevant to know *where* the SO_2 emissions take place (upwind or downwind from large population concentrations or in ecosystems that are very sensitive to acid rain), or *when* they do (*e.g.* time of year). Concerning airport noise, it is usually important to know whether the planes pass over large population concentrations during take-off and landings, at what time of day (or night) the noise is created, etc.

Another important consideration for policy design is whether it is technically feasible to address the environmental problem directly – or if some "proxy" for the underlying externality would have to be addressed instead. For example, it is technically infeasible to

measure and control run-off of nutrients from each farm directly. Similarly, one cannot address NO_x emissions from individual motor vehicles directly – in part because the costs of continuously measuring the emissions from each vehicle would be prohibitive.

If we are facing a "single-aspect" environmental issue that can be addressed directly, a "first-best optimum" can theoretically be reached through the use of a single instrument – under certain frequently-made assumptions regarding the functioning of relevant markets. These assumptions include the existence of well-defined property rights, full information and perfect foresight among all actors, and that no buyer or seller is sufficiently large to be able to hold market-power. If these assumptions are reasonably valid for the case at hand, the use of only a single instrument – rather than a mix of instruments – could be preferable.

If, on the other hand, we cannot assume effective functioning of the relevant markets, "second-best" theory comes into play. As emphasised by Lipsey and Lancaster (1956), in such situations, *one instrument per market failure* will be needed. For example, if the assumption about full information is invalid (while all the other assumptions hold), one instrument would be needed to address the environmental externality *per se*, and another instrument would be needed to address the information failure.[1]

In the more common case of "multi-aspect" environmental problems, a first-best optimum *cannot* be reached by applying only one instrument – according to the "Tinbergen Rule".[2] A combination of several instruments – an "instrument mix" – will be needed.

But how many instruments? In practice, it can be rather difficult to count exactly how many relevant policy objectives exist concerning a given environmental issue. Counting the number of instruments is not easy either. Many instruments include a long list of "independent mechanisms", including: separate tax-rates or exemptions and refund mechanisms that are associated with a given tax; various flexibility-mechanisms (*e.g.* banking and borrowing provisions) and/or restrictions on trade that are built into tradable permits schemes; and different legal requirements applying to different environmental or economic situations that are built into many types of "command and control" regulations. Each of these "mechanisms" can in principle serve a separate objective, so (consistent with the "Tinbergen Rule") each should probably be counted as a separate "instrument".

The end result is that, although the "Tinbergen Rule" is a very useful starting point for discussion of the policy responses to particular problems, it cannot be used "mechanically" to assess whether too few or too many instruments are being applied in a given situation. In order to "evaluate" a specific instrument mix, it seems more fruitful to undertake a careful analysis of the case at hand, and to try to reach a reasonable judgement on what each of the instruments (be they separate legal entities or individual "mechanisms" within one legal entity) contribute in terms of environmental effectiveness and economic efficiency.

1.4. Policy coherence issues

To increase the coherence of the instrument mix being applied in a given environmental area, different levels of policy co-ordination will be needed. In addition to co-ordination *among environmental policies*, co-ordination *with other related policies* will also be needed, either to address possible negative interactions, or to develop reinforcement mechanisms.

1.4.1. Co-ordination among environmental policies

When addressing a particular environmental problem, it is important to avoid transferring pollution between different environmental media. For example, it could be of little use to limit leakage of pollutants from waste landfills into groundwater if this resulted in increases in air pollution from waste incinerators or composting plants.[3] Similarly, changes in agricultural practices to limit run-off of nutrients could be of little help if they lead to increases in ammonia emissions to air – especially since part of the ammonia will eventually be deposited back onto the ground – and thus exacerbate the original ammonia problem.

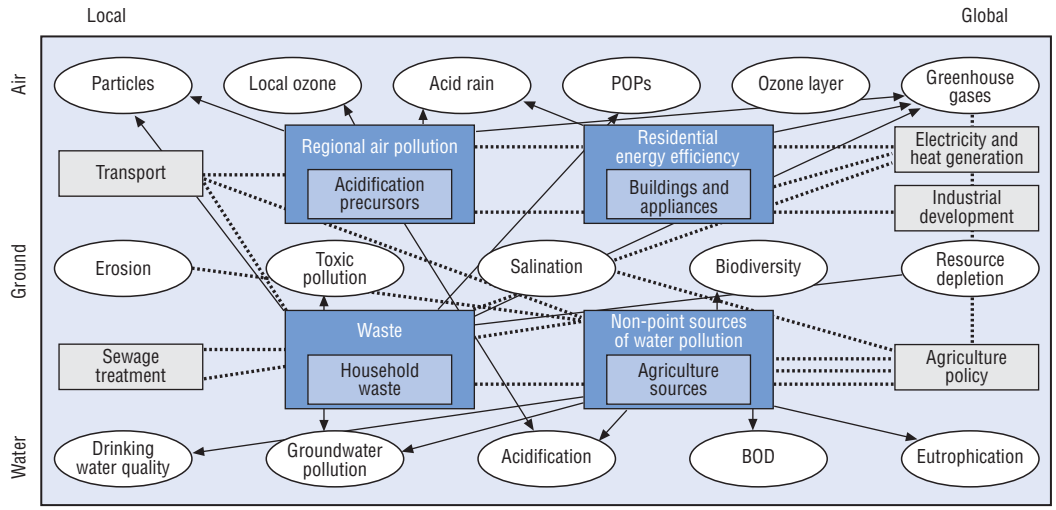
However, the importance of this problem could depend on local circumstances. In some areas, nutrients-related water pollution will represent a more serious environmental problem than ammonia emissions to air. In other areas, increased water pollution could be a major problem – or no real environmental problem at all.

This "site specific" problem reinforces the need for a broad-based cost-benefit analysis (based on a life-cycle approach, and applying monetary values to the anticipated environmental changes), when judging the usefulness of any change in instrument use to address environmental problems that could result in a transfer of pollution from one environmental medium to another.

1.4.2. Co-ordination with other related policies

Figure 1.1 illustrates some of the inter-relations between environmental issues (or "externality categories") and various environmental policy fields. Along the vertical axis, some important environmental issues (represented by ovals on a white background), have been grouped according to whether they (primarily) concern the air, the ground (or soil) or water. (A rough attempt has also been made to place the issues on a local *vs.* global scale along the horizontal axis of the graph, but this "ordering" makes most sense in the context of air-related issues.)

Figure 1.1. **Interrelations between some environmentally related issues and policy areas**



Each oval therefore represents the "negative environmental externalities" that can cause harm to human health, the environment, etc. Ideally, environmental policies should address these negative externalities *as directly as possible*. However, in practice, environmental policies tend to be built up by a large number of different approaches that target different parts of the economy, or different categories of polluters.

Four of the five environmental policy areas addressed in this book (regional air pollution; residential energy efficiency; waste; and non-point sources of water pollution) have been represented in Figure 1.1 as rectangles, and with the particular sub-topics emphasised in those case studies being indicated inside that box. For example, while there are many different non-point sources of water pollution – including *inter alia* road transport – this project only deals directly with (some) agricultural sources.

Figure 1.1 also includes a few other policy areas which have close inter-relations with some of the areas primarily addressed (these are represented by rectangles with a grey background). This illustrates the need to co-ordinate policies and instrument use in the areas addressed in this project with other relevant policies.

The close links between waste policies and policies concerning electricity and heat generation can serve as an example. Several waste treatment techniques can produce electricity and/or heat – which would lower the need to produce energy from other sources, thereby limiting the negative externalities related to those activities. This is most notable concerning waste incinerators, but is also relevant for anaerobic digestion of organic waste, as well as for landfills equipped with landfill gas-collection systems (which produce methane gas that can be burned to produce electricity and/or heat). Recycling of wastes can also generate significant savings in energy use associated with the production of "virgin materials". Such energy savings are frequently found to be among the most important benefits of policies to promote waste avoidance and/or recycling.

There is also an indirect link between energy and waste policies, through differences in emission regulations for waste incinerators and electricity power generation plants. For example, the Protocol on Persistent Organic Pollutants sets a legally-binding limit for the emission of dioxins and furans for municipal waste incinerators. A similar *common* upper limit is not set for emissions of dioxins and furans from *e.g.* electricity power generation plants (*cf.* further discussion in Section 7.13.1).

There are also links between waste policies and non-point sources of water pollution. For example, compost stemming from certain wastes can be a partial substitute for organic or chemical fertilisers. Due to the large manure surplus in the Netherlands, less emphasis is now being placed in that country on promoting waste composting compared to what was the case a few years ago.

Instrument mixes aimed at controlling emissions of mercury to the air also have close links to instrument mixes aimed at pollutants that cause acidification problems, because coal-based power plants are among the most important sources of mercury emissions in many countries. Mercury-related policies are also related to instrument mixes aimed at emissions from waste incinerators, because the latter are important sources of air emissions of mercury.

Co-ordination with other policies is also needed regarding household waste. Among environmental policies, for example, there are the links mentioned above between instruments used to stimulate composting of waste and instruments used to address nutrient run-off. Similarly, instrument mixes aimed at controlling air emissions of mercury need

co-ordination with instruments aimed at recovery of energy in wastes. One co-ordination paradox in the energy area is that emission regulations for several pollutants facing waste incinerators tend to be stricter than emission regulations facing other energy producers.

1.5. Positive interactions among instruments

1.5.1. *Providing information*

Combining information-based instruments (*e.g.* labels) with measures that more directly target the environmental externality (*e.g.* tax; direct regulation) can make both instruments more effective. Labels provide information to the relevant decision-makers (*e.g.* consumers; farmers, industrials) so they can make informed choices, thereby addressing information-related market failures.

For example, the provision of information to consumers can be one argument for policy intervention aimed at improving residential energy efficiency. (The slow adoption of environmentally preferable consumer durables may be due to information failures and search costs.) Combining energy-efficiency labels for appliances with other instruments (*e.g.* a tax on domestic electricity use) can enhance the environmental effectiveness of the other instruments.

However, the importance of information failures and the need to address them are not the same for all environmental issues. Mutual reinforcement among instruments is most likely to be important where there are significant *private benefits* associated with a change in behaviour (*e.g.* buying labelled products), and the existence of these private benefits will vary with the environmental issue being addressed. For instance, users themselves get a direct private benefit from using energy-efficient products (*e.g.* lower operating costs). The value of labelling in the area of residential energy efficiency is primarily to impart information concerning such private benefits. To a lesser extent, there are also private benefits associated with the use of water-efficient products when water use in metered. However, there are hardly any private benefits associated with buying products just because they are part of a recycling scheme. Hence, recycling-related logos are less likely to increase the environmental effectiveness of other instruments in a given instrument mix.

In addition to differences between *environmental issues*, the impacts of instrument combinations will also vary between different *products*. For instance, in the area of energy efficiency, the increase in the relevant price-elasticities due to the introduction of a label is likely to be more significant for the use of some electrical appliances (*e.g.* a television) than for a motor vehicle (for which the energy consumption is likely to be a product characteristic which has already been factored into the initial decision to buy the motor vehicle).

Finally, the environmental effectiveness will vary according to the *type of instrument* to which an eco-label is being attached. The use of information-based instruments in combination with economic instruments designed to internalise the environmental externality (*e.g.* a tax) is likely have a larger impact on environmental effectiveness than if the label is combined with *e.g.* a direct regulation. The degree of choice left by the instrument with which the label is combined is of importance. For example, if an energy-efficiency label is combined with an energy tax (*e.g.* a carbon tax levied on domestic electricity use) to improve residential energy efficiency, the label could increase the effect of the tax – by increasing relevant price-elasticities. In the area of residential energy efficiency, for example, labels can make the introduction of an environmentally related tax on energy more environmentally effective.[4] On the other hand, the introduction of a tax

would often increase the attention decision-makers pay to a related label. But, if an energy-efficiency label is combined with a standard that requires all products to comply with a "strict" energy-efficiency level, it is more difficult to see that the two instruments enhance the performance of the other.

It is also important to note that applying several different labels to a given environmental objective (*e.g.* energy efficiency of appliances) can create confusion in the message being sent to the consumer. It addition, it may entail high administrative costs.

1.5.2. Stimulating innovation

There are also positive externalities related to research and development activities – those who invent seldom reap all the benefits from the innovations they make (although patenting can help them capture a significant share of the benefits). Whereas market-based instruments can provide important incentives for research and innovation, it might also be useful to introduce complementary measures as part of the instrument mix, in order to directly promote technological innovation. In the area of non-point sources of water pollution, the provision of financial support to develop better feedstuffs for animals has, for example, played an important role in addressing nutrient run-off in Denmark and the Netherlands, in combination with their respective nutrients accounting systems, etc.

Providing subsidies for R&D may, however, also undermine the environmental effectiveness and economic efficiency of the instrument mix, if these subsidies are not properly designed and targeted. It is important to ensure that financial support provides an inventive for technological innovations with a *positive return* for the society as a whole, and that these innovations would not have occurred otherwise.

More generally, the efficiency and effectiveness of instrument mixes will depend upon proper targeting of financial support programmes. On the other hand, the better targeting of measures can also entail significant additional administrative costs. There is clearly a trade-off here.

1.5.3. Addressing "split incentives"

An important market failure of relevance for environmental policy is the existence of any *split incentives* – for example, when a landlord would bear costs related to investments in energy efficiency improvements, while the benefits of that investment would accrue to the tenant.[5] In such situations, a combination of instruments could be useful for promoting environmental effectiveness. Measures targeted at private landlords are used to make energy-saving improvements of rental properties more attractive, such as the UK's *Landlord's Energy Saving Allowance*.

The case of household domestic waste offers another example of split incentives. Municipalities pay landfill costs, but it is households that make decisions concerning the amounts of waste that are generated. This particular market failure can be overcome by having households pay waste collection charges that vary with the amount of waste they put out for collection.

1.5.4. Limiting monitoring and enforcement costs

Sometimes it is possible to reduce the total costs of compliance monitoring by adding another instrument to the mix. For example, Dutch authorities found it necessary to maintain the relatively *easily controllable* manure quota system (that limited total manure

generation), when they introduced the MINAS accounting system. It is easier to count the number of animals on a given farm than it is to verify all aspects of the nutrient balance. A similar argument is used for keeping in place quotas for pig and poultry production – in addition to the EU milk quotas – alongside the new application standards that were implemented in the Netherlands in 2006.

1.5.5. *Reducing compliance-cost uncertainty*

A well-known argument for combining a quantity-based instrument (*e.g.* a quota-based tradable permits system) with a price-based instrument (*e.g.* a tax) is that introducing the latter could help limit compliance-cost uncertainty. In effect, instead of paying a very high price for an emission permit, a firm could use the "safety valve" and pay the pre-determined tax or "fee" instead for any emissions for which it does not hold an emission permit.[6]

However, whereas the case studies made in this project have found several examples of quota systems being used in combination with taxes and similar, actual use of a "safety valve" to limit compliance-cost uncertainty has not been found in these cases. See Section 7.4 for further discussion.

1.6. Negative interactions among instruments

1.6.1. *Different approaches at different administrative levels*

The country case studies examined for this report have highlighted the vital role EU legislation plays in most European OECD member countries. Obviously, this necessitates close co-ordination between domestic instrument design with the design of (commonly-agreed) EU Directives and Regulations. For instance, in the area of non-point sources of water pollution, the EU Nitrate Directive requires regulation of nitrogen application at the farm level, while both the Netherlands and Denmark have found that it could be more efficient to apply instruments that address the nutrients problem at either the sectoral or national level.

Similarly, in many OECD member countries, assuring co-ordination of policies implemented at the national, the State, Provincial or "Länder" level *and/or* at the local level can be problematic. This is especially relevant in the waste area, where local authorities are responsible for important services, but responsibility for securing the necessary co-ordination often rests with central authorities.

1.6.2. *Redundancies*

Overlap between policy instruments can also be a problem. For example, the tax rate for "active waste" under the UK *Landfill Tax* and the UK *Landfill Allowance Trading System* applies to almost exactly the same type of "biodegradable waste". See Section 7.12.2 for further discussion.

Again in the household waste area, there is a high degree of overlap between the (almost prohibitive) tax rates of the Landfill Tax, the (almost general) ban on landfilling, and the "minimum standard for waste management" that applies to the environmental permits firms are required to have in order to operate in the Netherlands.

In addition to creating additional administrative costs, instrument overlap can prevent polluters from utilising opportunities for seeking low-cost compliance options offered by an economic instrument.

1.7. Political context

1.7.1. *Constitutional setting*

The institutional context prevailing in a country is likely to affect the nature of the instrument mixes that are applied to a given environmental issue. For example, in Canada, the Constitution assigns responsibility for NO_x and SO_2 emissions at the *facility level* to the Provinces and Territories. However, the federal government has the duty to deal with *transboundary* air pollution, and has jurisdiction over *product policies* in general. Along the same line, Canadian Provinces and Territories have jurisdiction over measures to improve residential energy efficiency, whereas the federal government has responsibility for the inter-provincial trade of energy-using equipment.

Denmark provides another interesting example. In that country, the Constitution requires that all tax-bases and related tax rates be spelled out explicitly in the tax laws. It is therefore not possible to leave the exact classification of different tax-bases to by-laws or regulations (*i.e.* to administrative decisions). This is, for example, one of the factors that have stopped introduction of a tax on pesticides based on the recommended area dose of each product – because such a new tax would have to specify the recommended area dose for hundreds of different pesticides currently available on the Danish market.

1.7.2. *Social policies*

The implementation of theoretically "optimal" or (more realistically) "improved" instrument mixes is often complicated by the "political realities" which exist in particular countries. These may reflect long-standing political compromises, and can clearly represent important constraints on the implementation of new policy instruments.

Social concerns often lie at the root of these constraints. A key point here is to recognise that social concerns do not necessarily have to be addressed through environmental policy – they can, and often should, be addressed through other public policy measures (*e.g.* via social security schemes). When new environmental policies are adjusted in order to address social/distributional policy objectives, policy-makers should ensure that the economic efficiency and environmental effectiveness of the combined policy remain as high as they can be.

In particular, any subsidies that are involved should be – but often are not – targeted to the "right" beneficiaries. For example, making subsidies for energy efficiency retrofitting dependant on income levels can entail significant administrative costs, and also discourage labour-force participation.

1.7.3. *"Status quo bias"*

There is often hesitation among policy-makers to radically change the approach used in a given area, even if it seems likely that such a change could provide net benefits to society. One example comes from Denmark, where several in-depth investigations of possible changes to the instrument mix used to address nutrient run-off indicated that *significant* savings for society as a whole could be achieved by replacing the current (non-tradable) quota system for nutrients applications with a tax on the nitrogen surplus, calculated at the sectoral level. Such a change was, however, not embodied in the latest *Action Plan for the Aquatic Environment* – in part because the prevailing regulatory regime was perceived to be working well, and delivering the expected environmental results. As in sports, there is sometimes resistance to "changing a winning team" in policy-making.

Notes

1. As for the underlying environmental externality, it will sometimes be difficult to address the information failure – or any other relevant market failure – directly. For example, if the relevant information failure involves consumers not being aware of the energy consumption associated with various household appliances, one could make *more* (or *better targeted*) information available through labelling schemes, etc. However, the existence of the labelling scheme will not *guarantee* that the information failure will be overcome – one cannot "force" the relevant information into the brains of all consumers. Hence, there *could* again be a case for applying several instruments to overcome a particular market failure. More specifically, a tax on energy use or a subsidy for the purchase of energy-efficient appliances might make consumers more aware of the labelling scheme, thereby limiting the information failure.
2. Tinbergen (1952) showed that the number of policy instruments needs to be as large as the number of policy objectives involved. Johansen (1977) discussed the "Tinbergen Rule" in relation to economic policy-formulation in general; Tietenberg (1995) discussed the use of tradable permits in situations where the location of emissions "matters".
3. The net cost or benefit to society of such a "pollution transfer" would depend on the magnitude of the changes and on the value society places on the various types of environmental impacts. There can also be equity issues involved, as generally other parts of society (and other ecosystems) will be affected if pollution is transferred from one environmental medium to another.
4. If increased price responsiveness means that a given environmental target can be achieved through a smaller change in relative prices, the economic efficiency of the instrument mix would be enhanced.
5. The importance of this market failure will *inter alia* depend on the length of the rental period. Tenants that expect to stay for a long time in a dwelling do have an incentive to improve the energy efficiency of the house – but in an apartment block, it can be difficult for one tenant to act alone.
6. The other side of this "coin" is – of course – that the uncertainty about achievement of the environmental target increases, unless complementary instruments are put in place.

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Chapter 2

Instrument Mixes Addressing Household Waste Generation

This chapter reviews instrument mixes applied in the Netherlands and in the United Kingdom to address the issue of household waste. In the case of the United Kingdom, attention is focused on the instrument mixes applied in England and in Wales, alongside the nationwide instruments.

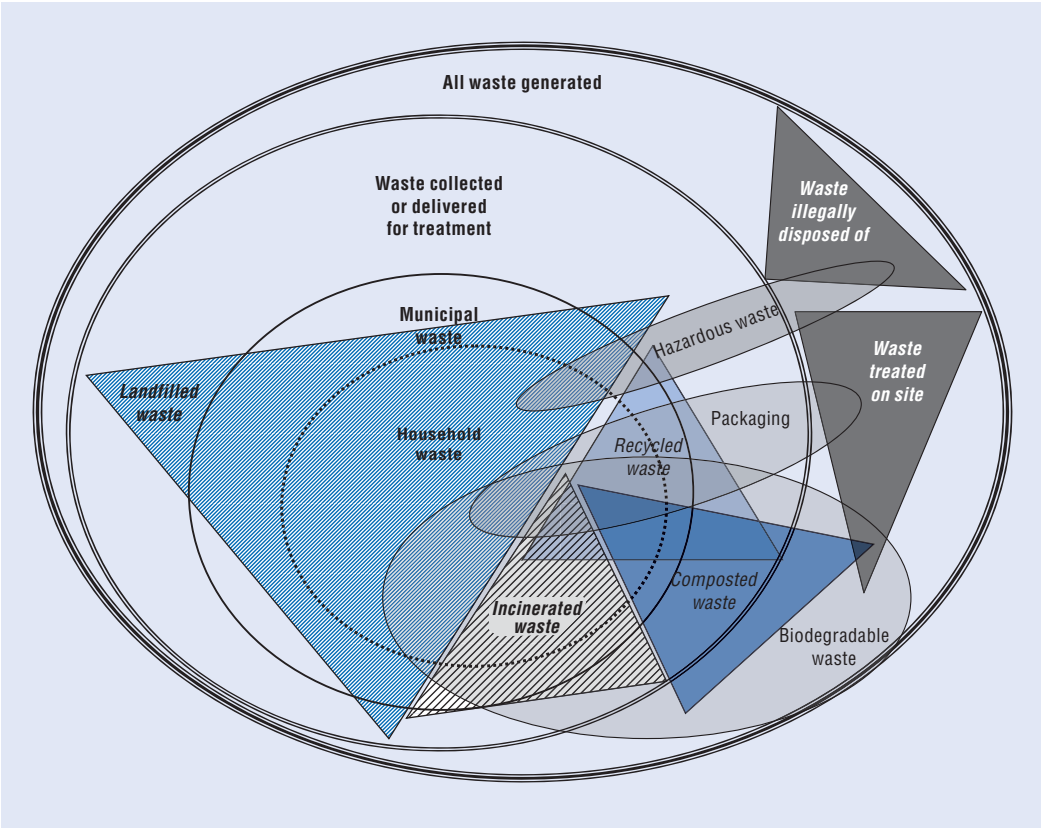
2.1. Introduction

This chapter discusses instrument mixes used to address municipal waste – more specifically, household waste – in OECD member countries. To start off the discussion, an indication is provided of the *complexity* of the municipal waste policy area. An overview of responses to a questionnaire on waste policies that was sent to all OECD member countries is then given, before more in-depth country case studies for the United Kingdom and the Netherlands are presented.

Figure 2.1 gives a highly simplified – but still quite complicated – picture of the waste policy area as a whole. Slightly modified versions of this figure will later be used to illustrate the "domain" of various instruments applied in United Kingdom, described in detail in Section 2.4.

The *circles* in the figure represent different sub-groups of waste. Not all waste is collected or delivered for treatment: significant amounts of waste are *e.g.* treated by industrial firms on site, while large amounts of garden waste only get counted as waste

Figure 2.1. **Waste categories and treatment options**



once they are put out for collection by municipal services or delivered to a drop-off centre – old grass that is left to rot on lawns is not counted; neither is "green waste" which is part of a home composting scheme.

Among the collected or delivered waste, municipal waste only constitutes a part – normally a much smaller part than what Figure 2.1 indicates. Other parts are industrial wastes, construction and demolition waste, sewage sludge, etc. Among municipal wastes, household waste – the prime focus of this study – constitutes a large share – in general, even larger than what Figure 2.1 indicates (about 80% of municipal waste in OECD on average).

The *triangles* in Figure 2.1 represent different ways waste can be treated, such as landfilling, incineration, recycling, composting – and illegal disposal. The relative importance of the different treatment methods varies significantly from country to country (the size of the triangles is not proportional to the use of the various methods).

Many policies in the waste area address particular types of waste (*e.g.* hazardous waste, packaging and biodegradable waste[1]). These particular waste types have been represented by the three *shaded ovals* in the figure.

Contributing to the complexity of Figure 2.1 is the fact that many of the different shapes overlap. For example, composted waste and (most of) incinerated waste is "biodegradable waste", as is a large part of "recycled waste" and a limited share of "landfilled waste". In addition, there is overlap between "packaging" and "biodegradable waste".

2.2. Responses to a questionnaire on instrument mixes addressing household waste

Sixteen countries[2] responded to a 2004 questionnaire, designed to provide an overview over the instrument mixes used to address household waste in OECD member countries.

Table 2.1 shows that 95 *national* policy objectives related to waste *amounts* were identified in the questionnaire responses. The questionnaire focused on certain products that receive particular attention in waste policies. Some specific instruments are in frequent use, such as take-back requirements [often part of "Extended Producer Responsibility" (EPR) systems]; obligatory requirements regarding the use of recycled materials in the manufacture of "new" products; waste-related product taxes and deposit-refund systems. Separate information was also sought regarding policy instruments that member countries believe have special "Design for Environment" impacts.[3]

Table 2.1. **Overview of national and state/provincial policy instruments addressing household waste**

	National level	State/provincial level	Total
Policy objectives related to waste amounts	95	–	95
"Main instruments" to address waste amounts	64	24	88
Policies addressing selected products or waste streams	143	43	186
Take-back requirements	70	25	95
Obligatory requirements on the use of recycled materials	11	0	11
Waste-related product taxes	66	2	68
Deposit-refund systems	11	7	18
Instruments believed to have special "Design for Environment" impacts	38	18	56

Source: Member countries' responses to a questionnaire.

Most of the instruments identified are being applied at the *national* (as opposed to the *state/provincial*) level, in part because the questionnaire only asked for *examples* of the latter category.[4] However, almost 25% of the products or waste streams that are given special attention in waste policies have been singled out at the state/provincial level. These products or waste streams include *e.g.* different types of batteries, packaging, tyres, end-of-life vehicles, "white goods", etc. One can also notice that waste-related product taxes are almost exclusively applied at the national level.

Table 2.2 provides additional information on the reasons for the special attention given to these products and on the types of instruments that are applied to them. (A given product can be addressed by several instruments.) The table details this information across different product categories at both the national and state/provincial levels. One can first notice that packaging is by far the *product group* with the most policy instruments. Responses concerning the *motivations* for addressing different product types do not cover all the instruments detailed in the lower part of the table. One can nevertheless notice relatively clear differences in the motivations for action between the different product categories – with safety and handling problems dominating for instruments addressing batteries and hazardous products, and large volumes being the prime motive for addressing packaging and "other products". It is also striking that information about the "motivation" for about half of the products – among them three-quarters of all the packaging products – is lacking.

Both at the national and the state/provincial levels, *take-back requirements* are the most frequently used instruments to address almost all product types.[5] One exception is – not surprisingly – hazardous products, where *bans* are most commonly used. For packaging, refrigerators, tires, vehicles and waste oil, various product *taxes* also play a relatively important role.

Table 2.3 provides further details on the take-back requirements identified in the questionnaire responses. Such requirements often form part of *Extended Producer Responsibility* (EPR) policies, and address products like packaging, batteries, some hazardous products, end-of-life vehicles, tires, "white goods", etc. Most of the requirements identified through the questionnaire are being applied at the national level, and – where information on the required take-back share is available – close to 60% require that at least 50% of the products should be taken back by the producer (or importer) once the *products'* economic life has come to an end. Most of the schemes identified include an option of voluntary participation in a Producer Responsibility Organisation (PRO), which takes care of the practical collection of the end-of-life products. But almost half of the schemes (also) build on each producer collecting the products they initially placed on the market. Obligatory participation in a PRO is rare.

The take-back schemes are most often financed through either a fee ("Advanced Disposal Fee") on all the items sold in a certain product category (*e.g.* all tires or vehicles sold), or through a fee levied by the PRO on member firms according to their respective *market shares*. Public subsidies also contribute to the financing of about 15% of the schemes, while a PRO fee based on the *recycling costs* of each individual product is much rarer. This is significant for any "Design-for-Environment" impacts of take-back schemes, as it is precisely PRO fees based on actual recycling costs – or a system based on individual collection – that could give producers a strong DfE incentive.[6]

Table 2.2. **Further details on waste policy instruments**

	National level									State/provincial level									Total
	Ba	El	Ha	Pa	Re	Ti	Ve	WO	Ot	Ba	El	Ha	Pa	Re	Ti	Ve	WO	Ot	
Products in total	17	11	13	48	8	8	8	5	25	4[1]	5[1]	5	9	1[1]	3[1]	3	3[1]	10[1]	186
Targeted due to																			
Safety reasons	12	3	10	2	3	1	4	2	3	1	2	3	2	1	1	0	1	0	51
Large volumes	0	5	0	10	3	2	3	1	13	0	1	0	1	1	0	0	0	3	43
Handling problems	6	3	11	2	3	4	2	1	6	1	1	2	1	1	1	0	1	2	48
Information lacking	5	4	1	34	4	4	3	3	9	3	2	2	5	0	2	3	2	6	92
Covered by																			
Bans, *e.g.* on landfilling	1	1	9	0	0	1	1	0	2	1[1]	1[1]	1	0	1[1]	2[1]	0	2[1]	1[1]	24[1]
Take-back requirements	12	9	2	28	5	4	5	2	8	4	4	2	3	1	2	3	2	8	104
Product taxes	8	1	3	26	5	5	4	3	6	1	1	0	0	0	2	0	1	0	66
Deposit-refund systems	2	0	1	7	1	0	1	1	0	0	0	0	2	0	0	0	0	0	15
Information instruments	4	2	1	3	0	1	1	1	4	1	1	2	1	1	1	0	1	4	29
Voluntary approaches	1	2	1	3	0	1	0	0	5	2	1	4	2	1	1[1]	0	2	2	28[1]
Other instruments	0	0	0	2	0	0	0	0	3	0	0	1	2	1	0	0	0	0	9

Explanation: Ba: Batteries; **El:** Electronics; **Ha:** Hazardous products; **Pa:** Packaging; **Re:** Refrigerators; **Ti:** Tires; **Ve:** Vehicles; **WO:** Waste oil; **Ot:** Other products.

1. There is an element of arbitrariness in the counting method. The US response, which for each product category here is counted as *one* single entry, explained that there are *bans on landfilling* of lead-acid batteries in 29 separate States, of electronics or cathode ray tubes in 4 States, of white goods in 17 States, of whole tires in 30 States, used oil in 21 States and of yard trimmings in 21 States. In addition, all 50 States have voluntary approaches as concerns whole tires.

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Table 2.3. **Further details on take-back requirements**

	National level									State/provincial level									Total
	Ba	El	Ha	Pa	Re	Ti	Ve	WO	Ot	Ba	El	Ha	Pa	Re	Ti	Ve	WO	Ot	
Take-back requirements	10	6	2	27	6	4	4	2	9	3	3	2	3	0	2	3	2	7	95
With a required take-back share																			
Larger than 80%	3	1	1	3	0	1	1	0	0	0	0	0	0	0	0	0	0	0	10
Between 50 and 80%	3	2	0	10	2	3	2	2	1	0	0	0	2	0	0	0	0	0	27
Lower than 50%	2	2	0	5	2	0	0	0	5	0	1	0	0	0	0	0	0	0	17
No explicit percentage	1	0	1	0	1	0	1	0	3	0	0	0	0	0	0	2	0	0	9
Information lacking	1	1	0	9	1	0	0	0	0	3	2	2	1	0	2	1	2	7	32
Organisation of the collection system																			
Individual collection	6	4	1	12	4	3	2	1	4	2	1	1	1	0	0	0	0	0	42
Voluntary PRO	8	5	1	25	5	4	2	2	7	0	1	1	3	0	1	0	1	0	66
Obligatory PRO	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	2	0	0	5
Information lacking	1	0	0	0	0	0	0	0	0	1	2	1	0	0	1	1	1	7	15
Financing of the collection system																			
No separate financing	1	1	0	3	0	0	2	0	3	1	0	1	1	0	1	2	1	0	17
A fee on all products	4	3	1	9	2	3	3	1	4	1	0	0	0	0	0	0	0	0	31
PRO fee: market shares	5	4	0	10	4	2	0	2	3	0	1	0	0	0	0	2	0	0	33
PRO fee: recycling costs	1	0	1	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	5
Public subsidies	3	1	0	2	2	1	1	1	2	0	0	0	0	0	0	0	0	0	13
Other financing	1	0	1	1	1	0	0	0	0	0	0	0	2	0	0	0	0	0	6
Information lacking	0	0	0	7	0	0	0	0	0	1	2	1	0	0	1	1	1	7	21

The waste-related product taxes identified through the questionnaire responses cover a broad range of products. Examples of such taxes include taxes on various containers and other types of packaging, disposable diapers, car batteries, tires, refrigerators, etc. Compared to the number of products identified, taxes are widely used to address refrigerators, tires, vehicles, waste oil and packaging.

Most of the deposit-refund systems identified address some type of packaging. In several cases, more than 90% of the packaging concerned are returned for reuse or for recovery.

How local authorities charge households for the waste collection services can have important impacts on the amounts of waste generated – and for the incentives transmitted to producers to put products on the market that generate less waste.[7] Table 2.4 summarises how waste collection charges are calculated in countries that responded to the questionnaire. The main message is that of great diversity in the charging methods. In about one-quarter of the countries, it is estimated that 80% or more of the households are charged according to the weight or volume of the waste being collected. Such a basis is in use for *some* part of the households in about three-quarters of all the countries that have responded. In one country, the charging is mostly based on the number of persons in the households, while the size of the house or apartment serves as a basis of charging – in a few cases in two other countries.[8]

Table 2.4. **Basis of municipal waste collection charges**

Charging according to	> 80%	60-80%	40-60%	20-40%	< 20%
Weight or volume of the waste	5	3	1	1	3
The number of persons in the household	1	0	2	0	1
The size of the house/apartment	0	0	0	1	1
Other basis for separate charges	0	1	1	2	4
No separate charge, integrated in local taxes	3	0	0	4	0

2.3. EU Directives addressing household waste

Before describing in some detail the instrument mixes addressing household waste in the United Kingdom and the Netherlands, this section briefly describes some EU Directives addressing household waste, and which both of these countries are obliged to implement and comply with.

2.3.1. The Landfill Directive

The *Directive on Landfill of Waste* [see CEU (1999)] aims to prevent or reduce as far as possible negative effects on the environment, and any resulting risk to human health, from landfilling of waste, during the whole life-cycle of the landfill. In this regard, the Directive requires member States to set up a national strategy to reduce the amount of biodegradable waste going to landfill, and sets targets for the reductions member States should achieve.

Most member States were required to reduce the absolute amount of biodegradable municipal waste landfilled to 75% of their 1995 level by 2006, to 50% of the 1995 level by 2009, and to 35% of the 1995 level by 2016.

Due to a historically larger reliance on landfilling in the United Kingdom than in many other EU member States, that country obtained a 4-year derogation from these targets. Hence, the Directive requires the UK volume of biodegradable municipal waste sent to landfill to be reduced to 75% of the 1995 arisings by 2010; 50% by 2013; and 35% by 2020.

The Directive also prescribes a number of technical requirements for the operation of landfills to reduce their environmental impacts. These include acceptance criteria for waste in various types of landfills, groundwater protection and collection and treatment of landfill gas.

2.3.2. The Incineration Directive

The aim of the *Incineration Directive* [see EU (2000b)] is to "prevent or to limit as far as practicable negative effects on the environment, in particular pollution by emissions into air, soil, surface water and groundwater, and the resulting risks to human health, from the incineration and co-incineration of waste".

With this objective in mind, the Directive spells out a number procedures and emission standards for waste incinerators, *inter alia* as regards SO_2, NO_x, dioxins, mercury and other heavy metals, etc.

2.3.3. The Packaging and Packaging Waste Directive

The *Directive on Packaging and Packaging Waste* [see EU (2004)] is also of great relevance for EU member state's policies addressing household waste. The Directive states, *inter alia*, that:

- "no later than 31 December 2008 60% as a minimum by weight of packaging waste will be recovered or incinerated at waste incineration plants with energy recovery;…
- no later than 31 December 2008 between 55% as a minimum and 80% as a maximum by weight of packaging waste will be recycled;…
- no later than 31 December 2008 the following minimum recycling targets for materials contained in packaging waste will be attained:
  - 60% by weight for glass;
  - 60% by weight for paper and board;
  - 50% by weight for metals;
  - 22.5% by weight for plastics, counting exclusively material that is recycled back into plastics;
  - 15% by weight for wood."

2.3.4. The Framework Directive on Waste

The *Framework Directive on Waste* [CEC (1975)], *inter alia* provides definitions of terms used in other EU waste directives, etc. It also obliges member states to apply the so-called "waste hierarchy", in stating that:

"Member States shall take appropriate measures to encourage:

a) firstly, the prevention or reduction of waste production and its harmfulness, in particular by:
   - the development of clean technologies more sparing in their use of natural resources;
   - the technical development and marketing of products designed so as to make no contribution or to make the smallest possible contribution, by the nature of their manufacture, use or final disposal, to increasing the amount or harmfulness of waste and pollution hazards;
   - the development of appropriate techniques for the final disposal of dangerous substances contained in waste destined for recovery;

b) secondly:

1. the recovery of waste by means of recycling, re-use or reclamation or any other process with a view to extracting secondary raw materials; or
2. the use of waste as a source of energy."

The Directive also requires member States to draw-up waste management plans.

2.3.5. Other EU Directives of importance for household waste management

A number of other EU Directives can also be important for EU member states' instrument mixes addressing household waste. These include:

- Council Directive of 18 March 1991 on batteries and accumulators containing certain dangerous substances, *cf.* CEU (1991).
- Directive 2000/53/EC of the European Parliament and of the Council of 18 September 2000 on end-of life vehicles, *cf.* EU (2000c).
- Directive 2002/96/EC of the European Parliament and of the Council of 27 January 2003 on waste electrical and electronic equipment, *cf.* EU (2002b).
- Directive 2002/95/EC of the European Parliament and of the Council of 27 January 2003 on the restriction of the use of certain hazardous substances in electrical and electronic equipment, *cf.* EU (2002c).

2.4. Instruments addressing household waste in the United Kingdom

2.4.1. Introduction

This section *describes* the instrument mix used to address household waste in the United Kingdom. For a further *discussion* of the mix applied, see Chapter 7. The United Kingdom was chosen as a country case study for a number of reasons:

a) It has an interesting mix of instruments already in use.

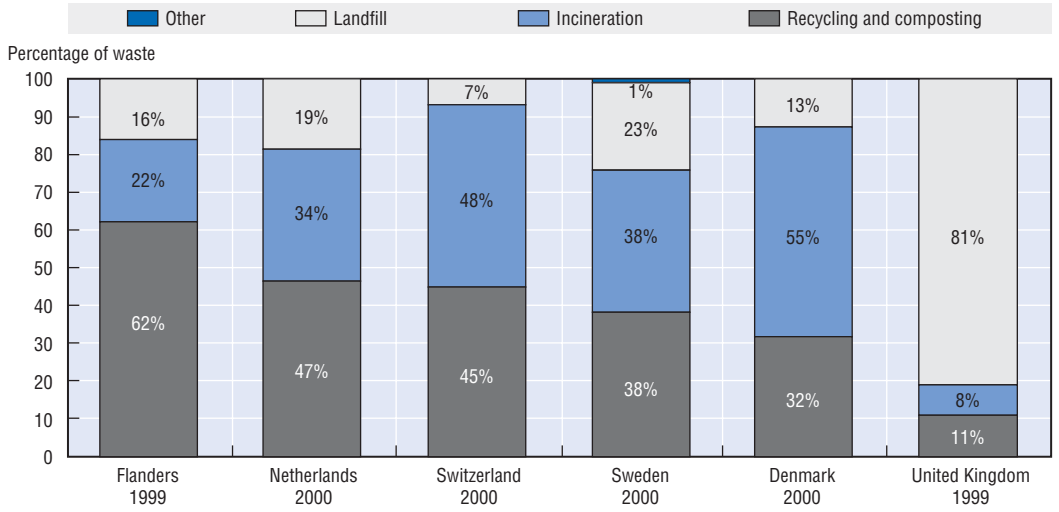
b) Many studies have been made – by both official sources and independent researchers – on the need for policy intervention, on the environmental impacts of various waste treatment options, on the behavioural impacts of certain policy instruments, and on the impacts on income distribution of certain policy instruments.

c) Waste management is a "devolved issue" – meaning that local administrations in Scotland, Wales and Northern Ireland have specific independent responsibilities and authorities in this area. The country, hence, provides an interesting example of the need for co-ordination between administrations at different geographical levels – and of some of the problems related to this.

An important starting point is recognising that implementation and funding of municipal waste management are the responsibility of *local authorities*.[9] Figure 2.2 illustrates existing methods of managing municipal waste in a few European countries – and shows that landfilling traditionally has played a much more dominant role in the United Kingdom than in many other European countries. With long-term growth in the amount of municipal waste estimated to be about 3%, unless waste management practices changed, there would be a widening gap between practice and the United Kingdom's legal obligations under the EU Landfill Directive.[10]

This was the point of departure for a report prepared by the Cabinet Office's Strategy Unit in 2002.[11] That report emphasised waste minimisation and recycling, and set out key measures needed to achieve the goals, focusing on an enhanced role for economic

Figure 2.2. **Methods of managing municipal waste in a few European countries**



Source: Strategy Unit (2002), based on Green Alliance "Creative policy packages for waste: lessons for the UK", autumn 2002. UK figures are for England and Wales only. Figures for municipal waste for the Netherlands and Denmark were constructed by the Green Alliance.

instruments, new investments and reformed delivery structures. The proposed measures included a significant increase in the Landfill Tax over the medium term, as well as increased freedom for local authorities to develop new financial incentives for households to reduce and recycle their wastes.[12]

The report also gave some support to increased emphasis on incineration as a treatment method for municipal waste. Public opinion is relatively negative toward this method, but a study undertaken by MORI, *cf.* MORI (2002), indicated that awareness of incineration as a waste management option is very low, To address the lack of awareness, Strategy Unit (2002) recommended that "an independent body should bring together the literature and evidence on the relative health and environmental effects of all the different waste management options; relative both to each other and to other activities affecting health and the environment".

DEFRA (2004a) provides a response to this recommendation. DEFRA (2004b) is an extended summary of the report. The report was based on an extensive review of available literature relating to waste management activities.

The aim of the analysis undertaken in DEFRA (2004b) was to find a figure for emissions of Pollutant X per tonne of waste processed for the different waste management options. The focus on amounts of waste *processed* means that environmental benefits of waste amounts *avoided* through some waste policies are *not* reflected in the numbers presented.

On a national scale, and taking into account the amount of municipal waste managed by each process, emissions to air from waste management were estimated to result in approximately five hospital admissions for respiratory disease per year, and one death brought forward due to air emission per year in the UK as a whole. Emissions to air from waste management are forecast to result in a much lower increase in the incidence of cancers – only about one additional case every five hundred years.

There are many possible effects of municipal waste management operations on the environment, including: noise; odour; dust; visual intrusion; damage to plant and animals; damage to soils; pollution of water; pollution of air; emissions of global warming gases; and

damage to buildings from acidic gas. Of these, the study found that probably the most important impacts on a national scale are emissions of methane from landfill sites (which affect global warming). Odours from landfill sites can also be important. The report did not find much evidence for other adverse environmental effects due to waste management. Figure 2.3 gives an overview of the main environmental impacts of waste management facilities. It shows that landfill and incineration are the waste management options with most potential for environmental effects.

Figure 2.3. **Summary of environmental impacts of different waste management options**

Category	Meaning
✓	Direct or indirect benefit
–	No effect
x	Unlikely to be significant
xx	Potentially significant impact in some cases, but can be controlled
xxx	Impact can normally be controlled, but an issue at sites if design, engineering or operation falls below best practice
xxxx	An issue at all sites

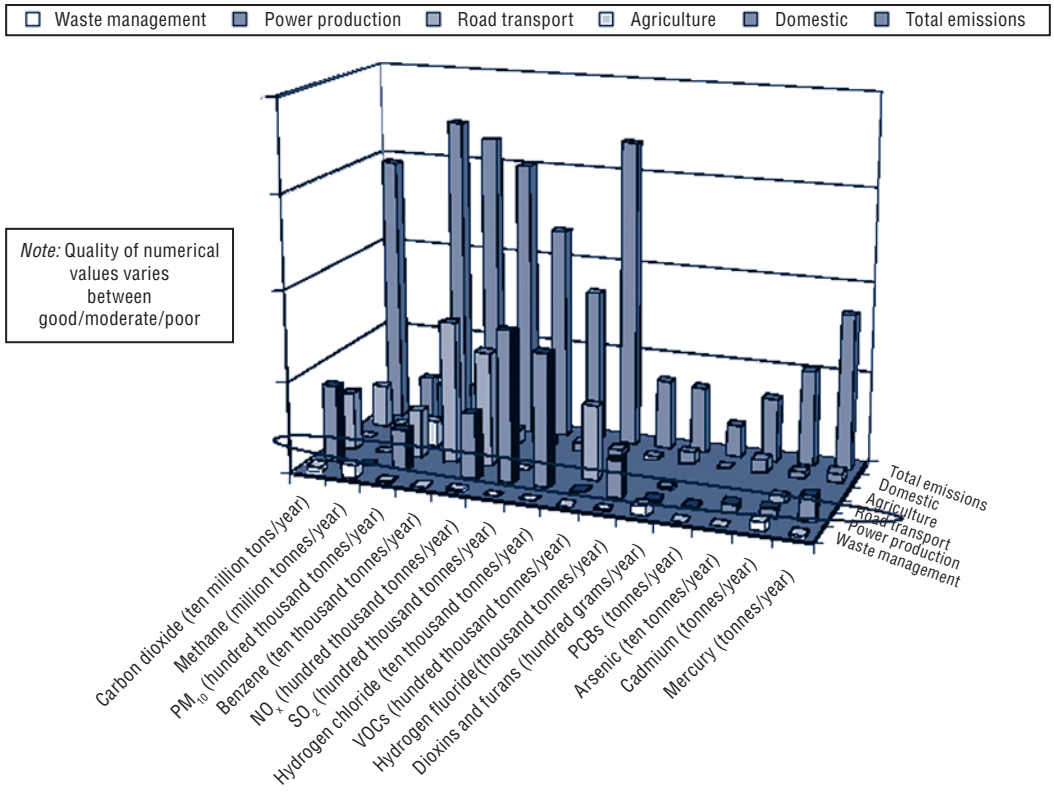
Activity	Noise	Odour	Dust	Flora/ fauna	Soils	Water quality/flow	Air quality	Climate	Building damage
Materials recycling facility	x	x	x	x	x	xx	xx	–	–
Composting	xx	xxx	xx	✓	x ✓	xx	xxx	x	–
Mechanical biological treatment	xx	xxx	xx	–	–	xx	xx	x	x
Anaerobic digestion	xx	xx	x	x ✓	x ✓	xx	xx	x	x
Gasification/pyrolysis	xx	xx	xx	–	–	–	xx	x	x
Incineration with pre-sorting	xx	xx	xxx	xx	xx	xx	xxx	x	x
Incineration	xx	xx	xxx	xxx	xxx	xxx	xxx	x	x
Landfill	xxx	xxx	xx	xxx ✓	xxx	xxx	xxx	xxxx	x
Waste transfer stations	xx	xxx	x	–	–	xx	x	✓	–

Source: DEFRA (2004b).

DEFRA (2004b) also sought to place emissions from waste management into a broader context. Figure 2.4 shows emissions of different pollutants from a few major sources in the United Kingdom – and indicates that for most pollutants municipal waste management is only a minor contributor. The exceptions to this are emissions to air of methane (where waste management accounted for nearly 30% of the UK total) and cadmium (10% of the national total). Almost all of the cadmium emitted to air from facilities dealing with municipal waste in the UK comes from landfill sites.

DEFRA (2004c) provides an economic valuation of the external costs and benefits to health and environment of waste management options, largely based on the scientific findings of DEFRA (2004a). The report is mostly built on a survey and critique of existing valuation studies, and the focus is on landfilling and incineration.

Figure 2.4. **Emissions to air from municipal waste management and other sources in United Kingdom**



Source: DEFRA (2004b).

HM Customs and Excise (2004) brings together the findings of DEFRA (2004a) and DEFRA (2004c). The results were summarised as follows:

- "the studies conclude that the treatment of municipal solid waste has at most a very minor effect on health and find no evidence of a link between modern incinerators and health impacts;
- there are significant uncertainties involved in measuring and valuing the environmental effects of different [municipal] waste management options. This means that estimates of the external costs of landfill and incineration have wide margins of error attached to them;
- against this background of considerable uncertainty, the results suggest that, in the central case, the external [cost] of landfill may be around GBP 10 per tonne of municipal solid waste and the external costs of incineration (with energy recovery) may be around GBP 13 to GBP 14 per tonne of municipal solid waste;
- sensitivity analysis around the central case results shows that the range of possible externality estimates is large. For example, varying the emissions levels from landfill sites and incinerators, and holding all other factors constant, gives a range of external costs of GBP 5 to GBP 20 per tonne for landfill and GBP 7 to GBP 21 per tonne for incineration;
- the external cost estimates for landfill are also sensitive to assumptions about the amount of methane escaping from landfill sites into the atmosphere. Varying these 'fugitive' methane releases in the central case shows that the landfill externality could be closer to GBP 15 per tonne and above the incineration externality; and

- assumptions about the benefit of displaced power generation by incineration (with energy recovery) and landfill engine (where landfill gases are used to generate electricity) also affect the externality estimates. For example, if the displaced power is assumed to be from a coal plant then, in the central case, the incineration externality is smaller than the landfill externalities."

2.4.2. Instruments addressing household waste applying across all of the United Kingdom

2.4.2.1. The Landfill Tax

One of the most important instruments addressing the amounts of all waste landfilled in all of United Kingdom is the Landfill Tax, first introduced in 1996. It applies equally to commercial and industrial waste and municipal waste, and there are separate tax rates for "active" and "inactive" waste.[13] The tax rate for inactive waste has remained at GBP 2 per tonne,[14] while the rate for active waste has been increased several times from an original level of GBP 7, reaching GBP 24 per tonne on 1 April 2007. The tax rate for active waste is set to increase further by at least GBP 3 per tonne per year, on the way to a medium-to-long term rate of GBP 35 per tonne.[15]

The tax was originally designed to internalize the negative externalities of landfill, based on estimates made by CSERGE Warren Spring Laboratory and EFTEC (1993) [see Davies and Double (2004) for an in-depth discussion]. Table 2.5 presents the different components of the estimated externalities. These exclude disamenity costs (that is, the nuisance value from landfill sites from noise, odour, visual intrusion, etc.) The CSERGE report reviewed the methods by which disamenity values can be estimated, principally by analysing the variation in property prices which can be attributed to the proximity of a facility; or by surveys of willingness-to-pay for (or to avoid) a facility being located at a specific site. All of the evidence was North American. Although clearly a second-best approach, the US "willingness-to-pay" estimates were then transferred to the UK landfill context. The use of US figures suggested willingness-to-pay estimates of GBP 160 per household (located within 4 miles of a site) per year for landfill sites. That gave an estimate of disamenity value of approximately GBP 2 per tonne of waste.[16]

Adding together the disamenity and non-disamenity externalities gave a total monetised cost for landfill of about GBP 5 per tonne, approximately equivalent to GBP 7 per tonne for active and GBP 2 per tonne for inactive waste.[17]

Returning to a slightly modified version of Figure 2.1, Figure 2.5 illustrates the "domain" of the UK Landfill Tax.[18] The tax applies to – and seeks to limit the size of – the whole area with thick vertical or horizontal lines. The higher tax rate of (at present) GBP 21 for "active" waste applies to the area with thick horizontal lines, while the lower tax rate of GBP 2 applies to the area with thick vertical lines.

2.4.2.2. (Non-) payment for municipal waste collection services

While many of the waste generators outside the household sector face a positive marginal cost per tonne of waste they generate – and hence are directly affected by the scheduled increases in the Landfill Tax rate – households in United Kingdom are *not* paying *separate* charges for municipal waste collection services. Instead, the costs of providing these services are (in part) covered by the Council Tax (and in part by various grants from central government – in turn financed through a number of different taxes, etc.).

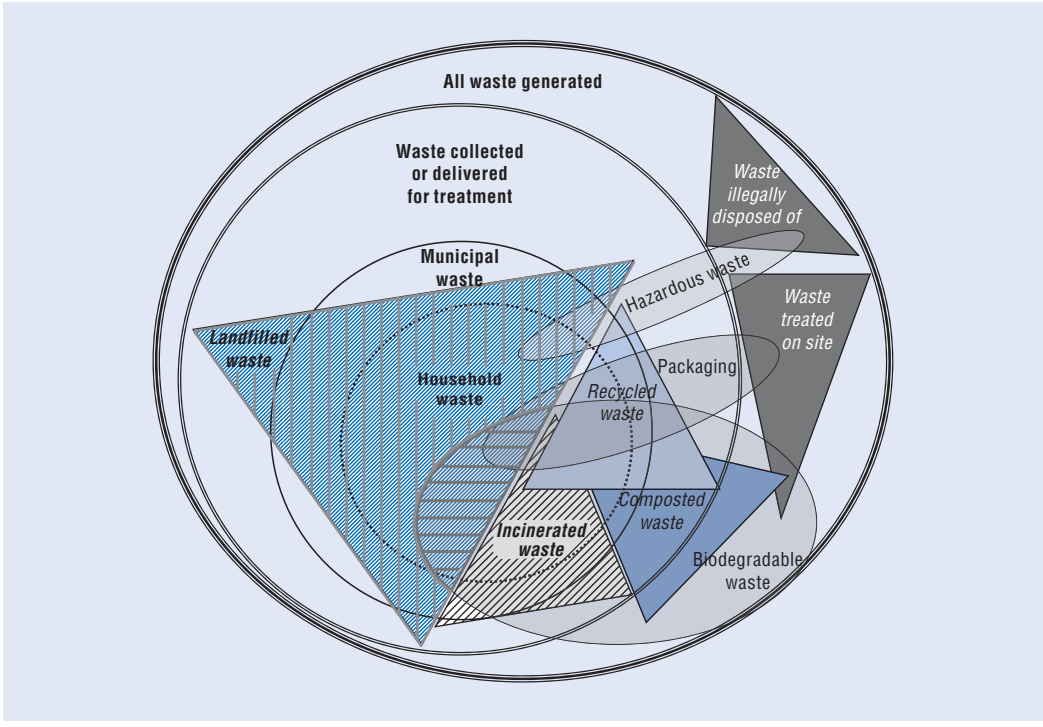
Table 2.5. **Estimated values of landfill externalities in United Kingdom**

GBP per tonne

			Existing urban landfill without energy recovery	New urban landfill with energy recovery	Rural landfill without energy recovery	New rural landfill with energy recovery
+	**Global Pollution**	CO_2 as C	0.32 (0.08-0.87)	0.46 (0.12-1.27)	0.32 (0.08-0.87)	0.46 (0.12-1.27)
		CH_4	2.36 (0.86-5.40)	1.36 (0.45-3.32)	2.36 (0.86-5.40)	1.36 (0.45-3.32)
+	**Air pollution**		n.a.	n.a.	n.a.	n.a.
+	**Transport impacts**	*Pollution – Conventional (UK only)*	0.09 (0.05-0.16)	0.09 (0.05-0.16)	0.38 (0.10-1.06)	0.38 (0.10-1.06)
		Pollution – Conventional (UK and ECE)	0.10 (0.06-0.17)	0.10 (0.06-0.17)	0.46 (0.14-1.19)	0.46 (0.14-1.19)
		Accidents	0.23 (0.13-0.33)	0.23 (0.13-0.33)	0.55 (0.31-0.79)	0.55 (0.13-0.79)
+	**Leachate**		0.45 (0-0.9)	0	0.45 (0-0.9)	0
–	**Pollution displacement**	*Conventional (UK only)*	0	0.81 (1.54-0.45)	0	0.81 (1.54-0.45)
	Pollution displacement	*Conventional (UK and ECE)*	0	1.12 (1.92-0.69)	0	1.12 (1.92-0.69)
=	**Total**	*Conventional (UK only)*	1.12 to 7.66 (3.45)	–0.80 to 4.63 (1.33)	1.35 to 9.02 (4.06)	–0.57 to 6.00 (1.94)
=	**Total**	*Conventional (UK and ECE)*	1.13 to 7.66 (3.45)	–1.17 to 4.91 (1.03)	–1.58 to 9.15 (4.14)	–0.91 to 5.89 (1.72)

Source: From CSERGE Warren Spring Laboratory and EFTEC (1993). The mean value shown for the total, and for pollution displacement, transport and global impacts reflect specific statistical techniques used to capture the uncertainty in CH_4 and CO_2 estimates. The mean does not therefore equal the midpoint of the range values.

Figure 2.5. **The "domain" of the Landfill Tax in United Kingdom**



"Domain" of the tax rate for "active" waste.

"Domain" of the tax rate for "inactive" waste.

The *Environmental Protection Act* of 1990 makes it a duty of the waste collection authorities to arrange for the collection of household waste in their areas. Section 45 (3) states that "No charge shall be made for the collection of household waste except in cases prescribed in regulations made by the Secretary of State" [see EUNOMIA (2003) and Section 7.8.2 for further discussion].

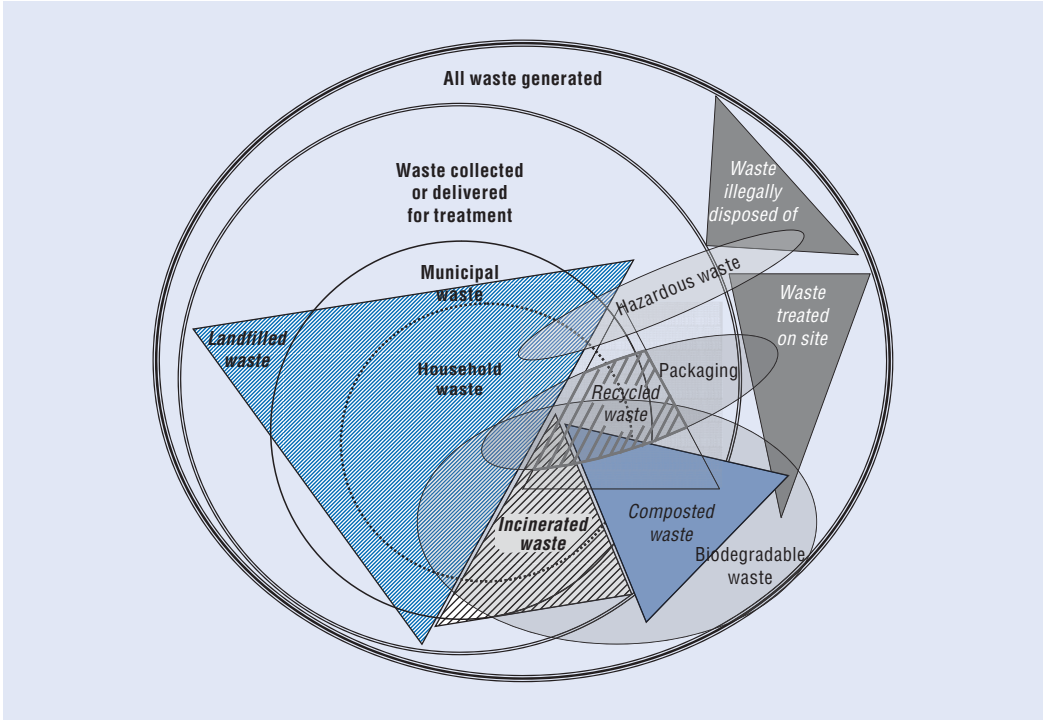
2.4.2.3. Producer requirements

In line with the relevant EU Directives, producer requirements have been introduced across the United Kingdom,[19] for example for packaging and packaging waste [*cf.* EU (1994) and (2004) and Section 2.3.3]. Any business with a turnover of more than GBP 2 million and handling more than 50 tonnes of packaging in a year is "obligated" by the Regulations.[20]

DEFRA *et al.* (2003) discusses in detail the "translation" of the national targets of the Directive into targets for obligated businesses. For example, the 50% recovery target for the UK as a whole was to be met by a 59% recovery achieved by each obligated firm. In 2001, against the first Directive target, the UK achieved the overall recycling and the material-specific recycling targets, but fell short of the 50% recovery target – achieving instead 48% recovery.

Returning once again to a variant of Figure 2.1, Figure 2.6 illustrates the "domain" of the packaging producer requirements – in the sense that the requirements states that the area with thick diagonal lines should cover a specified share of the underlying oval that represents all packaging. In practice, most of the packaging is not collected from households, as post-consumer packaging is dispersed, composed of many different materials and often dirty [*cf.* Forum for the Future (2004)].

Figure 2.6. **The "domain" of producer requirements concerning packaging in United Kingdom**



"Domain" of the producer packaging requirements.

Producers may meet their obligations under the *Producer Responsibility Obligations (Packaging Waste) Regulations* individually or choose to join a compliance scheme (a PRO) that will meet the obligations on the producer's behalf. There are currently 21 registered compliance schemes in the UK which operate in a competitive market and which, between them, have attracted 85% of the producers, according to DEFRA *et al.* (2003).

2.4.2.4. Packaging Recovery Notes[21]

The 1994 EU Directive on Packaging and Packaging Waste [*cf.* EU (1994)], was transposed into the UK legislation in March 1997 by the Packaging Regulations [see HMSO (1997)], which set a series annual targets for the amount of packaging that was to be recovered and recycled from 1998 onwards. The implementation of these regulations represented the culmination of a four-year process that had started with the UK Government's challenge to the packaging chain to set up a voluntary scheme to recover packaging waste. Following the sector's rejection of the voluntary approach in 1994, the Government embarked on a lengthy consultation process with representatives of the packaging chain (and other interested parties), before publishing draft Regulations in July 1996.

During this process, discussions focussed on two main issues – the level of the recovery and recycling targets, and who would be responsible for meeting the targets (*i.e.* whether the obligation would be imposed on a particular sector within the chain, or shared among all sectors). Much less attention was paid to *how* the targets would be achieved. In particular, no consideration was given to the potential for using a system of tradable credits to implement the targets, and the Packaging Regulations made no provision for the introduction of such a scheme. While there was a general acceptance of the advantages of a market-led approach, it was envisaged that this would be achieved largely through a system of collective compliance schemes, which would use the fees that they received from their members to fund the necessary expansion of collection and reprocessing activities – similarly to the PRO systems described in Section 2.2.[22]

The later emergence of a tradable credit system can be traced back to two separate initiatives. In July 1996 (*i.e.* at the same time as the draft Regulations were being published), the largest compliance scheme (Valpak) announced that – rather than arranging directly for the collection of waste packaging – it would contract out this responsibility to the material reprocessors. Contracts would be signed with reprocessors to recycle a specified amount of material. Once this had been done, each reprocessor would send Valpak a "certificate of confirmation". Up until that point, it had been implicitly assumed that compliance schemes would be responsible both for the financing of waste collection activities, and for arranging for those activities to be undertaken. The Valpak decision had the affect of de-coupling these two responsibilities.

The Packaging Regulations specified that proper evidence would be required to prove that waste packaging had actually been recovered, but they left the precise definition of what would be considered "acceptable" to subsequent statutory guidance from the Environment Agency. When this was published in July 1997, it introduced a standardised form of evidence – called Packaging Waste Recovery Notes (PRNs). The main motivations for the introduction of the scheme were the deterrence of fraud by reprocessors, and the reduction of the administrative burden on the obligated businesses. However, by creating a transparent "common currency", the introduction of PRNs facilitated the evolution of the reprocessor contracts introduced by Valpak into a tradable compliance credit system.

The PRNs enable producers (or compliance schemes, like a PRO) to obtain the necessary documentary evidence to show that they have had the specified tonnages of packaging waste recovered or recycled, according to their particular obligation. At the same time, a voluntary accreditation scheme for waste reprocessors was created, to give obligated businesses confidence that, when claiming a certain level of recycling, the recycling had actually occurred. Accreditation is available to any site that is carrying out a valid recovery activity on packaging waste and have an adequate documentation system is in place.

Only reprocessors who have been accredited can issue PRNs for packaging waste delivered for reprocessing. Reprocessors can only issue PRNs to obligated businesses or to their representatives, but the PRNs *can be traded*, similarly to allowances in an emissions trading scheme.

Reprocessors issuing PRNs receive a revenue stream. One condition of accreditation is that reprocessors are intended to use this revenue to fund the recovery infrastructure; it is not intended for use as additional profit to reprocessors. Reprocessors are expected to direct resources at the development of the collection, reprocessing or end-use market infrastructure. See DEFRA *et al.* (2003) and Section 7.12.3 for a discussion.

2.4.3. Instruments addressing household waste in England

2.4.3.1. Statutory Landfill Diversion Targets

As part of UK's measures to comply with the EU Landfill Directive, the *Waste and Emissions Trading Act 2003* [HMSO (2003a)] says that: "The Secretary of State must by regulations specify the maximum amount by weight of biodegradable municipal waste allowed, in each scheme year that is a target year, to be sent to landfills from each of:

a) United Kingdom;

b) England;

c) Scotland;

d) Wales; and

e) Northern Ireland."

Table 2.6 shows the distribution of these allowances between the different authorities, with – naturally – most of the total being allocated to England.

Table 2.6. **Allocation of targets for landfilling of biodegradable municipal waste**

United Kingdom – Maximum amounts, in tonnes

Area	Year ending in 2010	Year ending in 2013	Year ending in 2020
United Kingdom	13 700 000	9 130 000	6 390 000
England	11 200 000	7 460 000	5 220 000
Scotland	1 320 000	880 000	620 000
Wales	710 000	470 000	330 000
Northern Ireland	470 000	320 000	220 000

Source: HMSO (2004).

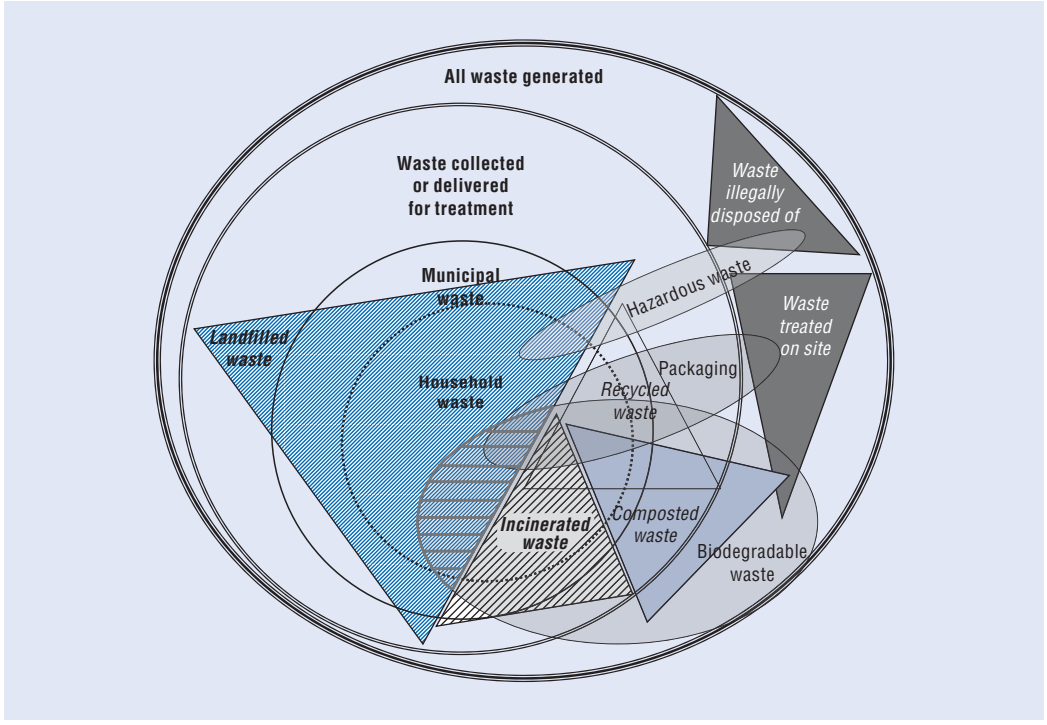
It is up to each of the administrations [*b)* to *e)*] to decide on the allocation of their total landfilling allowance between the different waste disposal authorities. In England, the allocation took place through the setting of *statuary Landfill Diversion Targets* for each waste disposal authority. Each waste disposal authority has been allocated provisional allowances (representing tonnes of municipal biodegradable waste that may be landfilled)

for every year up to 2020 at once – at a declining rate. For the first year of the system, running from 1 April 2005 to 31 March 2006, the total "cap" on landfill allowances for biodegradable municipal waste in England was set to 15.2 million tonnes. For subsequent years, the "cap" declines to 14.53, 13.64 and 12.53 million tonnes respectively, before reaching 11.2 million tonnes in 2009/10 – as shown in Table 2.6.

The number of allowances allocated to each waste disposal authority was based on biodegradable municipal waste landfilled in 2001.[23]

In Figure 2.7, the "domain" of the *Landfill Diversion Targets* is illustrated by the area with thick horizontal lines. The instrument seeks to limit the size of this area, assuring that the share of all biodegradable municipal waste that is being landfilled stays within the limits set in the EU Landfill Directive.[24] One can notice that this "domain" is exactly the same as the "domain" of the tax rate for "active waste" in the *Landfill Tax* (*cf.* Figure 2.5).

Figure 2.7. **The "domain" of the landfill diversion targets in England**



"Domain" of the landfill diversion targets.

2.4.3.2. *Landfill Allowance Trading Scheme*

In England, the statutory *Landfill Diversion Targets* are closely linked with the *Landfill Allowance Trading Scheme* (LATS). Each Waste Disposal Authority (WDA) can use its allocation of allowances in the most effective way for them. They can save unused allowances for use in later years (banking), or use a proportion of their future allocation of allowances in advance (borrowing). Banking or borrowing into a target year is, however, not allowed.

Each WDA will also have an incentive to trade allowances with other WDAs, depending on the relative costs to that WDA of diverting waste from landfill. The *Landfill Allowance Trading Scheme* encourages those waste disposal authorities with the lowest marginal diversion costs to landfill less than their allowances permit and sell the surplus to WDAs which face higher

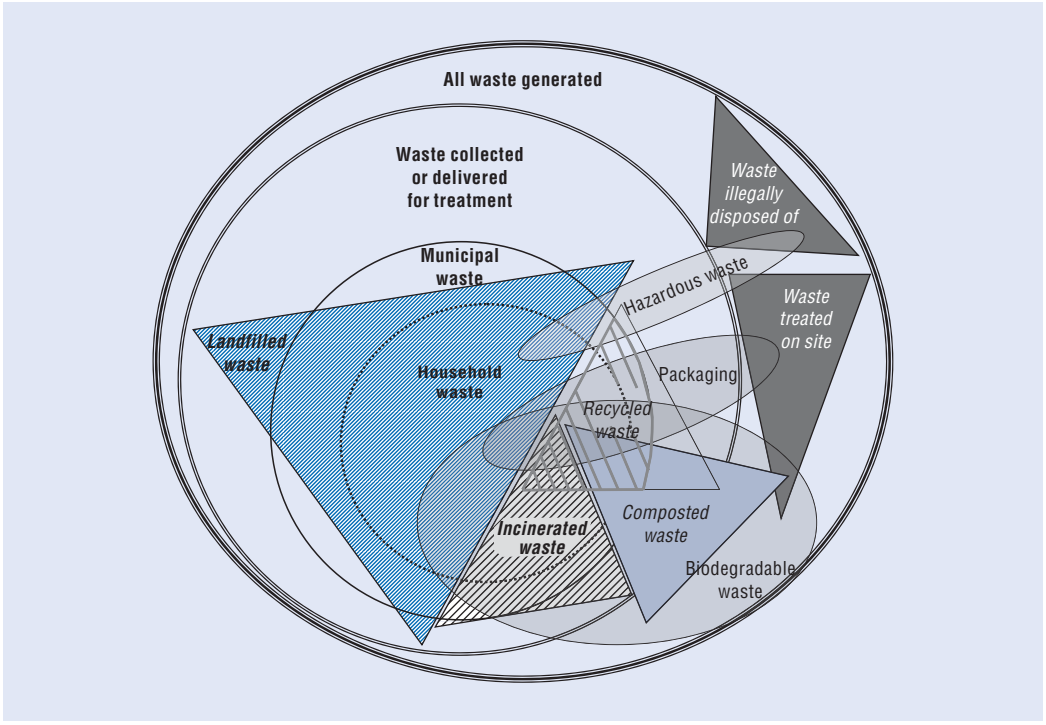
marginal costs of diversion. Conversely, WDAs with higher marginal costs of diversion may prefer to meet their obligations by buying allowances at a lower cost than actually undertaking the diversion themselves. Landfill allowance trading creates a market for landfill allowances and this flexibility helps meeting the targets of the Landfill Directive in a cost-effective way.

Trading under this scheme started in May 2005, and by March 2006, just over half a million allowances had been traded for periods up to 2008-09 at a cost (or income, for the selling authorities) of GBP 10 million, with 16 authorities being active in the market [see NAO (2006)].

2.4.3.3. *Statutory Recycling Targets*

In order to avoid WDAs turning too much towards incineration when stricter limits are placed on the amounts that may be landfilled, *Statutory Recycling Targets* have been set for local authorities. The "domain" of these targets is shown as the area with thick diagonal stripes in Figure 2.8.

Figure 2.8. **The "domain" of the statutory recycling targets in England**



"Domain" of the statutory recycling targets.

The original targets set used actual recycling levels in 1998/99 as a starting point. The local authorities were at first asked to double their respective recycling rates by 2002 and to triple them by 2005/6. The targets have now been somewhat modified.

2.4.3.4. *Household Waste Recycling Act*

The Household Waste Recycling Act of 2003 [*cf.* HMSO (2003b)] obliges all English Waste Collection Authorities to collect "at least two types of recyclable waste together or individually separated from the rest of the household waste" by the end of 2010, unless the cost of doing so would be unreasonably high or comparable alternative arrangements are available.

2.4.3.5. *Waste Implementation Programme*

Based on one of the recommendations from Strategy Unit (2002), DEFRA have also established the so-called Waste Implementation Programme. The aim of the programme is to help secure compliance with the limits on landfilling of biodegradable waste set in the EU Landfill Directive.

The Programme comprises eight work streams:

a) Local authority funding.

b) Local authority support.

c) Data.

d) Research.

e) New technologies.

f) Waste minimisation.

g) Kerbside recycling (focus on organics).

h) Waste awareness.

The work streams on waste minimisation, awareness and kerbside recycling are undertaken by the "Waste and Resources Action Programme" (WRAP), expanding that programme's original remit to develop secondary markets for recycled materials.[25] The budget for a three year period is approximately GBP 268 million. About 25 persons work on this programme at DEFRA, with another 20 working on it at WRAP.

2.4.3.6. *Waste Performance and Efficiency Grant*

As part of a general change in central government funding of local authorities, and as part of work stream *a)* under the Waste Implementation Programme, a three-year Waste Performance and Efficiency Grant scheme has been introduced, totalling GBP 260 million.[26] The scheme supports new ways to deliver waste reduction and increase recycling/diversion from landfill. It is intended to underpin and facilitate more effective use of the core funding available to local authorities. It can, for example, be used to improve separate waste collection services; promote home composting; investments in central composting or material recycling facilities; implement public awareness-raising campaigns; provide incentives for waste reduction and recycling by householders; and to improve co-operation between local authorities, etc.

With up-front certainty over levels of funding for the full three-year period, no bidding process and no prescription on usage, the Government expects local authorities to deploy their award in a way which makes most sense in their area, in order to comply with the statutory targets for recycling and composting. The grant allocation for each local authority is based on population size, adjusted to take account of commuters and day visitors, etc.

2.4.4. *Instruments addressing household waste in Wales*

2.4.4.1. *Statutory Landfill Diversion Targets*

As in England, gradually declining statutory *Landfill Diversion Targets* have been set for each of the local authorities in Wales, in order to reach the total maximum landfill amounts that can be seen in Table 2.6.[27] The initial allocation of allowances was based on how much each local authority *landfilled* in the year April 2001 to March 2002, while the allocation for

the year ending in 2010 is based on the amount of waste *arising* in each local authority in the year April 2001 to March 2002. Each of the local authorities has been mandated to produce a waste management strategy, taking the allocation of allowances into account.

Unlike England, however, there is no "Landfill Allowance Trading Scheme" in Wales. Local authorities can *not* comply with their diversion targets by buying allowances from other authorities where the diversion costs are lower. Implicitly, however, some flexibility is allowed – a case-by-case judgement of the reasons for any non-compliance in the initial years will be made. If local authorities can assure the responsible people at the Welsh Assembly Government that their non-compliance is temporary, and that necessary measures to (at least) comply with the obligation in subsequent years are being prepared or implemented, no sanctions are likely. To facilitate this approach the Welsh Assembly Government has established a peer review partnership with the Welsh Local Government Association, the collective body representing Welsh local authorities.

2.4.4.2. Voluntary targets for recycling and composting

In accordance with the National Waste Strategy for Wales (Welsh Assembly Government, 2002), central authorities in Wales seek to ensure that as much of landfill diversion as possible in local communities is done through recycling and composting – rather than incineration. To promote this, *voluntary* targets for recycling and composting have been set for local authorities – as opposed to the *statutory* recycling targets in England.

The difference in approach on this issue between England and Wales can in part be explained by the EU's system for provision of regional development aid. Wales qualifies for such aid, and projects related to waste recycling and composting can be supported – but only if the project goes beyond the *legal* obligations the authorities in question have imposed.

The targets set implied that by 2003/4 local authorities should increase recycling and (central) composting in total to 15% of the waste generated, with minimum 5% of both recycling and composting. The number for composting only includes collected green waste and food waste – no mixed wastes. By 2005/6, at least 25% of the waste generated was to be recycled or composted, with 10% minimum for both categories, measured by weight. By 2009/10, at least 40% of the waste generated is to be recycled or composted, with a minimum 15% of each.

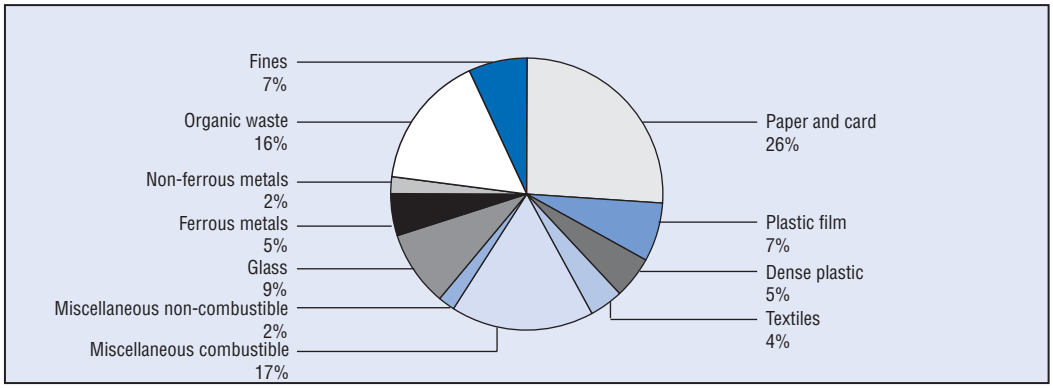
In addition to using financing from the EU, the Welsh Assembly Government has provided specific financial support to help the local authorities reach the recycling targets. Central authorities also want to promote home composting, but that requires new, robust data and methodologies on how to calculate the amounts the municipalities will be credited.

High collection rates for recycling and composting have been achieved in areas of Wales with low population densities. This is in part because the local authorities have found cheap, efficient collection methods, with sorting being done on the collection vans, etc. They have integrated the instruments to stimulate recycling with measures to employ low-skilled labour, and through a heavy involvement of the voluntary, community-based sector, people with learning difficulties and those on probation, etc.

2.4.5. Instruments addressing household waste in Scotland and Northern Ireland

An in-depth study has not been made of the instrument mixes used to address household waste in Scotland and Northern Ireland. However, some information is available. Figure 2.9 presents the findings of one study of the composition of household

Figure 2.9. **Typical household waste composition in Scotland**



Source: SEPA (2003), based on The Local Authority Waste Arising Survey 2000/01.

waste in Scotland, taken from SEPA (2003). When allocating the landfill allowances for Scotland, it has been assumed that 63% of municipal waste is biodegradable, *cf.* Roper (2004) – as opposed to the 68% assumption used in England and opposed to an estimate of 60% presented in SEPA (2003).

The proportion of allowances given to each local authority in 2004/05 was the same as their average share of landfilling in 2001/02 and 2002/03. The proportion of allowances given for 2009/10 was the same as the average share of waste arisings in 2001/02 and 2002/03, subject to a no increase rule. For the years in-between, a straight line extrapolation between the end years has been made, subject to a no increase rule, such that no local authority's allocation of allowances increases year on year [*cf.* Roper (2004)].

In Scotland, trading is allowed in the landfill diversion allowances (unlike in Wales), as is banking and borrowing of allowances, with an exception for target years and adjacent years.

There is a "Waste Aware Campaign" promoted by the Scottish Waste Awareness Group,[28] and The Scottish Executive provides funding to WRAP for waste minimisation work.

In Northern Ireland, parallel regulations are in place concerning landfill allowances and for producer take-back requirements for packaging. An evaluation on progress made on the implementation of the Waste Management Strategy for Northern Ireland is available [DoE (2004)]. Different phases of the information campaign "Wake up to Waste" have been evaluated [EHS (2003)].

2.5. Instruments addressing household waste in the Netherlands

2.5.1. *Introduction*

Current waste policies in the Netherlands build largely on initiatives taken in 1988-91 [VROM (2001)]. These include *The Memorandum on the Prevention and Recycling of Waste*, the introduction of producer responsibility, the establishment of the Waste Management Council and the signing of the "Packaging Covenant".

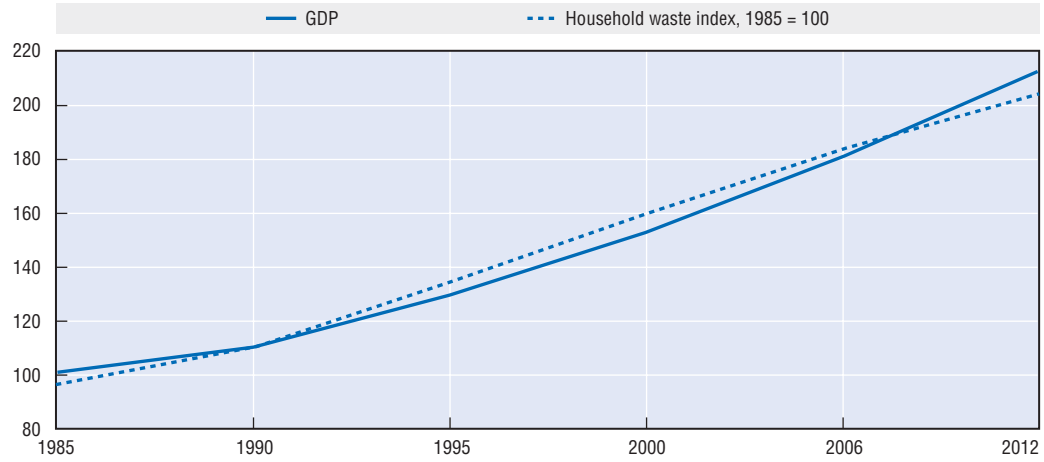
Waste treatment used to be largely a responsibility of the Provinces – while municipalities were responsible for collecting wastes from households. However, in 1997, it was decided that the central Government should take over many of the responsibilities of the Provinces. This change in approach came into effect in 2002. As part of the policy change, a decision was made that a National Waste Management Plan – covering *all types*

of waste – should be prepared every 4 years, and such a plan was indeed issued in 2002, *cf.* VROM (2004).[29] This plan is valid for the years 2002-06, and will be reviewed in 2006. The time-horizon of the plan is ten years (2002-12). The preparation of the plan included extensive consultation between the authorities, the business community and civic society. Preliminary plans were debated in the Parliament and underwent public enquiry.

In fact, household waste generation increased somewhat faster than GDP in the period 1990-2000. According to VROM (2004), this was caused in particular by a sharp increase in consumption and an increase in the number of households. In the National Waste Management Plan, however, it is projected that household waste generation will grow slower than GDP in the period up to 2012, *cf.* Figure 2.10.

Figure 2.10. **Changes in GDP and household waste generation in the Netherlands**

Historical data 1985-2000, projected change between 2000 and 2012



Source: VROM (2004).

Again according to VROM (2004), 45% of all household waste was reused or recycled in 2000. Of the remaining household waste, approximately two-thirds was being incinerated, while one-third, *i.e.* less than 20% of *all* the household waste, was landfilled. Since 2000, landfilling of combustible waste has decreased even further. Hence, fulfilling the limits of the EU Landfilling Directive concerning the landfilling of biodegradable waste is not much of a challenge in the Dutch case.[30]

However, Dutch authorities want to improve the environmental performance further and have set a number of new targets in the National Waste Management Plan. Among the new targets, it is said that "landfilling of the surplus combustible waste… must be ended within five years". While incineration is given priority over landfilling, the emphasis is prevention and recycling as far as possible. Section 7.14 discusses the high priority being given to avoiding landfilling in Dutch waste policy.

2.5.2. Charges for household waste collection

Dutch municipalities are *obliged to collect three types of waste separately*, namely compostable waste (*e.g.* food and garden waste), recyclable waste (*e.g.* glass, paper and textiles) and mixed waste. The compostable and the mixed wastes have to be collected at the kerb-side.

Dutch municipalities are – unlike the situation in United Kingdom – *free to determine how to finance* the collection system. The use of waste collection charges that vary with waste amounts has become increasingly common – but it is far from universal.

According to Dijkgraaf and Gradus (2004), approximately 20% of all Dutch municipalities had implemented a charge system where households pay per kg, per bag or per waste bin presented at the kerb for collection by 2000, *cf.* Table 2.7. This is in line with the response to the questionnaire administered for this report, which also indicated that an additional 40-60% of households pay a collection charge that is related to the number of persons in the household – which in turn *to some extent* can be related to the amount of waste being generated.[31]

Table 2.7. **Occurrence of unit-based waste collection charging systems in the Netherlands**

	1998	1999	2000
Municipalities with unit-based systems	**77**	**109**	**126**
Weight-based systems	9	10	13
Bag-based systems for mixed and compostable waste	6	6	6
Bag-based system for only mixed waste	13	12	14
Frequency-based systems	19	43	54
Volume-based systems	24	30	29
Unspecified type of system	6	8	10
Municipalities without unit-based systems	**461**	**429**	**412**
Total	**538**	**538**	**538**

Source: Dijkgraaf and Gradus (2004).

Dijkgraaf (2004) estimated the impacts of the choice of financing system on the total amount of waste, and on the amounts of unsorted, compostable and recyclable waste respectively, drawing on data on waste amounts collected by the Dutch Waste Management Council (AOO). His study indicates that the choice of financing system is important:

The estimated impacts on total waste amounts of charging systems based on *collection frequency* or on the *volume of the collection bin* were *relatively* modest – with reductions of 21% and 6% respectively.

Introducing bag-based collection charges for both unsorted and compostable waste was found to have almost the same impact on *total waste amounts* as a weight-based system – a reduction of 36%. If compostable waste is collected for free, Dijkgraaf (2004) found that the reduction in *total waste amount* would be much smaller – only 14%. The impacts on *mixed waste* were more or less the same in both cases (a reduction of about 50%), but the effects on the amount of *compostable waste* differed radically. If the compostable waste is collected for free in a bag-based system, the amount of such waste was found to *increase* by some 36%. If households also had to pay for the collection of compostable waste, the amount of such waste was estimated to *decrease* by more than 60%.[32]

In general, the administrative costs of a bag-based system are much lower than for a weight-based system. The finding that the two systems can have relatively similar impacts on waste amounts is therefore quite interesting.[33]

2.5.3. *Landfill tax*

The National Waste Management Plan places great emphasis on the waste hierarchy – or its Dutch variant: The "Lansink's ladder".[34] Prevention and recycling is given priority and landfilling of recoverable waste is described as "unacceptable".

In order to support the objective of avoiding landfilling of recoverable – or combustible – waste, a rather high tax on landfilling has been introduced, with a tax rate of EUR 85 per tonne. This is, for example, almost twice as high as the *long-term* "target rate" set for the *Landfill Tax* in United Kingdom.

There are no official estimates of the value of the externalities related to landfilling underpinning the high tax rate applied. The tax on landfilling is described in the *National Waste Management Plan* as an *instrument to achieve an end to landfilling* – not (necessarily) as an instrument to achieve an "optimal" amount of landfilling.

The amount of waste going to landfills *inter alia* depends on the competitive position of waste incinerators. Partly due to very strict air emission limits applied in the Netherlands, Dutch incinerators are quite costly to build and operate. In this context, the landfill tax is set at such a level as to make sure that the incinerators are able to compete. It is, for example, stated that "should it prove however during the period of the National Waste Management Plan (2002-06) that the landfill rates are nevertheless too low for managing the waste in the desired way, the need to raise the tax further will be reviewed".[35]

2.5.4. *Ban on landfilling*

The preference of Dutch authorities for incineration over landfilling is also illustrated by a ban on the landfilling of most types of combustible waste – which obviously overlaps considerably with the landfill tax. According to VROM (2004),

> "Until 1 January 2000, the date on which a massive increase in the landfill tax on combustible waste was introduced, full capacity utilisation at waste incinerators was achieved by being very circumspect in granting exemptions from the landfill ban. This circumspection was necessary because it was financially more attractive to landfill the waste than to incinerate it. Consequently, exemptions were only granted for the estimated surplus of combustible waste in a given period. Because it is extremely difficult to predict the quantity of waste arisings and available capacity from month to month, this produced a great amount of friction. Dispensations regularly had to be withdrawn or extra dispensations granted, resulting in a great deal of administrative red tape and uncertainty for waste suppliers and processors. Increasing the landfill tax greatly reduced the incentive to landfill the waste. This enabled full capacity utilisation of the waste incinerators to be better achieved without having to make constant use of controls by granting, or not granting, exemptions from the landfill ban."

As an example of this policy, landfilling of separately collected waste paper is prohibited under the Waste (Landfill Ban) Decree.

2.5.5. *Minimum standards for waste management*

The National Waste Management Plan contains specific "Sector plans" for a long list of waste streams. These sector plans include "minimum standards" for how the respective wastes are to be managed. The minimum standards set are to guide the conditions set in the environmental permits that firms are required to obtain in order to operate.

For example, the "minimum standard for the processing of domestic residual waste is removal by burning, with less than 5% of the input quantity,… being placed in landfill sites as residual substances". In the explanation given for this minimum standard in the *National Waste Management Plan*, it is said that it is "in keeping with an existing method of processing and is therefore feasible and reliable. Removal by burning of domestic residual waste is generally accepted as cost-effective".

2.5.6. Ban on exports and imports of waste destined for disposal

The Netherlands maintains a ban[36] on the export and import of waste destined for disposal and VROM (2004) states that as long as a "level playing field" as regards environmental requirements for incinerators is not created in the European Union, the borders will not be opened. It is explained that:

> "Opening national borders before this condition is met entails a real risk of the objectives of Dutch waste policy not being met. It may lead to the utilisation of Dutch incinerators for foreign waste, with the result that Dutch waste has to be landfilled, or to the incineration in foreign facilities of Dutch waste that is currently reused or recycled."

2.5.7. Incentive Programme for Waste Separation and Prevention

The rate of waste separation and separate collection from households is largely determined by local conditions. Co-ordination of national and local waste policy is therefore very important. Co-operation between the three tiers of government is arranged in the Waste Management Council (AOO).

According to VROM (2004), the respective roles with regard to waste separation by consumers are as follows:

- Municipalities are responsible for establishing a suitable collection system, for collecting waste, for promoting waste separation and for communication about the policy and the results.
- Provinces stimulate and facilitate local authorities in the area of waste prevention and separation, for example through monitoring and benchmarking (comparing and analysing the results achieved by local authorities).
- The Ministry of Housing, Spatial Planning and the Environment (VROM) promotes waste separation by households through regulation, research, financial assistance for projects and communication at national level.
- The Waste Management Council supports waste separation by households by co-ordinating national activities, collecting and disseminating information and through research, monitoring and benchmarking.

Subsidies are also given from central authorities to Provinces and municipalities for projects that contribute to household waste prevention and to separate collection.

2.5.8. Environmental Quality of Electricity Production Incentive Scheme

To stimulate the use of waste as fuel, the *Environmental Quality of Electricity Production Incentive Scheme* came into force 1 July 2003. This scheme included changes in the Regulatory Energy Tax and subsidy payments for electricity production based on waste.

2.9 euro cents is given in subsidies for each kilowatt-hour of electricity supplied by power plants, stand-alone installations and waste incineration plants that use biomass for auxiliary firing or co-incineration and that have an energy yield of more than 26%. Heat is also included in determining the yield threshold, but only electricity production counts for reimbursement. The purpose of the programme is to encourage companies to invest primarily in incineration as a *recovery* method, rather than in capacity for incineration as a *disposal* method.

2.5.9. Platform for secondary fuels

In the process leading to the preparation of the *National Waste Management Plan*, consultations were held with the business community to assess the obstacles the business community faces in actually launching planned initiatives related to post-separation, production of secondary fuels and incineration of waste streams with a high calorific value. Problems identified related to fuel quality, sales of the fuel produced, and financial costs.

In this connection, the Ministry of Housing, Spatial Planning and the Environment (VROM) and the business community set up the *Platform for Secondary Fuels*. The participants in the Platform include energy companies, waste incineration companies, relevant Ministries and the provincial authorities. The Platform has formed various working groups to study the problems mentioned. According to VROM (2004), one working group is developing a quality assurance system for secondary fuels, while a second is focusing on financial and other incentives to further support the initiatives and a third is focusing on the market in incineration capacity for the various streams.

2.5.10. Covenant with the coal-fired power stations

In order to promote fulfilment of the Kyoto targets for greenhouse gas emission reductions in the Netherlands, an agreement was concluded in 2002 between the Government and the Dutch electricity producers. The goal of this Coal Covenant is to reduce CO_2 emissions from power stations by 5.8 million tonnes a year in the period 2008-12. This reduction is to be achieved primarily through the use of biomass and waste, such as Refuse Derived Fuels and plastics, as replacements for the primary fuels, especially coal.

The approach of subsequent separation and production of secondary fuels presented in the *National Waste Management Plan* is meant to ensure that some of the surplus combustible residual waste is made available for auxiliary firing or co-incineration in power stations.

The reimbursement for producers on the basis of the Environmental Taxes Act originally provided for in the Coal Covenant has later been replaced by the new Environmental Quality of Electricity Production Incentive Scheme mentioned above.

2.5.11. Producer responsibility requirements

Different sorts of "extended producer responsibility" schemes are in place for a number of different product categories in the Netherlands. Table 2.8 summarises some of the waste streams and instruments that were involved when the *National Waste Management Plan* was elaborated.

As underlined in VROM (2004), instruments for promoting producer responsibility are generally used in combination with other instruments, *e.g.* the landfill ban and the landfill tax. Producer responsibility operations are often funded through the introduction of a waste management fee, to be paid into a common fund by producers or importers for each new product placed on the market.[37]

Table 2.8. **Waste stream where producer responsibility requirements apply in the Netherlands**

Instrument	Waste stream	Details of instrument
Rules and regulations	Packaging	Regulations governing packaging and packaging waste
	Car tyres	Decree governing management of car tyres
	White and brown goods	Decree governing management of white and brown goods
	Agriculture and horticultural plastic film	Decree governing management of agricultural and horticultural plastic film
	Batteries	Battery Management Decree
	Minor chemical waste	Minor Chemical Waste Logo Decree Regulation laying down further rules on the Minor Chemical Waste Logo
	End-of-life vehicles	Decree governing management of end-of-life vehicles
Covenant	Packaging	Packaging Covenant
Order declaring universally binding	Passenger cars	Agreement on waste management fee for passenger cars 2000
	Waste paper/board	Agreement on waste management fee for paper and board 2002
	Agriculture and horticultural plastic film	Agreement on waste management fee for agricultural plastic film 1999-2002
	Plastic cladding units	Agreement on waste management fee for plastic cladding units 2000-05
	Sheet glass	Agreement on waste management fee for sheet glass 2002-04
Voluntary	Plastic piping systems	

Source: VROM (2004).

Producers and importers often conclude an agreement concerning the payment of a waste management fee. Under the Environmental Management Act, the Minister can declare an agreement on a waste management fee universally binding if a substantial majority of producers or importers of a given product ask him to do so. This agreement must satisfy the requirements set out in the *Regulations governing requests for a waste management fee* and the *Guideline on declaring a waste management fee universally binding*.

In virtually every case for which producer responsibility has been introduced, producers have set up collective systems to meet their obligations. VROM (2004) finds that producer responsibility have added value especially where:

- Recovery of a product is accompanied by a chain deficit that can be bridged by levying the costs of waste management elsewhere in the chain.
- Getting the collection (separate or otherwise) and/or recovery of a product sufficiently off the ground requires producers and importers in the chain to play a managerial role.

According to VROM (2004), consideration will be given in the period covered by the Plan to a number of products for which the introduction of producer responsibility could contribute to better waste management. The advantages and disadvantages in terms of environmental benefits, costs and feasibility will be assessed.

Notes

1. The EU Landfilling Directive defines *biodegradable waste* as "any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and garden waste, and paper and paper board", *cf.* CEU (1999).
2. Austria, Belgium, Canada, Czech Republic, Germany, Hungary, Japan, Korea, Netherlands, Norway, Poland, Slovak Republic, Spain, Turkey, the United Kingdom and the United States.
3. The discussion here focuses in particular on instruments aiming to limit the size of various waste streams. For a further discussion of Design-for-Environment impacts, see OECD (2006d).

4. One of the potentially most important types of instruments to limit waste amounts – (variable) municipal waste collection charges – is not included in Table 2.1 (*cf.* Table 2.4 below). For a further discussion of the impacts of such charges, see OECD (2006b).
5. It is important to keep the footnote to Table 2.2 in mind here. In addition, as information concerning take-back requirements, product taxes and deposit-refunds systems draws on replies provided to *several* questions in the questionnaire, there is some over-focus on these instruments in the table. However, care has been taken to avoid double-counting of instruments.
6. Unfortunately, the information available does not say anything about how many producers actually participate in a PRO or collect their "own" products, in cases where both options are open to them. Hence, no exact picture of the DfE incentives actually can be given. For further discussion of Design-for-Environment impacts of waste policies, see Glachant (2004) and OECD (2006d).
7. See OECD (2006a) for further discussion.
8. See Section 7.8.2 for further discussion of the impacts of municipal waste collection charges.
9. Some parts of the United Kingdom have a two-level local administration system, while other parts have only one level of local administration. In the first case, the lowest level is responsible for waste *collection* (and are called Waste Collection Authorities [WCAs]) while the higher level is responsible for waste *disposal* (and are called Waste Disposal Authorities ([WDAs]). The WCAs are also responsible for recycling of municipal waste. The so-called "Unitary Authorities" of the latter case are responsible for both collection and disposal.
10. In recent years, developments have been better than the estimated 3% yearly growth rate, *cf.* Figure 7.3.
11. *Cf.* Strategy Unit (2002), *Waste not, Want not – A strategy for tackling the waste problem in England*.
12. DEFRA (2003a) is the "Government response to Strategy Unit" report *Waste not, Want not*. In general the Government agreed with the recommendations made by the Strategy Unit – and, for example, an annual increase in the Landfill Tax has been announced. No changes have yet been made concerning waste collection charges.
13. "Active waste" is defined as wastes containing putrescible material. This is close to the definition of "biodegradable waste" in the EU *Landfilling Directive*, *cf.* Section 2.3.
14. 1 January 2007, one GBP equalled about EUR 1.49 and USD 1.96.
15. HM Treasury (2006) stated that the "Government will also consider whether the standard rate of landfill tax needs to increase more steeply from 2008 onwards, or go beyond the GBP 35 per tonne already committed to for the medium to long-term, in order to encourage greater diversion of waste from landfill and more sustainable waste management options". HM Treasury (2007) announced an increase in the standard rate by GBP 8 per year from 1 April 2008, until at least 2010-11.
16. A very similar estimate of the disamenity value was made in DEFRA (2003b). This study concludes: "The equivalent 95% confidence interval estimate of the present value of fixed disamenity effects of landfill is between GBP 334 350 and GBP 478 990 per landfill site in GB, and this corresponds to a nominal measure of fixed disamenity cost of between GBP 1.52 and GBP 2.18 per tonne of landfill at current prices for an assumed average flow of 100 million tonnes pa for 28 years at a 6% discount rate."
17. These estimates from 1993 are – when counted in today's money value – of a similar order of magnitude as the new estimates of the externalities related to landfilling and incineration from HM Customs and Excise (2004) referred to in Section 2.4.1.
    It is of interest to note that ECON (2000) provided a significantly higher estimate (approximately EUR 40 per tonne waste) for the external costs of landfilling in Norway, see Martinsen and Vassnes (2004). While a careful analysis of the reasons for this difference has not been undertaken here, DEFRA (2004c) includes a discussion of the robustness of different valuation exercises.
18. No importance should be given to *size* of the various areas in the figure. In particular, the part of the area with thick horizontal lines representing the higher tax rate that is situated outside the circle representing household waste is far too small.
19. Formally, the Regulation discussed here covers England, Scotland and Wales. A parallel regulation is in force for Northern Ireland. This is an example of an administrative/legal system that seems to require considerable co-ordination effort.
20. DEFRA *et al.* (2003) is a *User's Guide to the Producer Responsibility Obligations*.
21. The description here draws on Salmons (2002) – which provides an in-depth discussion of this instrument.

22. The Regulations also made provision for obligated companies to meet their own recovery and recycling targets, either through their own activities, or by contracting with third parties to collect waste packaging on their behalf, and to arrange delivery to reprocessors.

23. At *www.defra.gov.uk/environment/waste/localauth/lats/pdf/tableb-latsallocat%20.pdf* one can see the allowance given to each WDA.

24. It should be kept in mind that, for practical reasons, the part of the area that stretches beyond the "household waste" circle is too small. The share of landfilled biodegradable municipal waste that is not stemming from households is larger than what the figure could indicate.

25. A more detailed description can be found in DEFRA (2003c).

26. Local authorities will receive GBP 45 m in 2005/06, GBP 105 m in 2006/07 and GBP 110 m in 2007/08. In 2005/06, GBP 40 m was allocated according to a formula meant to reflect individual local authorities' need to spend on waste. The remaining GBP 5 m in 2005/06 was distributed as part of a pilot programme focusing specifically on local authority incentives for households to recycle and reduce waste. Further information can be found at *www.defra.gov.uk/environment/waste/localauth/funding/index.htm*. AEA Technology (2006) provides an evaluation of this pilot programme.

27. In Wales, there is a one-level local administration everywhere. The allocation to each administration can be found at *www.wales.gov.uk/subienvironment/content/landfill/landfill-ltr-e.pdf*.

28. *Cf. www.wascot.org.uk/*.

29. The original plan from 2002 was slightly modified in 2004, to reflect two decisions of the European Court of Justice concerning the distinction between recovery and disposal of waste. The judgements say that when waste is incinerated in a plant designed to dispose of waste, it should be counted as "disposal", regardless of whether or not some or all of the heat produced by the combustion is recovered. If waste is incinerated in installations where the main purpose is to enable the waste to be used as a means of generating energy – for example, in a cement kiln or in a power station – it is counted as energy recovery, *cf.* VROM (2004) and European Court of Justice (2003b) and (2003c).

30. The focus in Dutch waste policy tends to be on whether the waste is *combustible* or not – rather than whether it is *biodegradable* or not, which is at the focus in the EU Landfilling Directive. As opposed to the term "biodegradable", "combustible waste" includes *e.g.* plastic wastes.
VROM (2004) states that: "A weak point in current waste management is that there is still insufficient grip on the quantity of waste produced, particularly consumer waste… which is still increasing. The situation is being exacerbated as these are also the waste streams for which the level of recovery is lagging behind the targets. A great deal of combustible and recoverable waste is consequently still being disposed of in landfills because, when planning incineration capacity, it had been assumed that the waste supply would be smaller and the degree of separate collection higher."

31. An important difference between a unit-based charging system and a charge depending on household size is that the latter does not give any incentive for households to change their behaviour. With a given number of persons in a household, there is no way to lower the charge payments by generating less waste.

32. The increase in the collected amounts of *recyclable waste* was also larger for municipalities that charged to compostable waste. Dijkgraaf (2004) states that:
"This suggests that in municipalities using a bag-based system only for unsorted waste, part of the recyclable waste is 'dumped' in the free compostable waste can. The intuition behind this result is that it takes less effort to use this can than to use the recyclables facility. The compostable waste can is in the direct vicinity of the house, while the collection infrastructure for recyclable waste is farther away, resulting in more time needed to deliver the recyclables."
It is not quite clear what happens to the *compostable waste* that is not collected when households have to pay per bag for such collection, but – as the impacts on mixed waste is about the same in both cases – it seems likely that a significant share of it is home composted. (Compostable) garden waste, such as lawn cuttings, might also be left on the ground.

33. Dijkgraaf (2004) points out that one reason that he found larger impacts of bag-based systems than some earlier US studies have indicated, is likely to be the fact that the sizes of the bags normally used in the Netherlands (50 to 60 litres) is only about the half of the sizes generally used in the United States (113 to 121 litres). While Dijkgraaf (2004) found the price elasticity for the amounts of unsorted waste when bag-based systems were introduced in Dutch municipalities to be between –0.5 and –0.7, Fullerton and Kinnaman (1996), for example, found a price elasticity of –0.08 when charging per bag was introduced in Charlottesville in the State of Virginia, and Podolsky and Spiegel (1998) found a price elasticity of –0.39 when comparing 149 municipalities in the State of New Jersey. When small bags are used, it is easier for households to respond to a change in the bag price by changing the number of bags of waste they set out for collection.

34. Named after the Dutch politician Ad Lansink who proposed a motion that was adopted unanimously by the Dutch lower House of Parliament in 1979, where waste prevention is ranked prior to waste recovery, which in turn is ranked higher than incineration and – as the least favoured processing method – landfilling [*cf.* VROM (2001) and MNP (2005)].

35. Previously, there had been a moratorium on the construction on new incinerators *for waste disposal* – as opposed to incineration for *energy recovery* in cement kilns, power plant, etc. However, in VROM (2004), this moratorium was lifted, and no limit was set for the expansion of capacity in waste incinerators. On the other hand, VROM (2004) makes it clear that no new landfilling capacity will be licensed during the plan period.

36. Section 7.7.2 discusses further this ban.

37. VROM (2004) makes some comments on the importance of "internalising" the fee in the prices of the various products. However, regardless of whether or not the fee is "visible" in the bills customers are facing, it seems likely that the consumers will base their purchasing decisions on the full product price – including any "hidden" or "visible" fee. Hence, it is difficult to see that the exact method of presenting the fee to the consumers is of major importance.
    It could also seem useful to reserve the term "internalisation" to describe the incorporation of the value of the relevant *environmental externalities* in product prices consumers and producers are facing. There is no *direct* link between these externalities and the *costs of operating* a producer responsibility organisation.

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Chapter 3

Instrument Mixes Addressing Non-point Sources of Water Pollution

This chapter describes instrument mixes dealing with non-point sources of water pollution in agriculture in four countries: Denmark, the Netherlands, the United Kingdom and the Chesapeake Bay area in the United States. For Denmark, separate studies are made on instrument mixes addressing nitrogen run-off, phosphorous run-off and pesticides use. Mixes addressing both pesticide use and nutrient run-off are also described regarding the United Kingdom, while the case studies of the Netherlands and the Chesapeake Bay area only focus on nutrients run-off. In the latter case, both Federal measures and State-level measures in Virginia and Maryland are analysed.

3.1. Background

This chapter describes instrument mixes used to address non-point sources of water pollution in OECD member countries, focussing on *nitrogen and phosphorous run-off from agriculture* and – in some of the country case studies – on *pesticides use in agriculture*.

Inputs of nutrients, such as nitrogen and phosphorus, are essential to agricultural production. If soils are farmed and nutrients not replenished, this can lead to declining soil fertility and may impair agricultural sustainability through "soil mining" of nutrients. At the same time, a surplus of nutrients in excess of immediate crop needs can be a source of environmental damage to surface and ground water (eutrophication), air quality (acidification) and contribute to global warming.

Surface water and marine water pollution into rivers, lakes and coastal waters, particularly from phosphates, can accelerate the process of eutrophication (*i.e.* algae growth and oxygen shortages in water). This can impair the use of surface water for drinking, and damage the biodiversity of these aquatic environments and harm their use for fishing and recreational purposes.

Groundwater pollution (*e.g.* of aquifers) from nitrates can be damaging to human health. Pollution of groundwater is more problematic than that of surface water, since groundwater, once polluted, may remain contaminated for many years, whereas surface water is refreshed relatively rapidly.

Agricultural pesticides contribute to agricultural productivity, but also pose potential risks to human health and the environment. The risks vary greatly, depending on each pesticide's inherent toxicity (or hazard) and exposure. Exposure to a pesticide depends on the way it is applied and its mobility and persistence in the environment.

3.2. Overview of instrument mixes used in OECD countries

This section presents an overview of instruments applied in OECD countries to address non-point sources of water pollution, based on responses to a questionnaire that was sent to member countries in February 2004, to which 17 countries responded.[1]

The questionnaire consisted of two main parts, focussing on nitrogen and phosphorous run-off and on pesticide use in agriculture respectively. Altogether, 93 *national policy objectives* were singled out in the responses – 44 concerning nutrients run-off, 35 regarding pesticide use and 14 relating to both issues, *cf.* Table 3.1. In all, 346 policy instruments were identified in the questionnaire responses – 198 concerning nutrients run-off, 119 concerning pesticide use, and 29 addressing both issues.[2]

Both concerning nutrients run-off and pesticide use, *regulatory instruments* are most common in the member countries that have responded, with 82 and 57 instruments being mentioned, respectively.[3] *Economic instruments* are the most common type of instrument addressing *both* environmental issues. For nutrients, the second largest group is economic instruments (54 cases), largely dominated by various types of subsidies. Most of the

Table 3.1. **Overview of policy instruments addressing non-point sources of water pollution**

	National level				State/provincial level				Total
	Nutrients	Pesticides	Both	Total	Nutrients	Pesticides	Both	Total	
Policy objectives	44	35	14	93	–	–	–	–	93
Policy instruments	137	78	25	240	61	41	4	106	346
Regulatory instruments	54	37	7	97	28	20	0	48	146
Economic instruments	37	8	9	54	17	7	1	25	79
Thereof taxes	*2*	*4*	*0*	*6*	*2*	*1*	*0*	*3*	*9*
Thereof subsidies	*32*	*1*	*7*	*40*	*13*	*7*	*1*	*21*	*61*
Information instruments	32	25	7	64	11	14	2	27	91
Other instruments	14	10	2	26	5	0	1	6	32

25 economic instruments listed regarding pesticides or *both* nutrients and pesticides use are also some sort of subsidy. For pesticide use, however, *information instruments* are significantly more common than economic instruments, with 39 cases identified in all; while for nutrients run-off, 43 information instruments have been listed in the responses. The smallest category for both environmental issues (*other instruments*) consists for example of voluntary approaches and research and development initiatives. Nineteen such instruments were found to address nutrients run-off, ten are used to affect pesticide use, and three such instruments address both environmental issues.

Table 3.1 also distinguishes between instruments applied at the national level and those applied at the state/provincial level.

Comparing the questionnaire responses concerning instrument mixes addressing household waste (Section 2.2) with the responses regarding non-point sources of water pollution, one can notice that taxes and charges play a much smaller role in the latter case than in the former. At the same time, *subsidies* – and cross-compliance requirements related to subsidies given for other purposes – play an important role in instrument mixes addressing non-point sources of water pollution. This can seem symptomatic: given the heavy reliance on subsidies in agricultural policy in most OECD member countries, it is perhaps not so surprising that, in the environment area as well, little emphasis is placed on making "polluters pay".

3.3. EU Directives addressing non-point sources of water pollution

Before describing in more detail the specific instrument mixes applied in some OECD member countries to address non-point sources of water pollution, it can be useful first to describe a number of Directives that apply to each of the OECD countries that are also EU member States. These Directives obviously play an important role in – and for – the relevant instrument mixes in the countries concerned.

Regarding run-off of nitrogen, the most important EU Directive is the *Directive concerning the protection of waters against pollution caused by nitrates from agricultural sources* [see CEC (1991a)]. This Directive has the objectives of:

- reducing water pollution caused or induced by nitrates from agricultural sources; and
- preventing further such pollution.

The EU Nitrates Directive obliges member states to decrease the nitrate load from agricultural sources to groundwater and surface waters, and to avoid further pollution from these sources. Member states have to:

a) assign areas that are vulnerable to nitrate leaching (*Nitrate Vulnerable Zones* – NVZ);

b) establish action plans to decrease nitrate leaching and to monitor the effectiveness of these plans; and

c) develop codes of good agricultural practice to guide farmers.

Nitrate Vulnerable Zones are areas where: *i)* surface freshwaters contain – or could contain, if preventative action is not taken – *nitrate concentrations greater than 50 mg per litre*; *ii)* groundwaters which contain – or could contain, if preventative action is not taken – nitrate concentrations greater than 50 mg per litre; and *iii)* natural freshwater lakes, or other freshwater bodies, estuaries, coastal waters and marine waters which are eutrophic, or may become so in the near future if protective action is not taken. The Directive also requires all areas of land known to be draining into nitrate-polluted waters to be identified as *Nitrate Vulnerable Zones*.

The core of the Directive is that a balance should be reached between nitrogen supply by soil, animal manure and fertilisers, and nitrogen demand by the crops. Actions are to be taken at two levels, depending on the environmental quality of the affected waters. Outside *Nitrate Vulnerable Zones*, voluntary codes of good practice have to be promoted. Within the vulnerable zones, *mandatory* measures are required (*e.g.* regarding minimum manure storage capacities, restrictions in time and location on manure and slurry application). The maximum application of *animal manure* is 170 kg N per ha and year; however, restrictions must be more stringent if this limitation is not enough to secure achievement of the set water quality objectives.

A higher rate of manure application *can* be allowed when justified on the basis of certain criteria laid down in the Directive, provided that the objectives of the Directive are achieved in time.

Another EU Directive of major – and increasing – importance for policy formulation related to non-point sources of water pollution is the *Water Framework Directive* [EU (2000a)]. The purpose of this Directive is to protect all waters (rivers, lakes, groundwaters, coastal and transitional water) and to achieve, as a rule by 2015, "good status" for these waters. In addition to nutrients run-off, this Directive is also *inter alia* of direct relevance for the instrument mixes that address pesticide use. Planning and action under this Directive has to be *river basin*-based, with mandatory co-ordination across administrative and political borders, aimed at delivering a management plan for the whole river basin. For groundwater, the objective of "good status" is complemented by an obligation to reverse any significant upward trend in pollutants. Certain issues will be handled by "daughter" Directives – on the protection of groundwater and on dangerous substances ("priority substances"), the progressive reduction of discharges, emissions and losses of priority substances and the cessation or phasing-out of discharges, emissions and losses of the priority hazardous substances. A first environmental assessment (pressures and impacts) of all river basins was due under the Directive by the end of 2004.[4]

The Directives on the *Quality on Drinking Water*, *cf.* CEC (1980a) (no longer valid) and CEU (1998), also played an important role in determining instrument usage related to non-point sources of water pollution in some EU member states. These Directives *inter alia* set an upper limit of 50 mg nitrate per litre water intended for human consumption. This limit

was the basis of the water quality targets in the Nitrates Directive discussed above, and hence has strong implications for the agricultural practices that can be allowed near sources of drinking water.

Council Directive 91/414/EEC [*cf.* CEC (1991b)] on the *Placing of Plant Protection Products on the Market Place* also plays a vital role among instruments addressing pesticide use in EU member states. The Directive seeks to harmonise the overall arrangements for authorisation of plant protection products within the EU. This is achieved by harmonising the process for considering the safety of active substances at the Community level, by establishing agreed safety criteria. Product *authorisation* remains the responsibility of individual member states. The Directive establishes a "positive list" of active substances (Annex I) that have been shown to be without unacceptable risk to people or the environment. Member states can only authorise the marketing and use of plant protection products after an active substance has been listed in Annex I, except where transitional arrangements apply.

In addition to this Directive and the Water Framework Directive, the Council Directive 80/68/EEC of 17 December 1979 on the *Protection of Groundwater Against Pollution Caused by Certain Dangerous Substances* [*cf.* CEC (1980b)] is also of particular importance for the instruments applied in some EU member states to address pesticide use. The purpose of this Directive is to prevent pollution of groundwater by substances belonging to the families and groups of substances in Lists I or II in an annex, and, as far as possible, to check or eliminate the consequences of pollution which has already occurred.[5]

To comply with these obligations, member states shall *inter alia* prohibit all direct discharge of substances in List I.

Finally, new provisions have been introduced in the EU *Common Agricultural Policy* (CAP), which have major importance for pollution stemming from the agriculture sector:

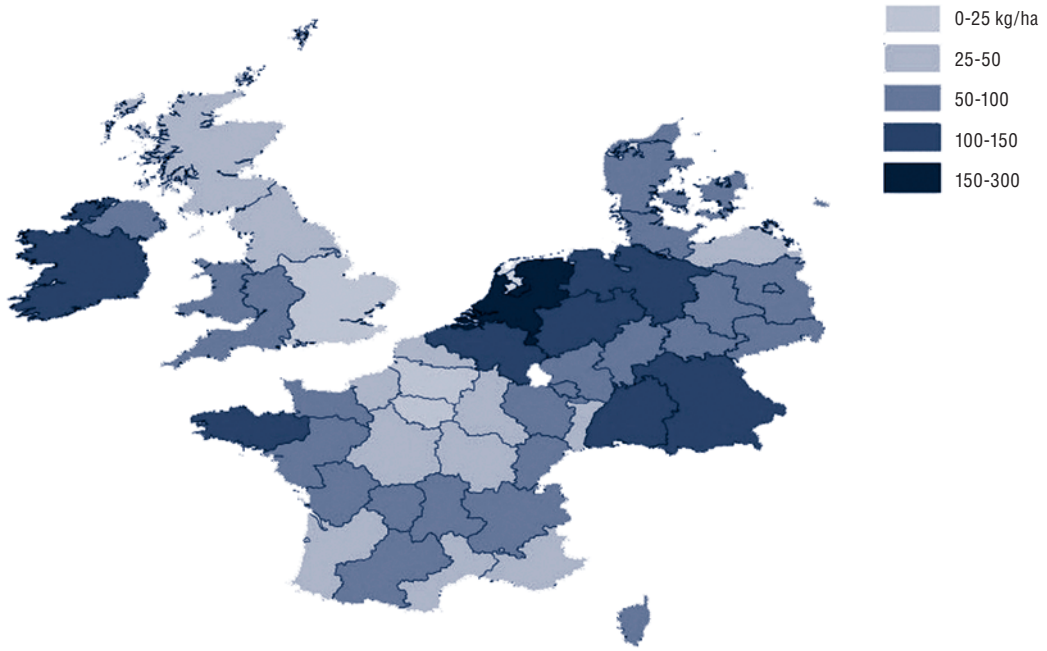
- *de-coupling* (farmers receive a single farm payment, independent of production levels);
- *cross-compliance* (the single farm payment is subject to compliance with a core set of EU legislation on environment, food quality and animal protection); and
- *modulation* (shifting considerable budgets from traditional market support mechanisms to measures for rural development and agri-environmental schemes).

Farmers also have to maintain their agricultural land in Good Agricultural and Environmental Condition (GAEC) as from 1 January 2005. Failure to comply with these preconditions will result in a reduction or exclusion of subsidy payments.

CEU (2003) is the EU Council Regulation that formally introduced the above-mentioned provisions. For the present analysis, it is of importance that CEU (2003) explicitly states that compliance with the *Nitrates Directive* and the *Directive on the Protection of Groundwater Against Pollution Caused by Certain Dangerous Substances* is a requirement for a farmer to receive direct payments.

Figure 3.1 illustrates the nitrogen surpluses existing in some EU member states in 1999. It is clear that there are very significant differences in these surpluses, both between and within countries. One can notice that the case studies of the United Kingdom, the Netherlands and Denmark described in this chapter cover areas with the highest and the lowest nitrogen surpluses according to this Figure. The geological and hydrological conditions in these countries also vary significantly, with the water-tables being particularly high in the Netherlands.

Figure 3.1. **Nitrogen surpluses in some EU member states, 1999**



Source: RIVM (2004a).

3.4. Instruments addressing non-point sources of water pollution in the United Kingdom

This section presents information on instrument mixes used to address non-point sources of water pollution from agriculture in the United Kingdom. The focus of the presentation is on instruments that are applied in *England*.

3.4.1. Instruments addressing nitrogen

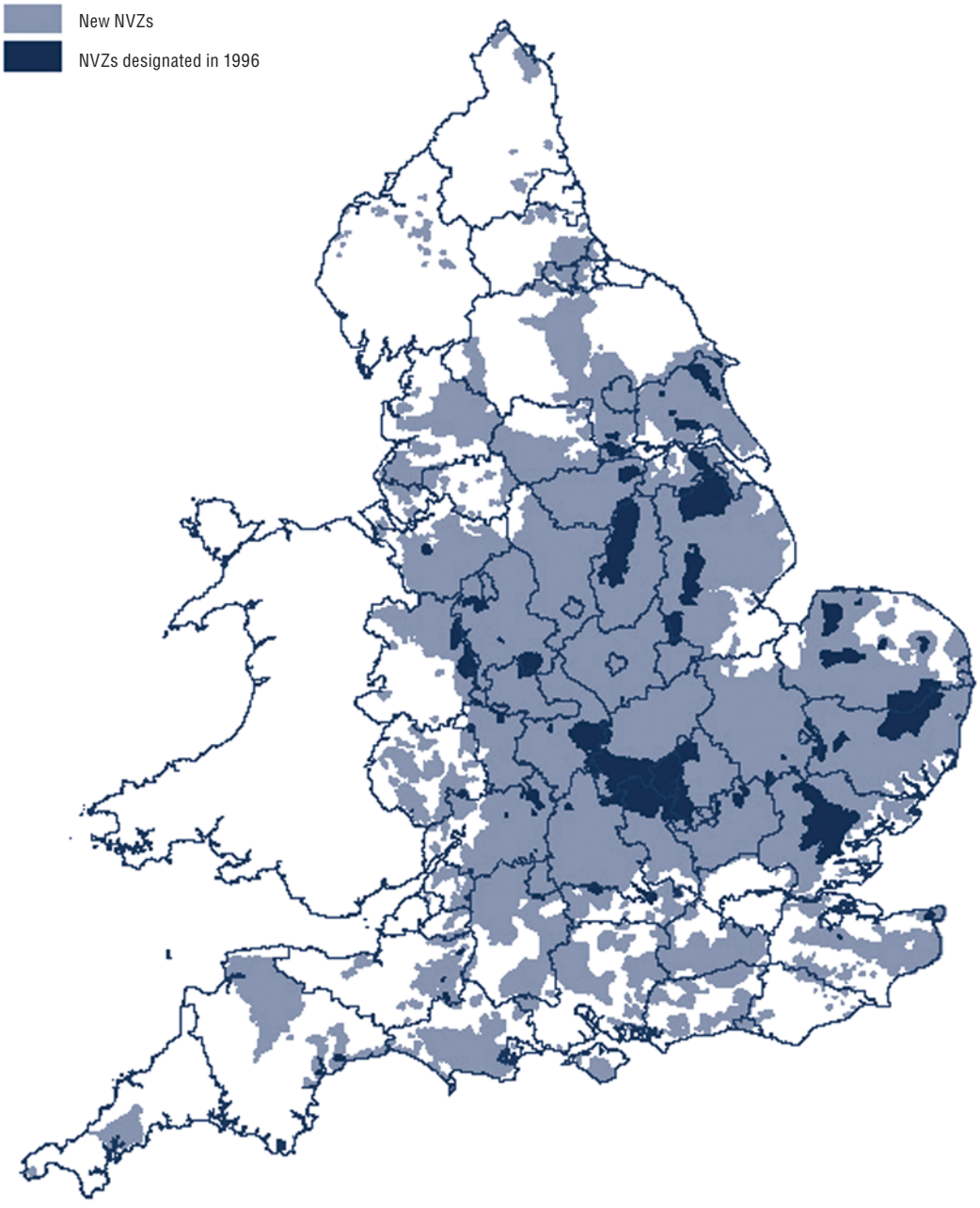
As can be seen from Figure 3.1, the nitrogen surpluses in most parts of England in 1999 were modest, compared to the surpluses in Denmark and the Netherlands. Compliance with certain aspects of the EU Nitrates Directive has nevertheless been a challenge for the United Kingdom.

3.4.1.1. Nitrate Vulnerable Zones

Like in other EU countries, UK authorities were obliged under the Nitrates Directive to designate *Nitrate Vulnerable Zones*. The *Protection of Water against Agricultural Nitrate Pollution Regulations 1996* [see HMSO (1996)] – that covers England and Wales – contains the original UK designation of such zones, representing about 8% of the total area of *England*. However, a judgment by the European Court of Justice in December 2000 ruled that the UK had failed to designate sufficient areas to protect all surface and groundwaters – not just *drinking water sources* – against diffuse nitrate pollution from agriculture.

In response to this ruling of the European Court of Justice, the (Additional Designations) Regulations 2002 extended the NVZ coverage with an additional 47% of the area of England [see HMSO (2002)]. The extent of both the original and the additional zones is illustrated in Figure 3.2.

Figure 3.2. **Nitrate Vulnerable Zones in England**



Source: DEFRA (2002a).

An Action Programme has been prepared, *inter alia* to limit nutrients run-off in the NVZs. The programme is set out in a separate regulation, see HMSO (1998a). The measures apply only in NVZs, promote "best practice" in the use and storage of fertiliser and manure, and build on the guidelines set out in the *Code for Good Agricultural Practice for the Protection of Water* [see MAFF (1998a)].[6]

The Nitrates Directive provides for some discretion over the content of Action Programmes but there are certain measures which must be included. There are four key aspects to these measures:

- Limit inorganic nitrogen fertiliser application to crop requirements, after allowing fully for residues in the soil and other sources.
- Limit *organic manure* applications to 170 kg of nitrogen per ha each year on the area of the farm not in grass, and *total nitrogen* applications to 250 kg per ha of each year on the area of grass on the farm.
- Not apply slurry, poultry manures or liquid digested sludge on sandy or shallow soils between 1 September and 1 November (grassland or autumn sown crop), or between 1 August and 1 November (fields not in grass without autumn sown crop). The storage capacity available for those animal manures which cannot be applied during the closed period must be sufficient to cover these periods, unless other environmentally acceptable means of disposal are available.
- Keep adequate farm records, including cropping, livestock numbers and the use of organic manures and nitrogen fertilisers.

Failure to comply with the measures set out in the NVZ Action Programme may result in prosecution. The Environment Agency may also serve a notice requiring action to be taken, or any other relevant steps which it considers necessary, to ensure that there is no continuation or repetition of the activity to which the notice relates.

If action is not taken to remedy the situation within the timeframe stipulated by the Environment Agency, the case may be brought to court. If found guilty, the individual concerned may be liable, on summary conviction, to a fine not exceeding the statutory maximum of GBP 20 000.

3.4.1.2. Compliance with environmental regulations as a condition for receiving subsidies

At least as important as the possibility of fines referred to above could be the changes to the EU's Common Agriculture Policy described above. DEFRA (2004d) provides a consultation paper discussing the practical implementation in England of CEU (2003). Naturally, DEFRA (2004d) includes compliance with the UK regulations relating to the Nitrate Directive as a condition for future farm subsidies. In general, DEFRA (2004) states:

> "In many cases, the conditions will be no more than compliance with existing legal requirements. There will be no extra regulation for some other requirements although the frequency of some inspections may increase, for a number of standards, we will adapt the existing inspection regime so that it includes the check for cross compliance purposes. These inspections will be co-ordinated and operated in order to minimise the same questions being asked by many agencies.
>
> In other cases, there will be new standards. These are being designed to meet the EU requirements – for example on protection of soil. We are consulting on these standards and will make the final decision after considering views and carefully assessing a number of aspects including cost/benefit analysis – which will be set against the backdrop of cumulative existing and proposed burdens on the relevant sectors of English farming. Where relevant we are proposing to adopt existing standards, reinforce existing legislation and where there are strong environmental drivers – introduce new measures."

3.4.1.3. *The Farm Waste Grant Scheme*

To assist farmers in *Nitrogen Vulnerable Zones* who needed to construct and upgrade manure handling and storage facilities in order to comply with the NVZ Action Programme measures, the Farm Waste Grant Scheme was put in place. A grant rate of 40% on eligible expenditure, up to an investment ceiling of GBP 85 000 for each agricultural business, was available on eligible expenditure up to 2006.[7] The scheme required approval by the European Commission because it is construed to be "state aid". The Commission required that applications under the scheme must be prioritised.

3.4.1.4. *The Nitrate Sensitive Areas Scheme*

Prior to the introduction of the *Nitrate Vulnerable Zones* in England, a special subsidy scheme was introduced to stimulate changes in agricultural practices in *Nitrate Sensitive Areas* (NSA). Ten such areas were included in the initial scheme, launched in 1990, while 22 other areas were brought into the scheme in 1994. The areas were selected based on non-compliance with the (former) EU Directive on Drinking Water Quality [see CEC (1980)] – *i.e.* where the concentration of nitrate in drinking water exceeded 50 mg per litre.

The Scheme was *voluntary* and compensated farmers for 5-year undertakings to significantly change their farming practices to help reduce nitrate pollution of supplies of drinking water. The 32 *Nitrate Sensitive Areas* covered approximately 35 000 ha of eligible agricultural land.[8] The Scheme required entrants to follow that part of the *Code of Good Agricultural Practice for the Protection of Water* on all their land within the boundary which:

- restricted organic manure applications within 10 metres of a watercourse or 50 metres of a source that supplies water for human consumption;
- limited total organic manure applications to 250 kg per ha of total nitrogen in any one 12-month period.

Participants in the scheme were also required not to damage, destroy or remove environmental features on or bordering land which was part of the Scheme, and to keep full records of all organic and inorganic nitrogen applications.

The voluntary NSA scheme requirements were distinct from the mandatory Action Programme measures introduced under the EC Nitrate Directive. The NVZ measures, which restrict the quantity and timing of applications of nitrogen fertilisers and livestock manures, *today* largely equate to good agricultural practice, and are mostly uncompensated – even if the limits on manure applications in the NVZs now are significantly stricter than the criterion used under the NSA scheme.

Subsequent to a general *Comprehensive Spending Review* of the entire state budget in the UK, the NSA scheme was, however, closed for new entrants in the summer of 1998.[9]

3.4.2. *Instruments addressing phosphorous*

There are a few possibilities contained in relevant laws that in principle could be used to address phosphorous run-off in England and Wales – but in practice they would be very difficult to implement.

One theoretical option would be to serve notices under the *Groundwater Regulations 1998*. These relate to the protection of groundwater from List I and II substances, and phosphate is a List II substance. However, in practice, the burden of evidence required means that it is very difficult to do so, particularly for diffuse sources of phosphate. These powers are therefore hardly ever used.

The *Water Resources Act 1991* (Section 93) makes it possible to designate *Water Protection Zones* (WPZ) and prohibit or restrict such activities as described in the designating order, in order to prevent or control the entry of poisonous, noxious or polluting matter to controlled waters. In principle, these powers could apply to diffuse water pollutants, such as phosphates. However, in practice it is difficult to designate a WPZ due to the amount of information and effort required to justify the designation. On a rural catchment, with relatively little data available, it would be very difficult to build a comprehensive case for a WPZ, even though ecological impacts were being recorded.

The *Water Resources Act 1991* (Section 161) also allows the Environment Agency to issue *Anti-Pollution Works Notices* to prevent or deal with the consequences of the entry of any poisonous, noxious or polluting matter or any solid waste to controlled waters. These powers are, however, more suited to preventing water pollution from small scale point source discharges, such as from oil tanks or chemical stores. For diffuse phosphate run-off, these powers would be very difficult to use.

Except for these – largely theoretical – options, no instruments are applied in England or Wales to specifically address phosphorous run-off.

3.4.3. Instrument mixes addressing pesticide use

3.4.3.1. The Plant Protection Products Regulations

The EU *Directive on the Placing of Plant Protection Products on the Market Place* is transposed into English and Welsh legislation through the *Plant Protection Products Regulations* of 2003 [see HMSO (2003)]. This regulation is believed to have the highest importance of all instruments in limiting harm to human health or the environment from the use of pesticides – in that potentially (very) harmful products are completely banned from the market.[10]

3.4.3.2. The Groundwater Regulations

The EU Groundwater Directive was transposed into the legal system of the whole United Kingdom through the *Groundwater Regulations* [*cf.* HMSO (1998b)]. Any disposal of pesticides will require an authorisation.

The Environment Agency can serve notices to farmers ordering them to change practices, even if the farmer is following all the formal rules – if he nevertheless is causing pollution of groundwater by List I or List II substances. Compliance with these regulations is (since 2005) required in order for farmers to receive the "Single Farm" subsidy payment of EU's *Common Agricultural Policy*.

3.4.3.3. The Pesticides Forum

While responsible use of pesticides principally is sought in the United Kingdom through the product approval system, many issues concerning pesticide use relate to the *way* products are handled and used on the farm. To address these issues, the *Pesticides Forum* was established in 1996. The Forum's membership includes a wide range of organisations representing those who make, use or advise on pesticides use, as well as environmental, conservation and consumer interests. Many of the Forum members engage in initiatives to promote "best practice". The Forum provides a mechanism for exchanging ideas and for encouraging joint initiatives to address particular issues. It also provides advice to the Government on pesticide usage matters.

3.4.3.4. *Codes of Practice*

The *Code of Practice for Using Plant Protection Products* [see DEFRA *et al.* (2006)] is binding for all professional users of plant protection products in England and Wales in respect of Part III of the *Food and Environment Protection Act* 1985 and the regulations controlling pesticides, particularly plant protection products, under that part of the Act. It replaced three previous codes of practice from 15 December 2005:

- a Code of Practice for the Safe Use of Pesticides on Farms and Small Holdings [*cf.* MAFF (1998c)];
- a Code of Practice for the use of Approved pesticides in Amenity Areas and Industrial Areas – a voluntary code produced by the National Association of Agricultural Contractors and the British Agrochemical Association;
- those parts of the Code of Practice for the safe use of Pesticides for Non-Agricultural Purposes – The Control of Substances Hazardous to Health Regulations 1999 which deal with forestry.

The Code's statutory basis means that it can be used in evidence if people are taken to Court for offences involving pesticides.

Sheep dips are pesticides with a very high degree of toxicity that are used to keep parasites, etc., away from sheep. Most dips contain organophosphorus or synthetic pyrethroid compounds which can leach through the soil and pollute groundwater. Once groundwater becomes contaminated with sheep dip compounds it will be unfit for drinking without prior treatment. A special protection code has hence been prepared under the Groundwater Regulation to promote the safe use and disposal of these compounds, *cf.* DEFRA (2001).

3.4.3.5. *Organic farming*

There is in place a special *Organic Farming Scheme*, as part of the agri-environmental measures in the England Rural Development Programme. Almost no pesticides may be applied to the participating farms. Funding for the scheme totals GBP 23 million for each year up to 2006/07. Some 2 250 farmers have committed approximately 175 000 ha under the scheme.

3.4.3.6. *The Voluntary Initiative*

The "Voluntary Initiative" on pesticides use in the United Kingdom is a formal, negotiated agreement between the Government and the farming industry, which was launched as an alternative to a pesticides tax that also was – and still is – under consideration. 18 Indicators were selected in 2002 to measure the progress of this initiative. The targets set under the initiative are related to improving water quality, benefiting biodiversity, and addressing behavioural changes. See OECD (2007a) for a further discussion.[11]

3.5. Instruments addressing nutrients run-off in the Netherlands

3.5.1. *Introduction*

As shown in Figure 3.1 above, the situation concerning nutrient surpluses in the Netherlands in 1999 was more serious than in the United Kingdom. No other region in the countries covered had such a large nitrogen surplus as that found in the Netherlands. According to MANMF (2001), agriculture accounts for almost 70% of the total nitrogen leached to Dutch surface waters.

The nutrients policy of the Netherlands is complex. It also addresses a stubborn and complicated problem. It has a history of about 20 years of changes, successes and failures. The first Dutch policies and measures regulating animal manure date from 1984. Roughly four phases can be distinguished:

a) Attempts to stop the growth in animal production (1984 to 1990);

b) A step-wise decrease of the manure burden (1990 to 1998);

c) Attempts to balance inputs of N and P to outputs of N and P (1998 to 2005); and

d) Application standards for N and P (2006 onwards).

At the outset, policies focused on the phosphate content of manure. Phosphate was chosen over nitrogen because the phosphate cycle is less complex than the nitrogen cycle, and a relatively constant amount is produced per animal.[12] It was initially hoped that a decrease in nitrogen problems would follow naturally from imposing limits on phosphates, based on the assumption of a fixed nitrogen-to-phosphate ratio per type of animal manure. Experience has shown, however, that such an indirect steering of nitrogen loads was too rough and incomplete. Specific nitrogen application rates were consequently introduced in 1998.

3.5.2. *The first phase of manure policy – manure quotas*

Initially, the manure surplus was strongly associated with pig and poultry production, and the first phase of the manure policy *banned a further growth* of these sectors. These sectors had grown almost exponentially between 1960 and 1985, in part as a response to the Common Agriculture Policy of the EU, see Table 3.2.[13] The ban on further growth of the pig and poultry sectors coincided with the introduction of milk quotas in EU member states, aimed at limiting the growth of the surpluses of dairy products – and thereby the intervention costs for the EU. The implementation of the milk quotas prevented further growth of the dairy sector. As the milk quotas decreased in subsequent years (while the milk production per cow continued to increase), the number of dairy cows started to decrease for the first time.

During the first phase of the manure policy, limits were also set for the application of manure to agricultural land, based on the amount of phosphate in the manure. Initially, these limits were much higher for maize land than for grassland and arable land.[14] The initial limits reflected where the pressure of the manure burden was highest, especially in

Table 3.2. **Changes in some characteristics of agriculture in the Netherlands**

Million units, 1950 to 2003

	1950	1960	1970	1980	1990	2000	2003
Surface area agricultural land, ha	2.3	2.3	2.2	2.1	2.0	2.0	1.9
Area of greenhouses, ha	0.003	0.005	0.007	0.01	0.012	0.016	
Milking cows, number	1.4	1.6	1.9	2.5	1.9	1.5	1.5
Pigs, number	2	2	6	10	14	13	11
Poultry, number	41	45	55	81	93	105	100
Horses, number	0.2	0.1	0.05	0.05	0.1	0.2	
Tractors, number	0.02	0.08	0.14	0.18	0.18	0.16	
Fertilizer N, kg	156	224	396	485	412	340	
Fertilizer P, kg	52	49	48	36	33	27	
Fertilizer K, kg	128	115	107	93	81	70	
Energy use inland economy, PJ	606	925	2 014	2 732	2 702	3 024	
Use of natural gas, PJ	0	11	635	1 274	1 290	1 467	

Source: Smaling *et al.* (1999), CBS/LEI (2000); CBS (2001) and CBS (2004).

Figure 3.3. **Livestock concentration areas in the Netherlands**



Source: Based on Wossink (2004). The nutrients surpluses are most pronounced in Surplus region South.

regions with sandy soils in the east and south (*cf.* Figure 3.3). The first phase of the manure policy was further characterized by a belief that the manure surplus could be solved by technological innovations.

Under the 1986 *Manure Act*, each farm was required to calculate an annual reference level of manure production in phosphate terms. This was obtained by multiplying the number of animals held on the farm on 31 December 1986 by a *fixed phosphate coefficient for each animal species*.[15] The Act also made it illegal for a farm to produce more manure per year than what would result in a manure-to-land ratio of 125 kg phosphate per ha, based on land either owned or in long-term lease. An important exception was provided to existing farms that had a higher ratio on the reference date, but these farms were: *a)* not permitted to produce more manure than their reference level; and *b)* required to pay a tax of EUR 0.11 per kg phosphate above 125 kg per ha and EUR 0.23 per kg phosphate above 200 kg per ha from 1 May 1987.

Farms with a manure-to-land ratio of less than 125 kg phosphate per ha (defined as "manure deficit" farms) or new farms could increase animal numbers until this level was reached. Farms with a ratio above 125 kg phosphate per ha ("manure surplus" farms) were much more restricted. They could increase animal numbers by acquiring additional land,

but had to acquire enough land to reduce the ratio to below 125 kg phosphate per ha before they could actually hold more stock. Second, they could increase their manure reference level. However, the manure reference level was *not tradable* and was only transferable under certain strict criteria (marriage, heritage or the transfer of the complete farm). Pig and poultry farmers were also given the opportunity to show, through the "mineral input registration system", that the phosphate excreted by their animals was less than the set coefficients by improving their nutrient feeding regime. This incentive was only applicable to reducing the amount subject to levy, and did not change their manure reference level.

In the lead-up to the introduction of farm-specific manure reference levels, the Dutch farmer's organisation (LTO) protested strongly against their non-tradability. They feared this constraint would limit the expansion, and hence the international competitiveness, of livestock farming in the Netherlands. In response, the Minister of Agriculture promised to consider the question of tradability when introducing the manure production rights. There was considerable debate within the farming community over this issue. Small farmers and those about to commence were against trading, while large farmers were in favour of it. Some wanted to exclude certain farms according to their size or location.

Many farmers had anticipated the reference date of 31 December 1986 and had consequently *stalled more animals than normal*. After 31 December, they cut back their herds to normal size. In this way, the farmers acquired a higher manure reference level than they actually "needed". Officials were concerned that once trading was allowed, farmers would sell these "dormant" rights, leading to both a windfall financial gain for them and an increase in nutrient leakages, as the "dormant" rights came to be used by other farmers.

3.5.3. The second phase of manure policy – trading in manure quotas

The second phase started roughly in 1990/91, and included:

a) A lowering of the application limits for animal manure;

b) Restrictions on the timing of manure application and the resulting requirement to take care of sufficient storage capacity for animal manure;

c) Implementation of various measures to decrease ammonia (NH_3) emissions to air; and

d) Further facilitation of manure distribution and manure processing.

The application limits for animal manure were lowered step-wise, *e.g.* for maize land from 350 kg in 1987; to 200 kg in 1991; 100 kg in 1998; and to 80 kg phosphate per ha per year in 2002. Because of the relationship between N and P in animal manure, the application of N stemming from animal manure also decreased. Restrictions on the application of animal manure in autumn and winter forced farmers to build facilities for storing animal manure for up to 6 months. An agreement was made with the suppliers of animal feed to lower the phosphorus and protein contents of the animal feed – and thereby the P and N excretion of the animals and the emission of NH_3 from manure. Finally, the transport of animal manure from areas with a large surplus to areas with a potential demand for manure was subsidised and facilitated, to decrease the environmental burden in the areas with intensive livestock production.

On 1 January 1994, trading was introduced to the manure quota system. To signal the change in policy, the manure reference level for each farm was *renamed* as a "manure

production right" (still expressed in phosphate terms). Rules were designed to either prevent an increase, or to encourage a decrease, in environmental pressure as a result of trading:

- A farm's manure production right was divided into two parts: a *land-based quota* [derived by multiplying the hectares farmed (owned or in long-term lease) by 125 kg phosphate] and a *non-land-based quota* (the remainder). Trading was only permitted for the non-land-based quota.
- To account for the existence of "dormant" rights, the level of phosphate manure production was reassessed for each farm for the years 1988-90. This did not change a farm's manure production right, but if the original reference level was greater than the highest annual level of 1988-90, the difference was not tradable.
- For each farm, the non-land-based manure quota was allocated to three specific animal categories (cattle and turkeys, pigs and chickens, and others), reflecting the farm specific situation. Trading was permitted within – but restricted between – animal categories (*e.g.* manure production rights allocated to cattle and turkey *could not be purchased for pig and poultry production*, but manure production rights allocated to pig and poultry could be purchased for cattle production).
- Trading was also geographically restricted. Two surplus regions, with average manure production greater than 125 kg phosphate per ha were identified: one in the east and the other in the south (see Figure 3.3). Trading in manure production rights could take place *within* and *between these two surplus regions*, and production rights could be sold to manure deficit regions, but *rights were not allowed to be brought into the surplus regions* from a manure deficit region.
- To reduce the overall volume of manure, 25% of the quota involved in each trade was retired.
- Farmers who purchased manure production rights had to *certify that they had land* (either owned or under minimum two-year contract) to apply their total amount of manure at the appropriate land application rates. Farms with phosphate manure production greater than 125 kg phosphate per ha could expand by purchasing additional rights, but were restricted by the need to comply with land application rates set down by regulations.

The rules originally stated that the quota system would be terminated on 1 January 1997. This reflected the understanding reached in 1993 between the Ministry of Agriculture, the Ministry of Environment and LTO that by 1998 the quota system would be obsolete with the introduction of a farm-level nutrient accounting system, MINAS (Wossink and Gardebroek, 2003).

3.5.4. *The third phase of manure policy – tradable quotas and MINAS*

The third phase of the manure policy started with the implementation of the *MINerals Accounting System* (MINAS) at farm-level in 1998. This phase was meant to solve the remaining nutrient surpluses, once and for all. This phase also marks a change in understanding and policy. Technological innovations were no longer seen as the main solution for the manure burden. The insight grew that a drastic improvement of nutrient management would be needed to decrease N and P losses at farm-level to environmentally acceptable levels. The third phase also marked a change from a policy that was largely based on regulations to a policy that was based on economic stimulations. MINAS was seen

as a flexible instrument that would be able to address the large differences between farms in environmental performance. However, contrary to the initial idea, the manure quota system described above was *not* abandoned.

MINAS was implemented in 1998 on livestock farms with more than 2.5 livestock units per ha, and in 2001 on all other farms. MINAS involved registration of all N and P inputs and outputs at farm-level. Inputs of N and P via fertilisers, animal feed, animal manure, compost and other sources, as well as the N and P output ("export" at farm-level) in harvested products, including any animal manure, had to be recorded accurately, using official documents for sales and purchases from accredited firms only.[16]

If the difference between total N and P inputs and outputs exceeded specified levels, farmers had to pay a levy per unit above these levy-free surpluses. These levies provided an incentive to both lower the in-take of N and P from fertilisers, animal feed and/or animal manure, and to increase the "export" from the farm of harvested products and animal manure. The levy-free surpluses were differentiated according to soil type and land-use, and were lowered several times between 1998 and 2003 (see Tables 3.3 and 3.4).

Table 3.3. **Levy-free surpluses for nitrogen in MINAS**

Kg per ha per year

Land use and soil type	1998	2000	2001	2002	2003	2004	2005
Grassland, dry sandy soils	300	275	250	190	190	160	140
Grassland, other soils	300	275	250	220	220	180	180
Arable land, peat/clay	175	150	150	150	150	135	125
Arable land, dry sandy soils	175	150	125	100	100	80	80
Arable land, other soils	175	150	125	110	110	100	100

Source: OECD (2005a).

Table 3.4. **Levy-free surpluses for phosphorous in MINAS**

Kg per ha per year

Land use	1998	2000	2001	2002	2003	2004	2005
Grassland	17.5	15.3	15.3	10.9	10.9	8.7	8.7
Arable land	17.5	15.3	15.3	13.0	13.0	10.9	8.7

Source: OECD (2005a).

The levy-free surpluses for N were lower for grassland on nitrate leaching-sensitive dry sandy soil than on other soils, while there was no difference between soil types in the levy-free P surpluses, and differences between land uses were small.

In 1998, levies for surpluses exceeding the levy-free amounts were EUR 0.68 per kg for nitrogen and EUR 2.6-10.4 per kg for phosphate. In 2002, the levies were raised to EUR 2.53-5.07 per kg for N and to EUR 20.60 per kg for P, which is about 5 to 10 times the price of fertiliser N and 50 times the price of fertiliser P, respectively. Levies were increased again in 2002 to make them almost "prohibitive", in response to queries by the European Commission about the environmental effectiveness of the initial levies.

The size of the levy-free surpluses and the size of the levies defined the room to act for farmers. When levy-free surpluses were low, farmers had little room to act. They could not utilize all potentials of the farm, or they had to pay the cost for disposal of excess manure to elsewhere. They were forced to decrease the in-take of N and or P via fertilisers, animal

feed or animal manure and compost, and to increase the efficiency of utilization of the smaller amounts of N and P that could be bought, in order to maintain the yields at similar level. They were challenged to implement cost-effective measures to decrease N and P surpluses. In principle, farmers had the choice of decreasing the surpluses to the level of the levy-free surpluses and pay no levy, or to decrease the surpluses less and pay a levy. In practice, however, the increased levies were meant to be prohibitive, and taking measures was more economical than paying levies.

The size of the levy-free surpluses and the size of the levies also defined the losses of N and P to the wider environment. MINAS presumed a relationship between the size of the levy-free surpluses and the size of N and P losses. *In the long-term, N and P surpluses according to MINAS would indeed have been proper indicators for N and P losses to the environment*. In the short-term, however, this was not the case, in particular for P. The lack of a direct relationship between N and P surpluses in agriculture and N and P losses from agriculture was often an item for debate. Farmers often found it frustrating that their efforts to decrease N and P surpluses had so little direct impact on environmental quality. The lack of a direct relationship between N and P surpluses in agriculture and N and P losses from agriculture was also part of the basis for scepticism about the environmental effectiveness of MINAS – and at the same time the main political argument for decreasing levy-free surpluses less than initially planned.

There were about 40 alterations and changes in MINAS after its implementation in 1998. These included changes in the size of the levy-free surpluses and the size of the levies, but also changes in default values and various correction factors used in the calculation of the nutrients surpluses. These changes were made in response to increased insights over time, complaints by farmers and political lobbying. Not all of these changes made MINAS more effective and transparent: in some sense, they also led to confusion among farmers. A strongly debated change was the *exclusion of fertiliser P as an input* on the MINAS balance, in response to pressure from arable farmers threatening to block implementation of MINAS in arable farming in 2001, unless P fertiliser was excluded from the accounting.

3.5.5. Additional instruments addressing manure

In response to complaints by farmer unions and some politicians about the negative economic and social impacts of the manure policy on agriculture, additional instruments were implemented to support the objectives of the policy and to support an ecological modernisation of agriculture. These instruments included:

a) Action Plan for Nitrate Projects;

b) Quota Buy-out for the Termination of Livestock Farms;

c) Social Economic Plan for Animal Husbandry; and

d) The Manure Transfer Agreement System.

The *Action Plan for Nitrate Projects* was implemented in 1999 to support nutrient management at farm-level, in order to facilitate the acceleration of the decrease of levy-free surpluses for N and P in MINAS. The action plan included 20 projects focused on on-farm research, concerning validation of measures to improve nutrient use efficiency at farm-level and dissemination of the results into practice. The regulation for the termination of livestock farms focused on a *Quota Buy-out for the Termination of Livestock Farms* by the Government to decrease the manure surplus. The *Social Economic Plan for Animal Husbandry* focused on advising livestock farmers about future developments and about termination of farming.

The *Manure Transfer Agreement System* (MTAS) was introduced 1 January 2001, in an attempt to comply with the EU *Nitrate Directive*.[17] This system linked production and use of animal manure. Farms that produced more manure than the amount that could be applied on their own land according to the application standards according to the Directive were obliged to have a contract with a farmer who was willing to apply the surplus of manure on his farm. Livestock farms that produced more N in animal manure than 170 kg per ha arable land per year and more than 250 kg per ha grassland per year were required to submit to the government a double-signed agreement which stated that the surplus amount of animal manure was contracted by other farms that could adequately accommodate this amount of manure. If farms with a surplus amount of animal manure were not able to submit such a manure transfer agreement by 31 December, the license for animal production for the next year was lost. The implementation of the Manure Transfer Agreements was facilitated by intermediates that organised the contacts and agreements between farms. Farms with a surplus amount of animal manure paid farms with a "deficit" for signing the agreement.

3.5.6. The EU Nitrate Directive and MINAS

The consequences of the EU *Nitrate Directive* for animal agriculture in the Netherlands can seem to have been underestimated initially. Upon the approval of the Directive, there was still a belief that the manure problem could be solved using technological innovations, and that the Directive provided enough degrees of freedom so as to implement a regulatory system that fitted agriculture of the Netherlands, as long as the objectives of the Directive were achieved (*i.e.*, concentrations of nitrate in groundwater should not exceed the standard specified in the EU Drinking Water Directive of 50 mg nitrate per litre). This view turned out to be not correct.

In response to the adoption of the *Nitrate Directive*, the *whole of the Netherlands was assigned as a Nitrate Vulnerable Zone*. MINAS was seen as the main policy instrument for all grassland and arable and horticultural land to achieve the objectives of the Directive. Application limits for N from animal manure were not addressed in the first Action Plan that was prepared as a follow-up of the Directive. The Netherlands already had application standards for animal manure based on P, and as the ratio between N and P in animal manure is rather fixed, the application of N from animal manure was *indirectly* also regulated. Furthermore, the application limits for N from animal manure set in the Nitrate Directive were considered by Dutch authorities to be too low, and not realistic for agriculture in the Netherlands.

The European Commission criticised the lack of application limits for N from animal manure and the lack of specific prescriptions and regulations for the use of animal manure and fertilisers in the first Action Plan. The responses of the Dutch government to the criticism of the Commission were diverse. One response was the implementation of additional regulations and prescriptions for the use of animal manure and fertilisers. A second response was an acceleration (by five years) of the planned decrease of the levy-free surpluses in MINAS and a strong increase in the size of the levies. A third response was to demand a derogation under the *Nitrate Directive*, which would allow the application of a larger amount of animal manure on grassland than the given limit of 170 kg N per ha. Analyses indicate that up to 290 kg N per ha on dry sandy soils, and up to 360 kg N per ha from animal manure on other soils, may be applied to grassland without violating the limit of 50 mg nitrate per litre in the groundwater [see Willems *et al.* (2000)]. Reviews by the Commission expressed concerns about the size of the derogation demanded, and challenged the justification for grasslands on peat soils and for reseeded grassland.

A fourth response of the Dutch Government to the criticism of the European Commission to the first Action Plan was the implementation of the *Manure Transfer Agreement System* described above as of 1 January 2001.

The European Commission considered the responses of the Dutch Government to be insufficient. By the end of 1999, the Commission brought the Dutch government to the European Court of Justice, which subsequently condemned the first Action Plan of the Netherlands in an arrest of 2 October 2003 [see European Court of Justice (2003a)]. The main arguments given by the Court were that: *i)* the MINAS system does not comply with the regulatory system prescribed by the *Nitrate Directive*; *ii)* the application limits for animal manure and the levy-free surpluses set for the years prior to 2000 were too high; and *iii)* essential regulations of the manure policy were implemented too late.

In response to the decisions of the European Court of Justice and the European Commission, the Dutch Government decided in a third Action Plan that crop- and soil-specific N fertilization standards should be implemented and that MINAS would be abandoned as of 1 January 2006. Further, the *Manure Transfer Agreement System* was abandoned as of 1 January 2005. Other reasons for dismissing the *Manure Transfer Agreement System* were the administrative burden and the high costs it created for intensive livestock farms.

3.5.7. The fourth phase of manure policy – Application standards

Subsequent to the judgement of the European Court of Justice, Dutch authorities and the European Commission reached an agreement about the implementation of the *Nitrates Directive* for the period 2006-09. This agreement is reflected in the *Third Action Programme*, which will be reviewed at the end of 2007, and revised if necessary.[18]

The *Third Action Programme* introduced a system of application standards for both nitrogen and phosphate from 1 January 2006. The application standards system sets a limit on the total use of fertilisers for both nitrogen and phosphate on the one hand, and on the use of animal manure for nitrogen on the other hand, and thus serve to further reduce the use of these nutrients.

The Application Standard of 170 kg of nitrogen per ha concerning *animal manure* is, for pigs and poultry, based on calculations of the production of manure at each farm and the quantity of manure brought into or removed from each farm, taking changes in stocks into account, and making deductions for gaseous losses from the stables. As under MINAS, there is mandatory sampling, analysis and weighing of animal manure brought onto or shipped out from a farm.

For cattle and other grazing livestock, default values for excretion are used. These values are linked to the milk production per cow, so that a higher milk production goes hand in hand with a higher net nitrogen and phosphate excretion.

Dutch authorities did, however, ask for a derogation from the 170 kg nitrogen per ha limit for animal manure, giving farms with at least 70% of grassland the possibility to apply up to 250 kg nitrogen from animal manure per ha. The request was based on scientific findings that concluded that such application levels would not jeopardise the water quality objectives of the Nitrates Directive. This derogation has now been approved.

Concerning *total nitrogen fertilisation*, a set of application standards varying between crop categories and soil types has been determined. The starting point for setting the standards was the scientific fertilisation recommendations, but tighter standards were set for grassland, maize and several arable and vegetable crops on sandy soils, in order to reach

to reach the 50 mg nitrate per litre water quality standard of the Nitrates Directive by 2009. The application standards will also be made gradually stricter over time, especially the standards pertaining to sandy soils.

The system also includes estimates of the nitrogen efficiencies of different categories of manure – *i.e.* assumptions on how much of the nitrogen is available for uptake by the plants. These assumptions will be made stricter over time. Autumn application of slurry on clay arable land will be banned from 2009.

Going beyond the requirements of the *Nitrates Directive*, the new Dutch system also sets application limits for total phosphate fertilisation. The standards are higher for grassland than for arable land, but they do not vary between crop types. Phosphate application standards for arable land and grassland will be tightened gradually to reach equilibrium fertilisation in 2015.

Neither for animal manure, for total nitrogen fertilisation nor for total phosphate fertilisation will the system allow offsetting an excess one year against the result of a previous or later year. Non-compliance with the new standards can trigger sanctions in the form of administrative fines or criminal prosecution, in cases of very serious infringements and of fraud.

3.6. Instruments addressing non-point sources of water pollution in Denmark

3.6.1. Instrument mixes addressing nitrogen run-off

3.6.1.1. Introduction

A considerable amount of resources has been spent over recent decades on analysing policies and instruments addressing both nutrients run-off and pesticides use in Denmark. This should be seen *i.a.* in the context of: major eutrophication problems in Danish lakes and coastal waters; significant public focus on possible dangers to health and environment associated with pesticide usage; – and the need to comply with various EU Directives.

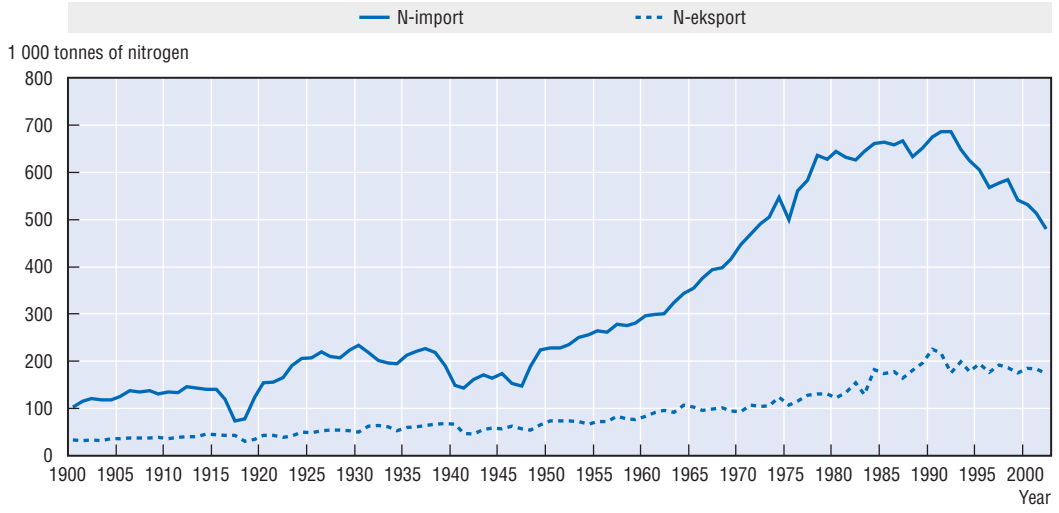
Concerning nutrient run-off, most of the work has focused on preparing and evaluating several consecutive *Action Plans for the Aquatic Environment*. Action Plan II covered the period 1998-2003, and aimed to secure a reduction in *nitrogen* leaching from agriculture by some 37 000 tonnes of N, in order to fulfil a target – originally set in 1987 for Action Plan I – of a 50% reduction in the nitrate leaching.

Developments in "imports" and "exports" of nitrogen to the Danish agriculture sector over the last century are illustrated in Figure 3.4. "Imports" in this graph include nitrogen brought into the sector through feedstuff and artificial fertilisers, from nitrogen in precipitation and by fixation of nitrogen from the atmosphere by nitrogen-fixing plants. The "exports" consist of nitrogen incorporated in plant and animal products sold from the sector. The difference between imports and exports represents the nitrogen surplus of the sector, that later can leach to ground- or surface waters. The figure shows that a strong decrease in the nitrogen surplus has taken place since the first *Action Plan* was implemented. At the same time, overall agricultural production has been relatively stable.

Action Plan III, which covers the years 2005-15, is very closely related to the implementation of the EU *Water Framework Directive* and the EU *Habitats Directive*. These state that objectives and programmes of measures for individual water bodies and natural habitats to apply from 2009 must be laid down. The main rule is that the objectives of the two Directives are to be met by 2015 the latest. Evaluations of the progress of the Action Plan will be made in 2008 and in 2011.

Figure 3.4. **Developments in nitrogen imports to – and exports from – Danish agriculture**

1900-2002



Source: Mikkelsen *et al.* (2004) – Modified from Dalgaard and Kyllingsbæk (2004).

The (rather comprehensive) preparatory work for Action Plan III drew on several formalised models of *environmental* impacts of different changes in agricultural practices and on two, somewhat different, formalised *economic* models, used to explore which instrument usage would be most cost-effective.[19]

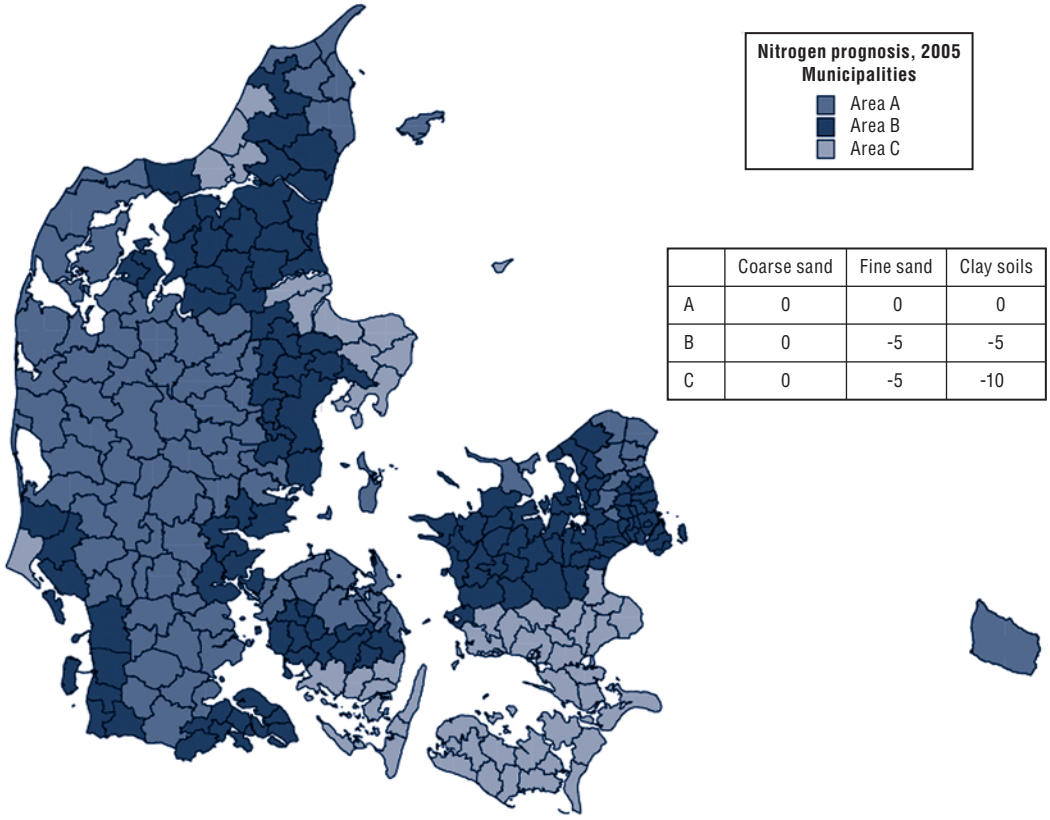
Most of the Danish studies focus on impacts on estimated nitrogen leaching – rather that impacts on the nitrogen surplus. Based on Blicher-Mathiesen and Grant (2003), it is estimated that about one-third of the nitrogen added to the soil leaches out from the root zone, and is thus no longer available for plants. Part of the nitrogen that leaches denitrifies into the air, while other parts could end up in rivers, lakes, costal areas or in the groundwater.

3.6.1.2. Nitrogen accounts and quotas at farm level

The cornerstone of the Danish policies addressing nutrients run-off is an obligatory, detailed *accounting* system for the use of nitrogen in inorganic and organic fertilisers and the use of catch crops (*i.e.* crops that fix nitrogen from the air) at the farm level, which sets limits on the plant-available nitrogen to be applied to different crops. Under this system, a *preliminary*, yearly nitrogen quota is calculated for each farm, depending on the size of the arable land, the crops planted, the soil category, etc. The quotas are based on norms for nitrogen application that are set *10% below the estimated economic optimum* for the different crops, thus causing crop production per area unit to be lower than what it otherwise could have been.

The preliminary quotas for each year are *modified, depending on weather conditions* that have prevailed in that year in different parts of the country, in order to make sure that fertilisation as closely as possible reflects the plants' needs – but maintaining the 10% reduction compared to the estimated economic optimum. Figure 3.5 illustrates the weather-related quota modifications that were made for 2005. Farms in areas B and C were obliged to reduce their application of nitrogen 5 or 10 kg per ha on fine sands and clay soils, respectively.

Figure 3.5. **Weather-related adjustments to farms' nitrogen quotas in 2005**



	Coarse sand	Fine sand	Clay soils
A	0	0	0
B	0	-5	-5
C	0	-5	-10

Source: Danish Plant Directorate, *www.pdir.dk/Files/Filer/Nyheder/Presse/2005/N-prognose_2005_Kort_pl07-531_b2_16.03.05.pdf*. Farms in areas B and C were obliged to reduce their application of nitrogen 5 or 10 kg per ha on fine sands and clay soils respectively in 2005.

Linked to the nitrogen quota system has been a continued increase in the assumption used in the quota calculations concerning how much of the nitrogen in manure is available for uptake by the plants. These assumptions affect how much manure farmers may dispose of on their fields. For pig slurry, it was in Action Plan II *e.g.* assumed that plants can use 75% of the nitrogen contained therein, while it was assumed that 70% of the nitrogen from manure from cattle can be used by the plants. Action Plan III included a further general tightening of the requirements for utilisation of nitrogen in livestock manure by 4.5 to 5 percentage points, concurrently with research creating a basis for this. This modification was expected to reduce nitrogen leaching by approximately 2 900 tonnes nitrogen. Costs for industry in this connection is expected to be in the order of DKK 50-90 million (approximately EUR 7-12 million) per year [see the Danish Government *et al.* (2004)].

The Danish accounting system also includes a possibility for farmers to have their nitrogen quotas increased somewhat if they can document (through invoices from sold crops) that their yields per area unit are consistently higher than the assumptions used in the calculation of the national norms. These claims have to be verified by certified experts.

There are substantial fines for exceeding the quotas – DKK 20 (approximately EUR 2.7) per kg N in significant cases. This is *e.g.* 4-5 times higher than the price paid per kg N in artificial fertilisers.

This nitrogen accounting system causes considerable administrative costs, both for public authorities and for each farmer. A lot of efforts have been put in to estimate optimal nitrogen applications for different combinations of crops and soil types, also taking into account the type of crops that were cultivated on each field in the preceding year. In isolation, this tend to lower the adjustment costs of the farmers, as they will make fewer changes in their production patterns than if uniform application rates were applied. Most farmers nevertheless hire outside consultants to prepare the nitrogen accounts and the fertiliser- and crop rotation plans.

However, as from 2005, the farmers receive *pre-filled nitrogen accounts* from public authorities – based on information collected in previous years and on information collected from wholesalers of feed and fertilisers, slaughter houses, etc. Farmers will henceforth be able to correct any errors and return the accounts electronically. At the same time, the data collection process has been co-ordinated with the application procedure for subsidies under the EU *Common Agricultural Policy*, which now is also being pre-filled and sent to each farmer for verification.[20]

3.6.1.3. Regulation on the livestock density

As mentioned, the EU *Nitrates Directive* sets limits on how much *nitrogen from animal manure* can be applied per area unit (170 kg N per ha). The Directive allows member states to implement this through a limit on animal numbers. Danish authorities have therefore set maximum limits on the livestock density on farms. Until 1998, the maximum limit for cattle was 2.3 animal units per ha, for pigs it was 1.7 animal units per ha and for other animals it was 2.0 animal units per ha. These maximum limits have since been decreased, and are now 1.7 animal units per ha for cattle and 1.4 animal units for other animal categories.[21]

Similarly to the Dutch system of manure contracts, Danish farmers that have more animals than allowed by the land they own (or lease though long-term contacts) may enter into agreements with other farmers, so that the surplus manure can be spread out on their fields. In part due to a lower livestock density in Denmark than in the Netherlands, trading in such contracts do, however, seem to less common here.

3.6.1.4. Purchase of nitrogen quotas

In 2002, farmers were given the option to sell part of their nitrogen quota for a 5-year period – through an auctioning system. The idea was that by only accepting the cheapest bids from the farmers, one would obtain environmental improvements at the lowest possible cost to society as a whole. To be eligible to participate in the bidding, farmers had to have utilised more than 50% of their quotas previously, and they had to offer at least a 10% reduction in their respective quotas. It was, however, not a requirement that the farm be situated in an environmentally sensitive area.

It is emphasised that what the farmers offered to sell was a reduction in their *nitrogen quotas* – not a reduction in their *actual nitrogen application*. If a farmer did not utilise a considerable share of his legal quota at the outset, he could sell (some of) the unused part, without any impact on either the actual nitrogen application or his yields.

This instrument was discontinued as from 2004 – in part due to the high budgetary costs per unit reduction in nitrogen leachate obtained (see Section 7.9 for a discussion).

3.6.1.5. *Research to promote better utilisation of animal food*

Changes in feeding techniques reduced nitrogen leaching considerably during *Action Plan II*, seemingly at relatively modest costs (mostly research and development costs). There were also some costs stemming from the fact that the improved feeding reduced the nitrogen content in animal manure – which made it necessary for some farmers to purchase (more) mineral fertilisers. Further improved feeding is also expected to contribute to significant low-cost reductions in nitrogen leaching in *Action Plan III*.

3.6.1.6. *Subsidies for wetlands creation*

The *Action Plans for the Aquatic Environment* include several area-related instruments – primarily payments to farmers for committing to make lasting changes in their area usage.

One of these instruments is a subsidy for wetlands creation, where farmers under Action Plan II were given a *one-off* payment of (on average) approximately EUR 7 000 per ha for permanent conversion of former agricultural land into wetland areas.[22] The reduction in the agricultural land area leads to a lower total nitrogen application, and the wetlands created constitute buffer zones that limit nitrogen run-off to coastal areas, etc.

Action Plan III includes slightly increased payments per ha for conversion of agricultural areas to wetlands and an additional reduction of nitrogen losses of more than 1 000 tonnes was expected to be achieved in this way in 2004 and 2005, at a total cost of close to EUR 3 million, or slightly below EUR 3 per kg N loss reduction [see Jacobsen (2004)].[23]

3.6.1.7. *Subsidies for forest creation*

Another area-related instrument applied in the Action Plans is subsidies for conversion of agricultural land to forests. Farmers receive about EUR 3 350 per ha as a one-off payment, plus an *annual* income compensation of about EUR 335 per ha over 10 years. A third of this is financed by the European Union.

According to Jacobsen (2004), the total cost of this instrument in relation to Action Plan II was about EUR 4.7 million; with a cost per kg reduction in nitrogen leaching equalling almost EUR 6, making the instrument one of the *least* cost-effective of all the instruments applied. Nevertheless, the focus on afforestation is further enhanced in Action Plan III, with an *annual cost over a 10-year period* of approximately EUR 28 million being expected to lead to an additional 900 tonnes nitrogen loss reduction, or more than EUR 30 per kg.

3.6.1.8. *Subsidies for Environmentally Sensitive Areas*

When *Action Plan II* was established, approximately 360 000 ha of agricultural land were designated as Environmentally Sensitive Areas. The area was later increased to about 900 000 ha – or about a quarter of the entire agricultural area. In these areas, additional economic support can – in accordance with the EU *Common Agricultural Policy* – be given in return for commitments by the farmer to improve environmental performance. Such agreements can comprise several different approaches – *for example*, commitments by farmers to restrict nitrogen fertilisation to 60% of the agronomically optimal amounts, or a 20 years set-aside for agricultural land. According to Jacobsen (2004), the cost-effectiveness of these measures as instruments to reduce nitrogen leaching was relatively low. Nevertheless, several additional such measures were included in *Action Plan III*.[24]

3.6.1.9. Subsidies for organic farming

Like in other EU member states, subsidies are available for organic farming. The average size of these subsidies in Denmark was about EUR 125 per ha under *Action Plan II*. The effect on this measure on nitrogen leaching has been estimated to 33 kg N per ha, giving a cost of EUR 3.8 per kg N in reduced leaching. The cost-efficiency of this measure was thus neither among the highest nor among the lowest of the instruments applied under *Action Plan II*.

The area used for organic farming increased, in any case, less than expected during *Action Plan II* – and is expected to *decrease* slightly over the course of *Action Plan III*. This is *inter alia* due to a surplus of organically produced milk on the market.

3.6.1.10. Tax on fertilisers

There is also a tax on mineral fertilisers in Denmark. However, this instrument only covers *non-agricultural use* of the fertilisers. Farms that are subject to the system of nitrogen norms and quotas (see earlier description) are all exempted from this tax.

3.6.2. Instrument mixes addressing phosphorous run-off

3.6.2.1. Introduction

As can be seen from Figure 7.7 (see Chapter 7), the phosphorous surplus measured in tonnes was reduced 36% between 1990-92 and 2002-04 in Denmark, while when expressed per unit of agricultural land, the surplus declined 33%, to 11 kg phosphorous per hectare – compared to *e.g.* 19 kg in the Netherlands.

Until recently, no instruments in Denmark specifically addressed run-off of phosphorous from agriculture – although many of the instruments addressing nitrogen run-off obviously also impacted on phosphorous application and leaching. This is, for example, the case for the regulation on livestock density and several of the area-based measures, such as subsidies for wetlands creation, organic farming, etc. These instruments clearly impact on several targets at the same time. It is, however, clear that in the Danish context, the "dosage" of these instruments has been based primarily on the political will to address the nitrogen problem.

As part of *Action Plan III for the Aquatic Environment*, a specific target of a 25% reduction in the phosphorous surplus by 2009 was established. Between 2009 and 2015, an additional reduction of 25% is to be achieved, and the political parties that agreed to the Plan have committed to consider a further tightening of the target, if it appears that technological developments would support this.

3.6.2.2. Phosphorous taxes

In order to help reaching the target of a 25% reduction in the surplus by 2009, a tax on phosphorus was introduced from 1 April 2005. This subsection gives a brief description of some of the tax models that were considered in this context, drawing largely on Larsen (2004). These included models focussing on the *phosphorous content in fertilisers* and models that address the *composition of animal feeds*. As the loss of phosphorus is determined by the stock of minerals in the soil built up over a long time period, the aim of a tax would be a reduction of the accumulation of phosphorus in the soil – not an "immediate" reduction of phosphorus leaching.

It was recognised that a tax on phosphorus in commercial fertiliser would be comparatively simple to administer. Such a tax would provide an incentive to limit application of phosphorus from commercial fertilisers, for example, by paying more attention to the amount of phosphorus already available in the soil and by cultivating crops that require less use of phosphorus. These effects were, however, estimated to be of limited importance [see Ministry of Taxation (2004)].

A tax on phosphorus in commercial fertiliser could, on the other hand, be seen to be "unfair", since farms without access to livestock manure would pay a disproportionate amount of the tax. The environmental effect of such a tax would chiefly derive from the fact that *livestock manure would be transported further away and spread on larger acreages* than at present.

At present, mineral phosphorus is added to animal feed that is lacking in available phosphorus. A second option considered was to introduce a tax directed at these phosphorous amounts. The environmental effect of such a tax would stem from a reduction in the amount of mineral phosphorus added to the feed, as a result of the increased use of phytase instead. The addition of phytase to the feed increases the proportion of phosphorus in the feed that animals can digest.[25]

A tax on *all* phosphorus in feed would also provide an incentive to replace feed phosphate with phytase, but it would in addition reduce the use of feed with a high content of non-utilisable phosphorus. Such a tax could have been levied on the total content of phosphorus in feed mixtures for livestock – but it would not cover feed that is never traded (*i.e.* feed grown on the farmers own fields).

Another option would have been to introduce a tax on phosphorus in feed, combined with a basic deduction corresponding to the natural phosphorus content of corn. This would mean that the natural phosphorus content in animal feed would be more or less exempt from tax. However, such a tax would still involve discrimination between domestic feeds and imported feeds with high phosphorus content.

Still another possibility would have been to tax both phosphorus in feed and phosphorus in commercial fertiliser. However, this might result in some feed being taxed more than once – first through a tax on the phosphorus in commercial fertiliser, next through a tax on the phosphorus content of the feed produced with taxed commercial fertiliser.

In the end, it was chosen to levy a tax (only) on mineral phosphorus added to animal feed. The tax rate was set to DKK 4 (about EUR 0.5) per kg phosphorous. It is too early to say whether the tax has had any significant environmental impacts. In the tax proposal, it was estimated that the tax would engender a reduction in the use of mineral phosphorous feed additives from 9 000 tonnes to around 4 000 tonnes in 2010. It was at the same time assumed that the use of such additives anyway (in the absence of a tax) would decline; from 13 500 tonnes in 2004 to 9 000 tonnes in 2010 [see Ministry of Taxation (2004)].

The reduction in the use of additives that is assumed to take place regardless of the introduction of the new tax is – of course – *partly* driven by the use of other policy instruments that impact on the total number of animals, like the tightening of the regulation of livestock density, conversion of agricultural land into wetlands, changes in the EU *Common Agricultural Policy*, etc.

3.6.2.3. Crop-free buffer zones along rivers and lakes

In addition to the tax on mineral phosphorous, about 30 000 ha of 10-metre *crop-free buffer zones* along rivers and lakes are to be established before 2009, and a further 20 000 ha are planned before 2015. Such zones will retain phosphorous from other areas and protect banks along rivers and lakes. The zones are to be established through voluntary land set-asides; additional subsidies under various agri-environmental measures will be introduced. For the period 2005-09, DKK 375 million (approximately EUR 50 million) in additional subsidies will be provided for this purpose – in part financed by the European Union.

3.6.3. Instrument mixes addressing pesticides use[26]

3.6.3.1. Introduction

Problems related to pesticides in groundwater – the main source of drinking water in Denmark – have for a few decades been high on the political agenda. A first *Action Plan* on pesticides was issued in 1986, with a goal being to reduce the use of pesticides 25% by 1990 and by an additional 25% before 1997. Further, a target was set that the "treatment frequency" should decrease 50% by 1997.[27] It was also a goal of the Plan to change the composition of pesticides use – in the direction of the least harmful products.

As part of the Plan, the rules regarding the approval of pesticides were tightened. The toxicological impacts of each of the products were to be considered. Large farms were required to keep a spraying log, farmers were obliged to have a spraying certificate, and the use of pesticides in environmentally sensitive areas was banned. In addition, a tax of *3% of the wholesale value* of pesticides was introduced – which hardly could be expected to have a major impact on farmers' behaviour.

As can be seen from Figure 7.11 (see Chapter 7), while the number of tonnes active ingredients of pesticides decreased significantly following the issuing of the *Action Plan*, the treatment frequency showed major variations from year to year, but did not decline much all in all. This led to a significant increase in the pesticide tax (*cf.* Section 3.6.3.3).

An evaluation of the first *Pesticides Plan* led to the setting-up of the Bichel Commission, tasked with considering overall impacts of a total or partial phasing-out of all pesticides use in Denmark. The Commission presented its findings in 1999, which *inter alia* concluded that the treatment frequency could be reduced 30 to 40% over a 5 to 10-year period *without any major negative economic impacts* for the farmers. This is in part explained by the finding that many farmers use more pesticides than what is agronomically optimal from a long-term perspective – possibly in an attempt to reduce the risk of production losses in individual years (*e.g.* if weather conditions should prove unfavourable).

The proposals of this Commission then led to the publication of *Action Plan II on Pesticides* in 2000, with a target of reducing the treatment frequency to 2.0 by the end of 2002, primarily through improved training and advice to the farmers, and through the establishment of "target values" concerning the use of pesticides for different crop categories. Emphasis was also placed on reducing pesticide pollution in connection with the washing of spraying equipment, etc. The overall budgetary cost of this Plan was EUR 30-35 million in total over the period 2000-03.

A third *Action Plan on Pesticides* was adopted in 2003, covering the years 2004-09. It sets as a target that the *treatment frequency be reduced to 1.7 by the end of 2009*. The main elements of this Plan are presented in the next section.

3.6.3.2. *The Pesticide Plan 2004-09*

A starting point for the third *Action Plan on Pesticides* was that the treatment frequency had been reduced to 2.04 at the end of 2002.[28] About 8 000 ha spraying-free zones along rivers and lakes had been established, and about 180 000 ha were cultivated as organic farming. An update of the estimates of the Bichel Commission had confirmed the original finding that large reductions in pesticide use could be achieved without significant income losses for farmers.

On this basis, it was (as discussed above) set as a target that the treatment frequency should be reduced to 1.7 by the end of 2009. The main instrument assumed to bring about this reduction is *increased advisory activities* at the farm level, focusing on farms that have not previously taken part in such activities. The aim is to achieve a better diffusion of existing knowledge among farmers. The budget for the period 2004-09 is approximately EUR 20 million in total.

A reduction in the use of pesticides is also being stimulated through the subsidies for organic farming (see Section 3.6.1.9) – even if the total area used for organic farming is expected to decrease. Subsidies are now *also* given for areas cultivated according to organic principles related to pesticides and fertiliser use, even if the farmer has not obtained certification for organic farming. Approximately EUR 30 million will be allocated for this purpose annually, with 50% being financed by the European Union.

3.6.3.3. *Tax on pesticides*

Several different tax models were considered when an increase in the pesticides tax was prepared in the mid-1990s. Ideally, the tax-base should be related as closely as possible to the environmental damage done by the different pesticide products. However, an exact match is impossible to achieve in practice.

One option that was considered was to let the tax vary with the amount of active ingredients in the pesticides. Such a tax base would be easy to administer – but the link with the environmental effects would be rather weak. This is in part because a number of low-dose products have come on the market over recent decades, with potentially larger negative environmental impacts *per kg or litre* than older, high-dose, products.

Another option would be to levy the tax according to the standard dose for each pesticide. A later expert group concluded that it *could* be possible to define unambiguously the standard dose for each product, but the group also meant that the environmental impacts of using such a tax-base would only be marginally better than if the tax was based on the value of each product, *cf.* Ministry of Taxation *et al.* (2001).

Eventually, it was decided to *relate the tax to the price of each product* – which is obviously relatively remote from the harm they cause to the environment and/or human health. The tax rates differ between different main categories of pesticides (*cf.* Table 3.5).

Table 3.5. **Tax rates in the Danish pesticide tax**

Pesticides category	Tax rate
Insecticides and soil disinfectants	54% of retail price, excluding tax
Herbicides, fungicides and growth regulators	33% of retail price, excluding tax
Wood preservatives, algaecides, rat poisons and microbiological agents, etc.	3% of wholesale value, including tax

There are a number of reasons for such a differentiation. First, there are substantial cost differences per treatment with the various agents. By imposing a higher rate on the cheap agents than on the expensive ones, one can approach the effects of a tax on each treatment. Second, there are differences in the degree to which different types of pesticides are disseminated in nature, and therefore how damaging they are to the environment.

Over the period 1996-2005, annual revenues from the tax varied between EUR 35 and EUR 60 million. These revenues are recycled back to the agricultural sector, in part through a reduction in the tax on the value of land, in part through special funds administered by farmer's organisations, and in part through R&D related to organic farming.

3.6.3.4. Agreement on the phasing-out of pesticide use on public areas

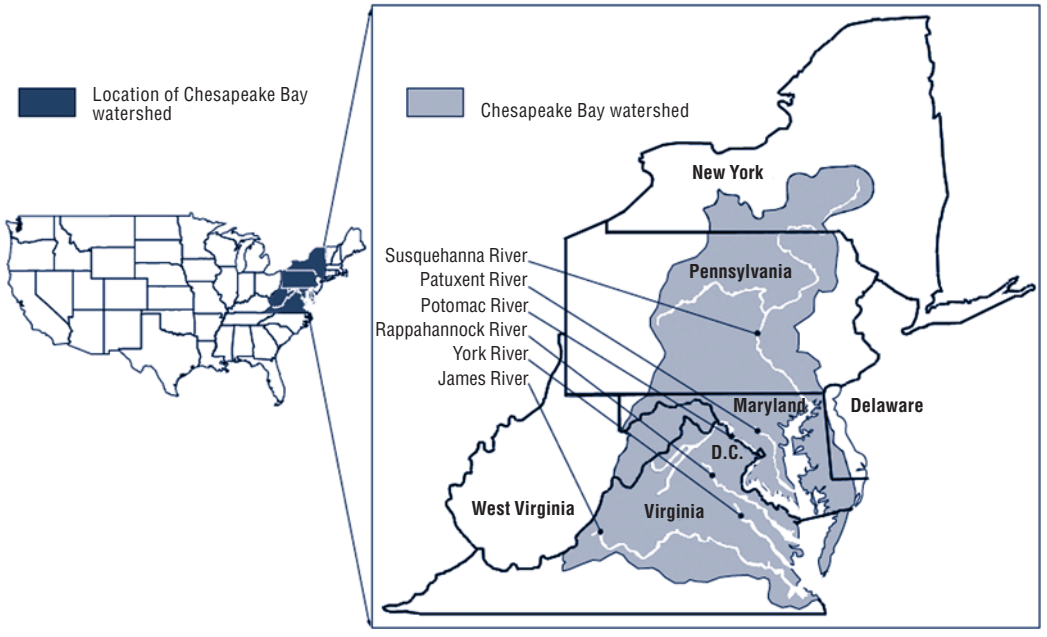
In 1998, public authorities in Denmark agreed to phase out the use of pesticides on all public property by 2003. The aim of the agreement was to reduce the risks of biodiversity loss and groundwater pollution.[29] Although this agreement does not address the agriculture sector directly, it is mentioned here because several studies indicate that the costs to society as a whole of its implementation are very high compared to those associated with alternative instruments.[30]

3.7. Instruments addressing nutrients run-off in the Chesapeake Bay area in the United States

3.7.1. Introduction

The Chesapeake Bay is the largest estuary in North America, situated on the East coast of the United States. Its watershed covers approximately 165 000 km^2, including parts of the states of Delaware, Maryland, New York, Pennsylvania, Virginia and West Virginia and all of the District of Columbia (*cf.* Figure 3.6). The Bay is relatively shallow, with an average depth of only around 7 meters. Roughly half of the Bay's water comes from the Atlantic Ocean,

Figure 3.6. **The Chesapeake Bay watershed**



Source: Based on GAO (2005).

while the other half is freshwater that drains from the land and enters the bay through a large number of rivers and streams. With a watershed land to bay water ratio seven times that of any other major estuary in the World, the Chesapeake Bay must process run-off from a large amount of land with a relatively small body of water [see CBP (2006a)].

The Bay, home to more than 3 600 species of plants, fish, and shellfish, is considered a national treasure in the United States, but its water quality has been deteriorating for several decades. The Bay is designated by bordering States as an "impaired water body" because the water quality is inadequate to support the uses designated for the water under State implementation of the *Clean Water Act*. According to the Blue Ribbon Finance Panel (2004), in its current state, the Bay supports less than half the underwater grasses that were there in 1950, and the population of the estuary's primary filter, the oyster, has fallen to two per cent the levels from the middle of the 20th century.

Figure 3.7 gives an illustration of the geographical distribution of the total loads of phosphorous that reaches the Chesapeake Bay – from *all* sources.

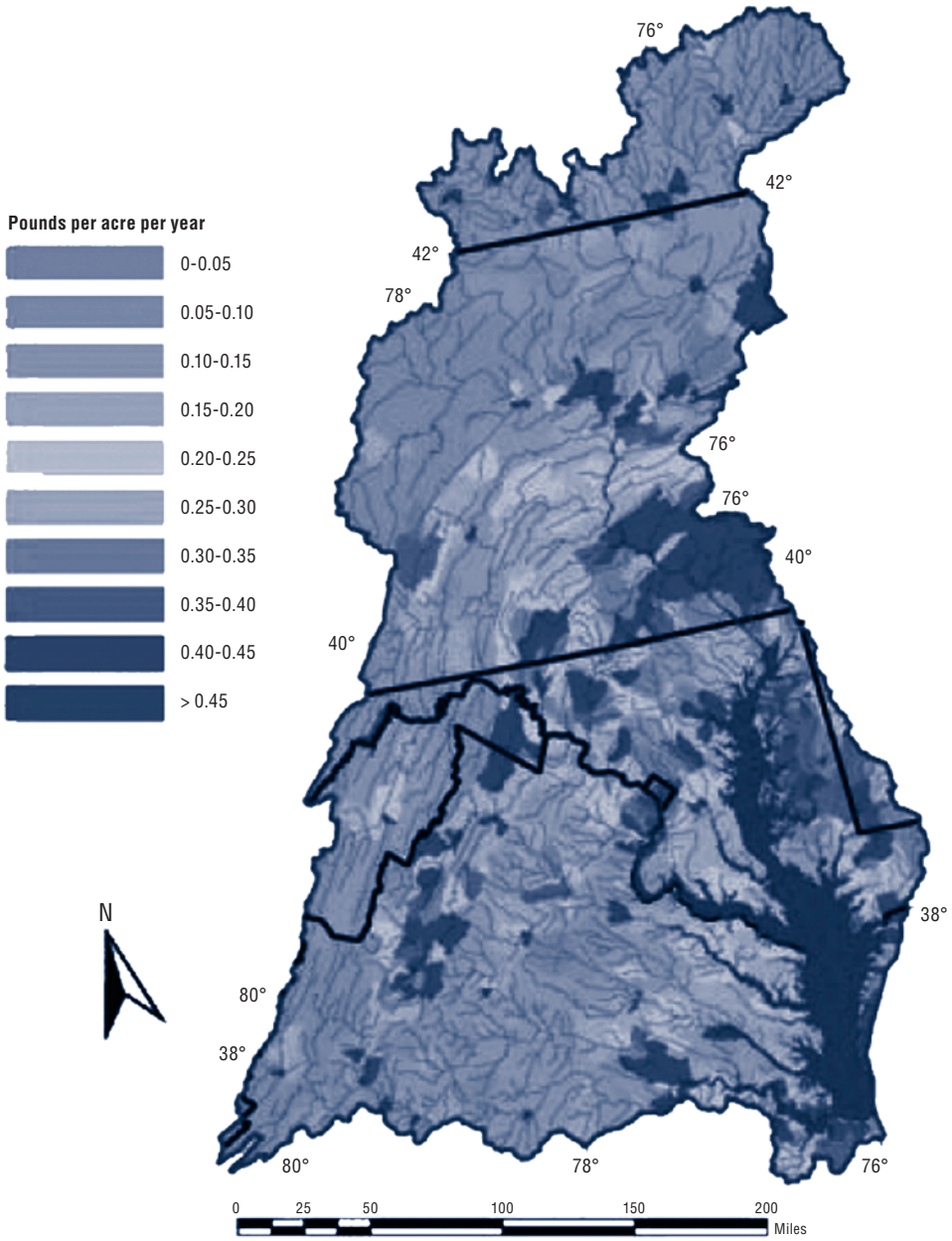
Since the early 1980s, Federal authorities and the affected States have co-operated on improving the water quality in the Bay, signing the *Chesapeake 2000 Agreement* in June 2000, through which total upper limits on nitrogen, phosphorous and sediment loads to the Bay were distributed between the affected States. Total nitrogen pollution is to be reduced from about 285 million pounds per year in 2000 to no more than 175 million pounds in 2010. In the same time interval, phosphorous loads are to be reduced from 19 to 12.8 million pounds. Compliance with these "caps" are due to be achieved (and maintained "ever after") by 2010 the latest – if not, "Total Maximum Daily Load" (TMDL) limits for the Bay as a whole would have to be introduced by Federal authorities as from 2011.[31]

A TMDL is a calculation of the maximum amount of a pollutant that a water-body can receive and still meet water quality standards, and an allocation of that amount to the pollutant's sources. Water quality standards, which are set by States, US territories and tribes, identify the uses (*e.g.*, drinking water supply, swimming, aquatic life support) for each water-body and the scientific criteria to support that use. This process can sometimes lead to complicated negotiations among dischargers to a water-body and other stakeholders.

In order to co-ordinate actions taken at different administrative levels, a special *Chesapeake Bay Program Office* has been established, situated in Annapolis, Maryland. Efforts to improve the water quality in the Bay are hampered by the strong economic development in the region – with the population having doubled over a few decades, now totalling 16 million persons. The *Chesapeake Bay Program* (CBP) partners include the States of Maryland, Pennsylvania and Virginia; the District of Columbia; the *Chesapeake Bay Commission* (a tri-state legislative body); the US Environmental Protection Agency (EPA) (representing the Federal Government); and participating citizen advisory groups. The *Chesapeake Bay Program*, which has been underpinned by extensive scientific research and advanced modelling over many years, addresses *all sources* of nutrient loads, such as wastewater treatment plants, run-off from urban developments, air depositions (representing up to one-third of total nitrogen loads[32]), and other non-point sources.

Whereas the *Chesapeake Bay Program* addresses all sources of pollution, the focus of *this case study* is only on the policies in the area used to address *nutrients* run-off from agriculture, with some comments on the links between agricultural sources and other sources of pollution. Brief descriptions are given of the instruments that are applied at the Federal level and in two of the States in the area – Virginia and Maryland.

Figure 3.7. **Total amounts of phosphorous reaching the Chesapeake Bay from all sources**



Source: Chesapeake Bay Commission (2004), based on information from US Geological Survey.

3.7.2. Instruments applied at the Federal level

The main Federal instrument addressing water pollution in the United States is the *Clean Water Act*, first enacted as the *Federal Water Pollution Control Act Amendments* in 1972. Whereas agriculture is regulated by several other statutes that include provisions for water quality, for the most part, the *Clean Water Act* does *not* regulate agriculture activities – with the exception of large *Confined Animal Feeding Operations* (CAFOs).

Hence, various types of voluntary approaches coupled with financial and/or technical assistance are the only types of federal instruments used to address run-off from non-point sources in agriculture, both specifically to the Chesapeake Bay – and more generally.

As part of a nationwide approach, the EPA announced a new strategy to help the agriculture industry address environmental issues in May 2006.[33] The strategy sets forth a framework in which to accomplish given environmental goals "… by:

- increasing the awareness and understanding within EPA of agricultural impacts and benefits to human health and the environment and working across media in a more systems approach to environmental protection;
- working with the agricultural sector – including production, processing and distribution – in developing and demonstrating environmental protection solutions that express the value of farmland environmental stewardship activities to the public;
- co-ordinating research and technology development and transfer so the needs of agriculture and EPA can be more efficiently met; and
- identifying existing environmental measures and developing new ones, where appropriate, to demonstrate environmental improvement related to agriculture."

The US Department of Agriculture is managing most of the assistance programmes addressing run-off from agriculture, *inter alia* to the Chesapeake Bay.[34] Among them is the *Conservation Security Program*, which provides financial and technical assistance to promote the conservation and improvement of soil, water, air, energy, plant and animal life, and other conservation purposes. The programme provides 5-10 years of payments to producers for maintaining or adopting structural and/or land management practices. Producers can participate at one of three tiers; higher tiers require greater conservation effort and offer higher payments. The lowest-cost practices that meet a given conservation standard must be used. Annual payments range from USD 20 000 for Tier I; USD 35 000 for Tier II and USD 45 000 for Tier III. Cropland must have been cropped in 4 of the 6 years prior to 2002, in order to be eligible for participation.

Another relevant programme is the *Conservation Reserve Program* (CRP), which includes the *Conservation Resource Enhancement Program* (CREP). CREP provides incentive payments for agricultural landowners to install specific conservation practices. Farmers can receive annual rental payments and cost-share assistance to establish long-term, resource conserving covers on eligible land. This Program offers additional financial incentives beyond the Conservation Reserve Program, to encourage farmers and ranchers to enrol in 10-15 year contracts to retire land from production.[35]

The *Environmental Quality Incentives Program* provides incentive payments and cost-share assistance to implement conservation practices. A plan of operation is developed in conjunction with the producer, identifying appropriate conservation practice or practices to address resource concerns. These practices are subject to technical standards developed by the National Resource Conservation Service and adapted for local conditions. The local conservation district has to approve the plan of operation.[36]

Agricultural Management Assistance is a voluntary program that provides cost-share assistance to agricultural producers to voluntarily address issues such as water management, water quality, and erosion control, by incorporating conservation into their farming operations. Producers may construct or improve water management structures or irrigation structures; plant trees for windbreaks or to improve water quality; and mitigate

risk through production diversification or resource conservation practices, including soil erosion control, integrated pest management, or transition to organic farming. The landowners must agree to maintain cost-shared practices for the life of the practice, and the contract periods are for 10-15 years. The Federal cost-share is 75% of the cost of an eligible practice. Participants are paid based upon certification of completion of the approved practice. The programme is only available in 15 States,[37] including all the States in the Chesapeake Bay watershed. Funds are distributed to the applicable States using an allocation formula based on a large number of environmental factors.

The *Resource Conservation and Development Program* is meant to improve the capability of State and local units of government (and local non-profit organisations) in rural areas to plan, develop and carry out programs for resource conservation and development – through the provision of technical and financial assistance. The projects and activities are determined by the areas' problems and needs, and the areas are sponsored locally, but designated by the Secretary of Agriculture. By 2004, 315 areas across the country had been designated as RC&D areas, covering 73% of the US counties.

3.7.3. Instruments applied in the State of Virginia

Virginia has an estimated surface area of more than 108 000 km^2, of which more than 102 000 km^2 is land area. Land used for agricultural purposes accounts for approximately 24% of the State's total area, while urban land uses constitute another 7%. Farm animal manure and commercial fertiliser are two major sources of nutrients in the State. Manure and litter from Virginia's farm animals contained over 110 000 tonnes of nitrogen and more than 35 000 tonnes of phosphorus in 1997. In 2003, about 86 000 tonnes of nitrogen and about 21 500 tonnes of phosphorus were sold in commercial fertilisers in the State [see JLARC (2005)].

In order to operate a "Confined Animal Feeding Operation" in Virginia, a Virginia *Pollution Abatement General Permit* must be obtained. Prior to being issued a permit, the facility operator must submit a Nutrients Management Plan (NMP) to the State's Department for Conservation and Recreation (DCR) for approval, outlining how the facility will "address the efficient use of nutrients".

For other agricultural activities in Virginia, various subsidy programmes are used to promote the use of Nutrient Management Plans.[38] The *Agricultural Best Management Practices Cost-share Program* reimburses farm managers for funds expended to implement certain Best Management Practices (BMPs),[39] including the development and writing of *Nutrient Management Plans*.[40] The Program is administered by DCR through the 47 local Soil and Water Conservation Districts. The State pays up to 75% of the cost of each BMP, a flat amount based on acreage, or a combination of the two (not to exceed USD 50 000 per operator). State funding for this program is prioritized by watershed, based on the greatest potential effect. Because some BMPs offer greater potential nutrient or sediment reductions than others, funding is also prioritized by the type of BMP. Between 2000 and 2004, Nutrient Management Plans covering almost 32 000 acres were written using the Cost-share Program [see JLARC (2005)].

Poultry litter has higher phosphorus content than animal manure, in part because chickens and turkeys do not process the phosphorus components of their feed as well as cattle and hogs. This has a substantial impact on where and how litter can be used. In general, poultry litter contains the same amount of nitrogen as phosphorus. However, plants use only about half as much phosphorus as they do nitrogen. As a result, while a

crop may use all of the nitrogen available to it from poultry litter, it will leave behind about half of the phosphorus. This "leftover phosphorus" will either bind with the soil, at which point it becomes a candidate for loss from erosion, or, if the soil is already highly saturated with phosphorus preventing binding with the soil, the excess phosphorus becomes a candidate for loss by means of run-off.

In light of the problem of excess phosphorus production in poultry, DCR and the Virginia Poultry Federation jointly funded a pilot litter transport project in an attempt to entice farm operators outside of the Shenandoah Valley to accept poultry litter produced in the Valley for field application. Initially, the Program was designed to pay for as much as 16 000 tonnes of poultry litter to be transferred. However, over the two years the program was operational (2003-04), only about 5 000 tonnes was actually moved [see JLARC (2005)].

Farm operators can also receive tax credits for adopting certain agricultural best management practices. The programme allows a tax credit against the State's individual income tax of up to 25% of the first USD 70 000 expended to adopt a BMP. Almost 600 tax returns were filed in 2001 and another 760 returns were filed in 2002, claiming the credit [see JLARC (2005)].

The State also offers a tax credit for the purchase of "advanced technology pesticide and fertiliser application equipment". To be eligible to claim this tax credit against income tax, an operator must already have in place an approved Nutrient Management Plan. The Program also provides incentives for replacing older applicators with more technologically advanced spreading equipment that can better address lower spreading rates. Almost 140 claims were filed for this credit in 2001, and almost 90 were filed in 2002 [see JLARC (2005)].

The State is also promoting a more wide-spread use of phytase in order to limit phosphorous pollution, having allocated about USD 1 million in matching grants to poultry companies in Virginia to add phytase to their feed programs.

According to the Virginia *Agriculture Stewardship Act*, complaints regarding major pollution can be filed with the Agriculture Commissioner. Staff of the Department of Agriculture then checks the facts and first try to solve any issue co-operatively, but can also issue orders, etc.

An additional "link" between efforts to limit run-off from agriculture to efforts in other sectors is being established in Virginia in connection with a new system of general permits for nutrient loads that are being introduced. Any *new point sources*, and any *expansions* in existing point sources will have to buy offsets for the nutrient loads they will create, and farmers will be able to sell such off-sets – if they implement control measures that go beyond a baseline that is to be defined.[41]

3.7.4. Instruments applied in the State of Maryland

As opposed to most other States, including Virginia, the development of NMPs is *obligatory* for all farms in Maryland that have a turnover higher than USD 2 500. As in Virginia, the Plans previously focused on the nitrogen requirements of the plants, but since 2005 the Plans are based on (the more stringent) phosphorous needs. The almost universal obligation of having a NMP in place also means that any manure transported away from a "Confined Animal Feeding Operation" (a CAFO) only can be applied on farms where such Plans exist.

The application of the NMP system in Maryland has gradually become stricter over time. The requirement to prepare such Plans is based on the State's (1998) *Water Quality*

Improvement Act – in response to widespread fish-deaths in the Chesapeake Bay. Farmers originally resisted the new law strongly, which caused major implementation problems. After a "nutrients management summit" in 2003, some simplifications in the procedures, requiring less paper work of the framers, were introduced – and "real enforcement" of the Nutrients Management Plans started in June 2005. The NMPs are valid for maximum 3 years, after which time new soil samples have to be made. And the Plans that are prepared have to be *implemented*. Farmers have to prepare annual implementation reports – and follow-up letters to those that had not yet done so are sent out. If the farmers do not react within specified time limits, they can be fined.[42]

In addition to the NMPs, a *Manure Transport Program* helps poultry, dairy, beef and other animal producers cover the costs of transporting excess manure off their farms. Animal producers with high soil phosphorus levels (or inadequate land to spread their manure) can receive cost-share assistance of up to USD 20 per tonne to transport excess manure to other farms or alternative use facilities that can use the product safely.

In 2004, this Program provided farmers with USD 295 356 in State grant payments to transport more than 40 000 tonnes of manure away from areas with high soil phosphorus levels – an increase of more than 25% over 2003.

Maryland also has a *Manure Matching Service* that links farmers who have excess animal manure with nearby farmers or alternative use projects that can use the waste as a nutrient source. The goal of the service is to reduce the potential impact from animal waste run-off, by establishing a marketplace where farmers can sell their excess manure to buyers who need the nutrients it contains for crop production or alternative use business ventures. The service is free and available to both sending and receiving operations. The service is also designed to foster new markets for manure suppliers, by encouraging the development of alternative animal waste management technologies, such as waste-to-energy, fertiliser manufacturing, and composting.

The laws of Maryland state that no-one has the right to pollute, and strict enforcement measures can be taken in cases of *serious* pollution stemming from agriculture. If such cases are discovered, representatives of the Maryland Department of Agriculture first try to make the farmers change their behaviour voluntarily. If this does not work, the Department for the Environment prepare court actions (in many cases this triggers the desired changes), which can lead to fines and – in rare cases – prison sentences.

Notes

1. Austria, Belgium, Canada, Czech Republic, Germany, Greece, Italy, Japan, Korea, New Zealand, Norway, Poland, Sweden, Switzerland, Turkey, the United Kingdom and the United States.
2. Countries were only asked to provide *examples* of instruments applied at a state/provincial level, which obviously means that the numbers given for such instruments are underestimates. The judgment that an objective or instrument concerns *both* issues has been made by the OECD Secretariat.
3. The numbers used in this paragraph include instruments applied both at a national and at a State/Provincial level.
4. The assessment for England and Wales is available at *www.environment-agency.gov.uk/wfd*.
5. Regarding nutrients run-off, ammonia and nitrite (NO_2) are List-II substances according to this Directive, in addition to pesticides. *Nitrate* (NO_3) is, however, not included in these lists.

6. Farmers outside the *Nitrate Vulnerable Zones* are also *encouraged* to follow the Codes of Good Practice, for the protection of the environment. The Government explains that "This will help to prevent nitrate levels rising to the point where regulation becomes necessary. It will also help to reduce other pollution, including phosphate losses, microbiological contamination of bathing waters and pesticide losses", *cf. www.defra.gov.uk/environment/water/quality/nitrate/action.htm*.
7. GBP 3.8 million was provided for each of the years 01.04.04-31.03.05 and 01.04.05-31.03.06.
8. Depending *i.a.* on the change in agriculture practise in question, the payments (in 1997) varied between GBP 65 and GBP 590 per hectare. These subsidy amounts can seem *relatively* modest compared to the subsidies provided in Danish schemes described in Section 3.6 below – but the Danish schemes seem to require significantly larger behavioural changes from the farmers' side.
9. As the agreements were of 5 years duration, payments under the scheme lasted until 2003.
10. The EU *Directive on the Placing of Plant Protection Products on the Market Place* also provides for a review of products already on the Community market. According to Pesticides Safety Directorate (2005), "approximately 70 active substances out of the 350 (20%) that have UK approvals have been taken off the market. Within the EU as a whole approximately 450 out of 970 (46%) active substances have been withdrawn".
    Pesticide Safety Directorate (2005) indicates that the reduction in the number of available pesticides can make it difficult for farmers in certain areas to find adequate plant protection products for some specialist crops they could like to grow.
11. HM Treasury (2006) states: "The pesticides voluntary initiative remains in place and the Government continues to consider potential improvements to it."
12. P is less reactive than N, it does not exist in different oxidation states, and is rather insoluable in its dominant natural form in crops, animals, manure, soils and water. 1 kg phosphate (P_2O_5) corresponds to 0.218 kg phosphorous.
13. According to Stolwijk (2005), the high minimum prices for grain in the EU made it possible for Dutch livestock farmers to out-compete farmers in other EU member states. See Section 7.10 for a discussion.
14. 350 kg phosphate per ha per year for maize land, 250 kg phosphate per ha per year for grassland and 125 kg phosphate per ha per year for arable land.
15. The act was initially applicable to cattle, pigs and poultry. In 1992 it was extended to manure from other animals.
16. Different methods for calculating the relevant manure/nutrient amounts were used in the manure quota trading scheme and in MINAS. The approach in MINAS gave farmers some more flexibility to adapt their production methods.
17. In its judgement of 2 October 2003 (see Section 3.5.6), the European Court of Justice found that the Kingdom of the Netherlands had failed to fulfil its obligations under the Nitrates Directive. The Court did not take into account any policy changes that took place after 6 December 1999, see European Court of Justice (2003a). Hence, in practice, MTAS had no effect on the outcome of this judgement.
18. See Government of the Netherlands (2004).
19. Jacobsen (2004) provides an economic evaluation of the instruments applied as part of Action Plan II. The study includes direct cost for the farmers and public expenditures at the state and county level.
    Another report, Jacobsen *et al.* (2004) discusses both the direct costs and the welfare economic costs of a number of administrative and economic instruments in preparation of *Action Plan for the Aquatic Environment III*, building on a large number of scientific and economic analyses. This report looks at a number of different levels of additional nitrogen loss reduction, and takes explicitly into account interactions between different instruments.
20. See Plantedirektoratet (2004).
21. Previously, an animal unit was defined as *e.g.* 1 animal for cattle (which was expected to produce about 115 kg N in its manure), 3 sows (which were expected to produce about 94 kg N in their manure) and 30 slaughter pigs (which were expected to produce about 79 kg N in their manure). In 2002, the definition was changed, to represent for each animal category the number of animals that "normally" would produce 100 kg N in manure. For sows and slaughter pigs, this change in definition made the density regulation less strict.
22. There are, in fact, significant variations in the payments made per kg estimated nitrogen loss reduction – because the payments are based on the estimated income losses of the farmers.

23. The cost estimate in Jacobsen (2004) reflects *annual costs over a 10-year period* and includes expenditures by both national and county authorities to stimulate wetlands development. National authorities provided DKK 140 million altogether in 2004 and 2005, while co-financing of approximately DKK 80 million was expected from county authorities.
In these calculations, the reduction in nitrogen losses is set at 100 kg N per ha converted to wetlands – as opposed to 265 kg N per ha for the areas converted under Action Plan II – reflecting the fact that areas less suitable for conversion would have to be included.

24. The fact that a major part of these costs are financed by the EU increases the cost-efficiency of these instruments from a *national* point of view. In addition, the subsidies can provide other environmental benefits, in addition to their contribution to reduced nitrogen leaching.

25. Phytase is a product used to make it easier for animals to utilise the phosphorous in the feed.

26. The discussion here draws on *inter alia* Schou (2005), Larsen (2004) and Det Økonomiske Råd (2004). The latter study includes an *ex ante* cost-benefit analysis of possible additional instruments to address water pollution in Denmark, building on simulations using several different economic, biological and geological models.

27. The "treatment frequency" expresses the number of times the total of arable land in Denmark can, on average, be treated with the sold quantities of pesticides when they are used at the normal dose rates.

28. The method of calculating the treatment frequency has since been changed. According to the old method, the frequency was 2.04 in 2002 and 2.17 in 2003. According to the new method – used in Figure 7.11 – the frequency was 2.1 in 2002, 2.33 in 2003 and increasing further to 2.39 in 2004.

29. According to Ministry of Environment and Ministry of Food, Agriculture and Fisheries (2003), the use of pesticides by public authorities decreased from 28.8 tonnes active ingredients in 1995 to 6.3 tonnes in 2002. In 2002, less than 0.2% of all Danish pesticide use took place on public areas.

30. See, for example, Environment Assessment Institute (2004).

31. Figure 7.9 in Chapter 7 illustrates developments in nutrient loads from agriculture in the different States in the area.

32. Including nitrogen that results from atmospheric deposition to agricultural lands, urban areas, forest lands, and tidal waters around the Bay, and which then runs-off or flows into the Bay.

33. See *www.epa.gov/agriculture/agstrategy.html*.

34. A 2003 report from the US General Accounting Office on the protection of erodible croplands and wetlands found that, in the process of selecting samples of cropland tracts to assess farmer's compliance with relevant support schemes that existed in 2002, the Natural Resources Conservation Service disproportionately emphasised tracts with little potential for non-compliance, leading to inflated overall compliance rates. The report also indicated that the US Department of Agriculture agency responsible for withholding benefits for violations that had been identified often waived these non-compliance determinations without adequate justification [*cf.* GAO (2003)].
GAO (2006a) indicates that the US Department of Agriculture provided almost USD 500 million between 1995 and 2004 in "indirect funding" for the Chesapeake Bay Program, where "indirect" means that the funding was part of broader agency efforts and/or would continue without the Bay restoration effort.

35. US EPA and USDA (2006) states that the Farm Service Agency (FSA) (the US Department of Agriculture agency responsible for running *i.a.* the Conservation Reserve Program) recognized the Chesapeake Bay watershed as a conservation priority area and devoted additional resources to it. As a result, the acreage in the Chesapeake Bay watershed devoted to FSA's CRP has grown by 177 000 acres, or more than 90%, at an approximate 10-year cost of USD 287 million, of which USD 122 million was for CREP rental and incentive payments in the Chesapeake Bay watershed.

36. According to US EPA and USDA (2006), about 2 000 *Environmental Quality Incentives Program* applications and 1 500 other conservation programme applications in the six States comprising the Chesapeake Bay watershed were *not* funded in fiscal year 2005. The same source indicates that there are approximately 87 000 farms in the watershed.
Cattaneo (2003) is one of the few *ex post* assessments of USDA conservation programmes that have been found in this project. That study examined the reasons why, among the EQIP applications approved to the end of fiscal year 2000, in 17% of the cases, one or more of the planned conservation practices were not carried out. In that period, farmers were "bidding" in an auction, competing in being willing to accept a low cost-share compensation for entering into a contract. The study found that there could be a "moral hazard" problem, in that the farmers had an incentive to include more practices in their bids than they intended to carry out, in order to

increase the probability of having their bid accepted. Once a contract was signed, they had an incentive to refrain from carrying out the conservation practises that generated the lowest private benefits to them – and the National Resources Conservation Service had neither the capacity nor the will to pursue such breaches of contracts. The study also indicates that changes made to the EQIP programme from 2002 should reduce these "moral hazard" problems.

37. Connecticut, Delaware, Maine, Maryland, Massachusetts, Nevada, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Utah, Vermont, West Virginia and Wyoming.

38. Chesapeake Bay Commission (2004) discusses "Cost-effective Strategies for the Bay". The report points to broad-scale coverage of agricultural lands in Nutrients Management Plans as one of the most cost-efficient measures that could be taken, and indicating that the potential for annual additional reduction across the whole watershed at the maximum feasible level of implementation is 13.6 million pounds nitrogen (at an average cost of USD 1.66 per pound) and 0.8 million pounds phosphorous (at an average cost of USD 28.26 per pound).

39. See US EPA and USDA (2006) for a description of environmental impacts of different BMPs.

40. Until recently, these Plans were based on the plants' nitrogen requirements. However, new Plans are based on the phosphorous requirements of the plants – which, in practice, leads to stricter limits on the amount of nutrients that may be applied to a given field.

41. The State of Pennsylvania is also developing a trading system for point and non-point sources. In Pennsylvania, existing sources will be allowed to purchase offsets to cover some of the obligations they presently have – *i.e.* their participation will not be limited to expansions of their current activity.

42. Much of the information in this paragraph was provided in a meeting with Maryland Department of Agriculture in May 2006.

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Chapter 4

Instrument Mixes Addressing Residential Energy Efficiency

This chapter reviews instrument mixes dealing with residential energy efficiency in the United Kingdom and in Canada. In the case of Canada, attention is paid to both Federal and Provincial instruments. The focus in both country studies is on the thermal energy efficiency of buildings and on the energy efficiency of household appliances.

4.1. Introduction

Residential energy consumption represents approximately 20% of total final energy use in the OECD on average, *cf.* IEA (2006), and contributes to variety of adverse environmental impacts, including local, regional, and global air pollutants, and the generation of hazardous and non-hazardous solid waste.[1] However, the share of the residential sector varies significantly among OECD countries, representing for instance 14% of total final energy consumption in Japan and 28% in the United Kingdom.

Heating and cooling of dwellings and water heating are the single largest contributors to residential energy use, representing as much as 80% of total residential energy demand in OECD member countries. According to Schaefer *et al.* (2000), space heating alone accounts for 70% of total residential energy demand in most countries – corresponding to a 15-22% share of total final energy demand.

4.2. Combining instruments to improve residential energy efficiency: an overview

In addition to price-setting for electricity,[2] three types of instruments are in general used in combination to address this particular environmental issue: direct regulations (*e.g.* standards), information-based measures (*e.g.* labels), and financial support programmes. However, a broader range of instruments may be applied by governments as summarised in Box 4.1. While economic instruments can be economically efficient and environmentally effective instruments, their use remains limited in instrument mixes applied by policy-makers to address residential energy efficiency.

Applying incentive-based measures to reduce domestic energy consumption is, however, receiving increasing interest in some of the member countries selected for case-study. The use of economic instruments to improve household energy efficiency is being extensively discussed in the United Kingdom (HM Treasury, 2003). Consultations have been carried out on the role of economic instruments to improve household energy efficiency and economic incentives, such as reduced VAT rates for specific energy-efficient products and a *Landlord's Energy Saving Allowance*, have been introduced. In a similar way, the scope for economic instruments in environmental policy is under consideration in Canada and some initiatives have already been taken to introduce financial incentives in the area of residential energy efficiency. The very recently announced *ecoENERGY Retrofit* incentive and the previous *EnerGuide for Houses Retrofit Incentive* are examples.

An overview of latest changes in the instrument mixes applied by the United Kingdom and Canada to improve residential energy efficiency is provided in Box 4.2.

4.3. Instruments addressing residential energy efficiency in Canada

4.3.1. *Instruments applied at the Federal level*

At the level of the Federal Government, several policy instruments are used in combination to increase residential energy efficiency in Canada. These include:

Box 4.1. **Main categories of policy instruments to address residential energy efficiency**

Energy-related taxation and price-setting

- Taxes on the use of energy products for residential energy (electricity, oil, gas, solid fuels) and price-setting (electricity and natural gas).
- Taxes on energy-using appliances.

Information-based measures

- Measures targeted at households:
  - For instance: energy labels for specific appliances, information campaigns on energy conservation, certification programmes concerning the thermal efficiency of dwellings.
- Measures targeted at firms:
  - For instance: demonstration projects for construction firms and developers, guidance on technological options for appliance manufacturers.

Financial support programmes

- Targeted preferential loans for households, grants and tax exemptions for the purchase of energy conservation equipment.

Direct regulation

- Performance and technical standards for appliances (*e.g.* refrigerators, lighting, space and water heating).
- Performance and technical standards associated with the thermal quality of new or existing dwellings.

Voluntary measures

- Industry-level agreements concerning thermal standards or appliance efficiency

Box 4.2. **Recent changes in the instrument mixes in the United Kingdom and Canada**

The mix used in the United Kingdom to address residential energy efficiency combines a number of different approaches including: direct regulation, information-based instruments and economic instruments. Recent changes in the mix of instruments applied include the following:

- Implementation of the EU Energy Performance of Buildings Directive (EU 2002/91/EC). Measures include the revision of the *Building Regulations* in 2005 to raise the energy efficiency standards of new and refurbished buildings.
- Introduction of the *Landlord's Energy Saving Allowance* in 2004 to provide all private landlords who pay income tax with upfront relief on capital expenditure for installations of insulation. In addition, the introduction of a "green landlord scheme" targeted at private landlords is considered.
- Extension to 2008 of the *Energy Efficiency Commitments* (EEC) under which electricity and gas suppliers are required to achieve targets for installing energy efficiency equipments/appliances in the household sector.

In Canada, information-based measures, performance standards, voluntary approaches and technical and financial support measures are combined to improve residential energy efficiency. Some recent modifications in the mix used include:

- Announcement in January 2007 of the ecoENERGY Efficiency Initiative which will start in April 2007. This package is a mix of information-based instruments (labelling) and financial incentives designed to encourage homeowners to retrofit their homes, businesses and industry and make them more energy efficient (Natural Resources Canada, 2007a). The ecoENERGY Retrofit grant for homes replaces the EnerGuide for Houses Retrofit Incentive introduced in 2003
- Proposed amendments to the *Energy Efficiency Act* of 1992 to strengthen and broaden the energy-efficiency standards of consumer products.
- Working with Provinces and Territories to update the Model National Energy Codes and support their adoption.

Information-based measures (*e.g.* EnerGuide Equipment labelling programmes), Performance and technical standards (*e.g.* Minimum energy efficiency standards for equipment), Technical assistance, Financial support programmes and Voluntary measures (*e.g.* the R-2000 Home Standard). Though energy-related taxation is not applied to improve residential energy efficiency, financial incentives targeted at improving the energy efficiency of houses and buildings are used.

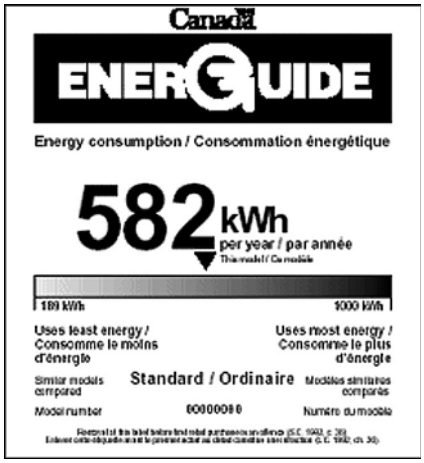
4.3.1.1. *Information-related instruments*

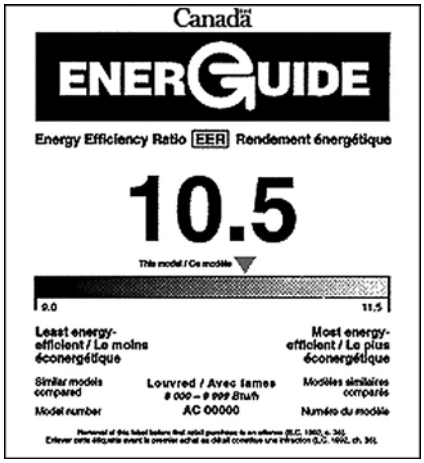
A number of measures have been implemented at the federal level to overcome market barriers, such as inadequate information and knowledge. The main information-based measures introduced in Canada include: *EnerGuide for Equipment*, the *Energy Star* symbol, *EnerGuide for Houses and the R-2000 standard for new construction*.

The amendments to the *Energy Efficiency Act* proposed by the Government in October 2006, as part of the *Clean Air Act*, were subsequently accompanied by complementary measures under the ecoENERGY initiative.

EnerGuide for Equipment and HVAC. This is a rating, labelling and information initiative to promote the production, purchase and use of energy-efficient major household electrical appliances and heating, ventilating and air-conditioning equipment. The EnerGuide label shows how much energy major appliances use in order to easily compare models of the same size and class. This programme supplements the minimum performance requirements and seeks to stimulate consumer awareness. It is supported through directories for consumers, information and education campaigns, and training for retail sales staff.

Major electrical household appliances and room air conditioners sold in Canada must meet minimum energy efficiency standards and are required to display an EnerGuide label. Information on the EnerGuide label is the result of extensive testing, based on Canadian Standards Association (CSA) test procedures. The EnerGuide rating is an average measure of how much energy individual appliances typically consume when used at different temperature and/or speed settings. The two types of EnerGuide labels are presented below.





EnerGuide ratings for major household appliances range from the most energy-efficient to the least energy-efficient in a given size and type category. For major appliances, an indicator arrow on the left side of the EnerGuide scale means lower operating costs and long-term savings.

The Energy Star symbol. In 2001, the EnerGuide for Equipment was teamed up with the international Energy Star symbol to help consumers find the most energy efficient refrigerators, dishwashers, clothes washers and room air conditioners on the market. A growing list of products in Canada qualify for the Energy Star symbol for energy efficiency including clothes washers, refrigerators, freezers, dishwashers, lighting, heating/cooling and ventilation and consumer electronics.

This symbol (see below) identifies the most energy-efficient products in their class. To be able to display the Energy Star symbol, products must meet or exceed technical specifications designed to ensure that they are among the most energy efficient in the marketplace. Requirements vary from one category to another, but, typically, Energy Star products are 10 to 50% more efficient than the minimum regulated standard in Canada. See Box 4.3 for further details.



Box 4.3. **International Energy Star symbol and Canadian minimum regulated standards – Examples**

Clothes washers: as from 2004, standard-size clothes washers must be at least 36% more efficient than the minimum federal energy performance standard in Canada's Energy Efficiency Regulations to qualify for the ENERGY STAR mark.

Refrigerators: Standard-size refrigerators in 2004 must be at least 15% more efficient than the minimum federal energy performance standard in Canada's Energy Efficiency Regulations to qualify for the ENERGY STAR mark compared to 10% previously.

Freezers: ENERGY STAR qualified standard-size freezers must exceed minimum Government of Canada energy efficiency standards by at least 10%. Compact freezer models must exceed minimum Government of Canada energy efficiency standards by at least 20%.

Dishwashers: Dishwashers must exceed the minimum Federal energy performance standard in Canada's Energy Efficiency Regulations by at least 25% to qualify for the ENERGY STAR mark.

4.3.1.2. Performance and technical standards

The *Energy Efficiency Regulations* establish energy-efficiency standards for a wide range of energy-using products, with the objective of eliminating the least energy-efficient products from the Canadian Market. Given the division of responsibilities between the Federal and Provincial levels which characterises the Canadian institutional context,

federal energy-efficiency standards apply to prescribed energy-using products imported into Canada or manufactured in Canada and shipped from one province to another for the purpose of lease or sale.

Regulations under the *Energy Efficiency Act* introduced in 1995, and subsequently amended, set minimum energy-performance levels for a number of energy-using products such as appliances, lighting, and heating and air-conditioning products.

The Government has proposed amendments to these Regulations which will come into force between 2007 and 2010, making them broader and more stringent. New Regulations under the amended *Energy Efficiency Act* will set a minimum energy performance standard for twenty additional products and increase the stringency of the standards for 10 products that are currently regulated.

Natural Resources Canada (2007b) estimated that the amendments will have an impact equivalent to eliminating the energy use of all households for one year in a city with approximately 130 000 households.[3]

4.3.1.3. Financial support/incentives

ecoENERGY Efficiency Initiative. The Government of Canada announced in January 2007 the introduction of the *ecoENERGY Retrofit Grant* that will offer Canadians financial incentives to retrofit their homes to make them more energy efficient. Under *ecoENERGY Retrofit – Homes*, the Government expects to provide incentives to some 140 000 homeowners for home improvements that will reduce their energy use and costs as well as make their homes more comfortable.[4]

A previous initiative introduced by the Government of Canada to encourage homeowners to make energy efficiency improvements in their home was the *EnerGuide for Houses Retrofit Incentive* (EGHRI). That programme was launched in 2003, and discontinued as of May 2006.

To be eligible for an EGHRI grant, homeowners needed to have had the *EnerGuide for Houses* evaluation and to have followed up on recommendations. Information about who will be eligible for the *ecoENERGY Retrofit* grants is not yet available.

The average *ecoENERGY Retrofit* grant is expected to be more than CAD 1 000 and to yield an average 30% reduction in energy use and costs. The financial support offered under the EGHRI depended on the difference between the energy efficiency rating of a home before the retrofits and after. The non-taxable grants provided by Natural Resources Canada were ranging between CAD 216 and CAD 4 348.

Extended Residential Rehabilitation Assistance Programme (RRAP). In addition, the Government of Canada, through Canada Mortgage and Housing Corporation (CMHC), introduced two Federal initiatives in 2004.

The *Extended Residential Rehabilitation Assistance Programme (RRAP)* offers financial assistance to low-income homeowners to bring their homes up to minimum health and safety standards. Energy efficiency improvements are now directly considered as eligible to receive the grant for the rehabilitation of the housing while this was not the case until recently.

Reduced mortgage loan insurance for energy-efficient homes. The second initiative introduced in 2004 by the Government of Canada through CMHC, Canada's national housing agency, is the adoption of a refund on the mortgage loan insurance premium for energy-efficient homes.

Under this measure, CMHC offers a 10% refund on its mortgage loan insurance premium when a borrower buys or builds an energy-efficient home or makes energy-saving renovations to an existing home. The CMHC *Mortgage Loan Insurance for Energy-Efficient Homes* is applicable to both owner-occupied and rental properties through participating financial institutions when a borrower obtains a new mortgage or refinances an existing one. CMHC's mortgage loan insurance products and policies already contributed to promoting energy efficiency in homes by recognizing lower heating costs in the calculation of the size of mortgage a borrower can carry, but this measure significantly expands this role.

To qualify for CMHC's premium refund, a home's energy efficiency must be rated using Natural Resources Canada (NRCan) EnerGuide for Houses rating system and meet certain minimum requirements. For new constructions, an EnerGuide for New Houses energy advisor must rate the building plans before the home is built. A new home that is R-2000 certified (see below), or that has an EnerGuide rating of 77 or above, is eligible for a CMHC premium refund. CMHC's premium refund is also applicable to rental properties (apartment buildings) to encourage developers and owners to construct new energy-efficient buildings or retrofit existing buildings to make them more energy-efficient.

In addition to the measures listed above, which have been introduced by the Government to provide financial support to households to make energy-efficient retrofits, the scope for the use of economic instruments and the introduction of incentives in environmental policy is being considered in Canada.[5]

4.3.1.4. Voluntary instruments and Guidelines

Model National Energy Code for Houses (MNECH). This Model Code specifies minimum performance standards for new Canadian houses. A similar model code, *the Model National Energy Code for Buildings*, sets the level for larger commercial and institutional buildings. The development of the MNECs in 1997 was supported by NRCan in collaboration with energy utilities, provincial and territorial governments, and the National Research Council Canada (NRC).

The 1997 MNECs requirements are based on life-cycle costing in a given area, taking into account such factors as climate, energy costs and construction costs. The energy codes requirements are based on actual regional values rather than on national average values. It allows designers and contractors to choose the degree of energy efficiency that is appropriate for specific climates.

The MNECs do not set compulsory national standards for buildings but they are used as reference documents by Provinces/Territories which have jurisdiction for housing and buildings; they can individually adopt it or amend it. The MNECs were in particular used during the consultations under the National Climate Change Process in 1999-2000 to provide basis for considering measures to improve the energy efficiency of the Canadian housing and building stock.

Agreements between federal organisations and provincial or territorial organisations of housing sometimes also require a reference to the *Model National Energy Code for Houses*. For instance, the current memorandum of understanding for the administration of a housing infrastructure project in the far north Nunavut territory requests that houses be built to perform 25% better than the *Model National Energy Code for Houses*.

The R-2000 Standard.[6] The R-2000 energy efficiency standard encourages the building of energy-efficient houses. It is based on an energy consumption target for each house and a series of technical requirements including for insulation and choice of materials. The requirements are about 40% above building codes and R-2000 homes use at least 30% less energy than conventional new houses.

The R-2000 is a voluntary industry-led standard which was developed through a consultation process between industry and the government. R-2000 homes can be built only by builders trained and licensed by the Government of Canada. Approximately 900 builders are licensed today. Natural Resources Canada's Office of Energy Efficiency (OEE) provides a certificate that guarantees the energy efficiency characteristics of each R-2000 home.

In 2005-06, the R-2000 Standard was delivered in eight provinces (British Columbia, Alberta, Manitoba, New Brunswick, Newfoundland and Labrador, Nova Scotia, Ontario and Saskatchewan) and in Yukon Territory. Approximately 12 000 "R-2000 homes" have been built and certified. R-2000 demonstrates that highly energy efficient homes can be built by the private sector for customers who are willing to pay for an independently certified level of performance. R-2000 has supported diffusion of energy-efficiency knowledge and practices in house construction.

The R-2000 standard is used as a criterion of eligibility to benefit from various policy measures aimed at improving energy efficiency of houses. For instance, to qualify for the refund on a mortgage loan insurance premium offered by the CMHC for homeowners who borrow money to build or buy an energy-efficient home or renovate an existing one, the home's energy efficiency must be rated using the EnerGuide for Houses rating system or be R-2000 certified. (An R-2000 home rates at least 80 points on the EGH scale.)

The R-2000 voluntary energy efficiency standard has been influential for advancing energy performance in housing. This voluntary standard played a specific role in the institutional context characterising Canada, where the responsibility for housing lies with Provinces and Territories. The R-2000 standard comes as a complementary instrument in the instruments mix used in Canada to improve residential energy efficiency, as the Federal Government has a role in providing guidelines, but where setting up national minimum energy performance standards for homes is beyond its scope of responsibilities.

4.3.2. Instruments applied at the Provincial/Territorial level

In Canada, Province/Territories have jurisdiction over the energy sector and a variety of measures are adopted at that level to reduce domestic energy use. Table 4.1 lists some examples.

Initiatives taken at the Provincial level to improve residential energy efficiency range from the adoption of minimum energy efficiency standards for houses to the introduction of financial incentives for energy efficient products.

Rebates and incentives are provided for selected Energy Star qualified products in some provinces and territories. For instance, British Columbia offers exemption from provincial sales tax (7%) from February 2005 to April 2007 for qualifying residential furnaces, boilers and heat pumps. Other initiatives include the provincial sales rebate for Energy Star qualified refrigerators, freezers clothes washers and dishwashers purchased on or after October 2003 offered by the Government of Saskatchewan.

Table 4.1. **Examples of Provincial and Territorial Energy Efficiency Policies in Canada**

Nova Scotia	Regulations on energy standards for new buildings
Québec	Expansion of the product range falling under efficiency standards and labels
Ontario	Canada's largest number of products regulated for minimum energy efficiency levels
Saskatchewan	Creation of energy conservation offices
Alberta	Newly created Energy Efficiency Office to deliver information on efficiency measures
British Columbia	Regulation of energy efficiency through a provincial Energy Efficiency Act and the British Columbia Building Code
Yukon, Northwest Territories, Nunavut	Wide range of energy-efficiency support measures to involve all stakeholders

Source: IEA (2005) Compiled from Canadian Energy Efficiency Alliance, National Report Card on Energy Efficiency.

Measures may also be introduced at a more local level (*e.g.* municipalities). Thus, city-specific measures exist like the city of Toronto's incentive on the purchase of eligible water efficient clothes washers or the city of Victoria's incentive on the purchase of eligible high-efficient clothes washers.

The *Model National Building Code* and the *Model National Energy Codes* developed at the Federal level under the National Research Council provide guidelines to the Provinces and Territories. The provinces have jurisdiction over the code and can decide whether they adopt it or not, and to which extent they want to modify it. For instance, though the Model National Building Code has no explicit energy-related provision, some provinces have added some energy requirements (*e.g.* Alberta, Ontario).

Different energy-efficiency labels are applied in combination in Canada. The Energy Star symbol is used together with the EnerGuide for Equipment labelling. The two labels are meant to work together to help consumers make informed choices when purchasing household appliances or heating and cooling products. Energy Star generally goes one step further and identifies specific models that meet a higher level of energy efficiency. The Energy Star symbol is sometimes used on the EnerGuide logo.

More indirect measures complement instruments targeted at landlords to address landlord/tenant failures, such as improving the energy efficiency of social housing for rental. One activity is the acceleration of the replacement of inefficient energy equipments by energy-efficient equipments in the non-profit social housing.[7]

4.4. Instruments addressing residential energy efficiency in the United Kingdom

4.4.1. *Introduction*

For OECD as a whole, natural gas is the most important source of energy in households, with electricity in second place. In United Kingdom, the fuel mix differs considerably from the OECD average, with natural gas playing an even more important role than what is the case in most other countries.

Household energy efficiency in United Kingdom has been improving steadily. New homes typically require only half as much heating fuel as the average building stock, and new appliances are more efficient. The energy efficiency of the existing housing stock has also been rising, as new central heating systems are installed and heating systems upgraded, and insulation is improved.[8] Yet, against this backdrop of rising efficiency, the demand for energy continues to rise, due to a growing number of households and higher levels of "energy services" (*e.g.* warmer homes, more appliances, etc.).

The instrument mix applied in the United Kingdom to address residential energy combines information-based instruments, energy efficiency standards and financial support measures which are presented in further details below.

As in the case of household waste, environmentally-related taxes is one important potential instrument that is *not* part of the instrument mix addressing residential energy efficiency in United Kingdom: whereas a *Climate Change Levy* is levied on energy use in other parts of the economy, the household sector is exempt.[9] On top of that, a lower-than-normal VAT rate applies to domestic energy use. Hence, public authorities have *not* sought to limit households' energy use through direct application of price-based instruments.

A reason given for exempting the residential sector from taxes on energy is the fear of aggravating the situation for households in "fuel poverty", *cf.* Chapter 7 for a discussion.

4.4.2. Performance and technical standards

Minimum energy efficiency standards for appliances: Stricter energy efficiency standards have also been introduced for some household appliances.[10]

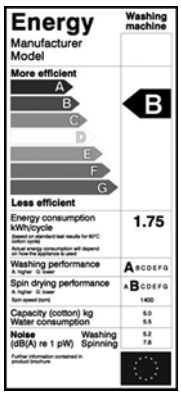
In the United Kingdom, minimum energy standards for appliances are complemented by energy labelling schemes. Minimum energy efficiency requirements for refrigerators, freezers and combinations are for instance used along with the European Union energy label (A to G rating).

Among the direct regulations, the Building Regulations were revised in 2005 to raise the standards of new and refurbished buildings.

The United Kingdom currently uses energy standards for buildings in combination with building certification and the government intends to further develop complementarities between the two instruments as Building Regulations are revised in 2005 and the European Union Energy Performance of Buildings Directive (EU, 2002) is implemented in the UK. Energy surveys and certification of all buildings will be introduced to transpose the EU Directive and the provision of a Home Information Pack will be requested.[11]

4.4.3. Information-based instruments

In order to provide information to households on the energy usage of household appliances, the United Kingdom combines EU energy efficiency label – that rates products from A (the most efficient), down to G (the least efficient) – with the *Energy Efficiency Recommended* logo promoting the most energy efficient products managed by the Energy Saving Trust (EST).





In addition, partly in response to EU (2002a), to inform household on the energy efficiency of their dwellings, from 2007 anyone selling their home will be required to put together a *Home Information Pack* for potential home buyers, which will bring together the key information as regards the quality of the house.[12] This measure is meant to make the home buying and selling process more transparent and thereby reduce transaction costs – and *inter alia* provide home buyers with a better picture of the future heating expenditures they can expect.

4.4.4. *Fiscal support and incentives*

To encourage landlords to invest in the energy efficiency of their properties, the Government introduced the *Landlords' Energy Saving Allowance* (LESA) in 2004. The scheme provides private landlords who pay income tax with upfront relief of up to GBP 1 500 per property per annum on capital expenditure for the installation of loft insulation, cavity wall insulation and solid wall insulation in residential property which they let.

In addition, a *Green Landlord Scheme* is being prepared. In this connexion, the Government intends to reform the existing "Wear and Tear Allowance" for landlords by making it conditional on the energy efficiency level of the property. The qualification of a property for this allowance could also be linked to a forthcoming system of Energy Performance Certificates. This package of instruments is meant to increase landlords' awareness of the importance of investing in energy efficiency and to provide an incentive for them to take action to improve the quality of their property. The exact design of the *Green Landlord Scheme* will be discussed further with stakeholders, *cf.* HM Treasury (2005b).

Besides, the Government has reduced VAT to 5% on certain energy-saving materials installed in the home. The UK government is also providing financial support to research, development and demonstration (RD&D) into new energy efficient technologies.

4.4.5. *Other subsidy schemes*

The *Energy Efficiency Commitments* (EEC) is a main mechanism applied to increase energy efficiency of existing homes in the United Kingdom. Under the EEC, electricity and gas suppliers are required to achieve targets for installing energy-efficiency measures in the household sector. For the period from 1 April 2005 to 31 March 2008, the overall target is an improvement in energy efficiency of 130 fuel-standardised, lifetime-discounted TWh; *cf.* The Stationary Office (2004). Approved measures include installing insulation or supplying low-energy light bulbs, high efficiency appliances or boilers. At least 50% of the energy savings must be focussed on a priority group of low-income consumers in receipt of certain benefits and tax credits/pension credits.

Another important subsidy scheme – closely related to the Energy Efficiency Commitments – is *Warm Front*. This is a major component of the UK Government's Fuel Poverty Strategy – which aims to "eliminate fuel poverty in England by 2016, and to eliminate it in vulnerable groups by 2010 as far as practicable". Under the instrument, grants to improve energy efficiency are provided to vulnerable people in owner-occupied homes or in private rented accommodation.

The *Energy Efficiency Commitments* and *Warm Front* are discussed in-depth in Chapter 7.

Notes

1. In 2004, transport represented around 33.6% of total final energy consumption in OECD countries, followed by industry (23%), residential (19%), other sectors (14%) and non-energy use (10%), *cf.* IEA (2006).
2. And to price-setting for natural gas in some countries.
3. For further details, see Natural Resources Canada (2007b).
4. For further details, see Natural Resources Canada (2007a).
5. See for instance the work of the National Round Table on the Environment and the Economy (NRTEE) and the conclusions of the "smart" regulation panel (Government of Canada, 2004).
6. For more details, see: *http://r2000.chba.ca/Links/index.php*.
7. For instance, by reducing the amortisation period for energy using equipments.
8. According to HM Treasury (2005a), the energy efficiency rating of properties is measured by Standard Assessment Procedure (SAP), with SAP ratings going from 1 (least energy efficient) to 120 (most energy efficient). The average energy efficiency rating in the domestic sector was 50.6 in 2003. The mean SAP rating of new dwellings is about 90, but the average level of properties is pulled down by the lower energy efficiency levels of existing stock.
9. For a discussion of the *Climate Change Levy*, see OECD (2005b).
10. For example, refrigerators placed on the market now have to qualify for an energy efficiency rating of "B" or better, *cf.* further description of the labelling scheme below.
11. The *Home Information Pack* will present an energy report for the sale of all residential properties. This measure is expected to make the home buying and selling process more transparent and thereby reduce transaction costs.
12. See DCLG (2006) for further information.

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Chapter 5

Instrument Mixes Addressing Regional Air Pollution

This chapter describes instrument mixes dealing with regional air pollution, focusing on the instrument mixes used to limit SO_2 and NO_x emissions in Sweden and in Canada. The Canadian case study addresses instruments applied at the Federal level and in the Provinces of Ontario and Alberta.

This chapter focuses on measures applied to reduce regional air pollution problems, focussing mainly on acid depositions arising from nitrogen oxides (NO_x) and sulphur dioxide (SO_2) emissions from stationary sources. These two pollutants are the main causes of the acidification problem (along with ammonia). They are also key contributors to smog and fine particulate problems.

NO_x and SO_2 emissions mainly arise from the combustion of fossil fuels, such as coal and oil. In the atmosphere, sulphur dioxide and nitrogen oxides are partially converted into *sulphuric acid* and *nitric acid*, which are to a large extent then returned to the Earth's surface. Because they contribute to acidification, NO_x and SO_2 emissions raise a number of concerns for environmental policy-makers, including: acidification of water (with resultant damage to plant and animal wildlife); soil quality problems that reduce fertility; forest damage; and accelerated corrosion.

Acidification is also a transboundary air pollution problem, because oxides of sulphur and nitrogen can be carried very long distances by the air. This has been a key issue addressed under the *Convention on Long-range Transboundary Air Pollution* (LRTAP) and its related Protocols.[1]

The two main factors that determine which areas are affected by acidification are: *i)* the amount of acid deposition; and *ii)* the sensitivity of ecosystems. Sweden and Canada are two countries that have been significantly affected by acidification, and have therefore developed a range of policy responses to it. This chapter relies heavily on information obtained from these two countries.

5.1. Instrument mixes addressing regional air pollution: an overview

A broad range of instruments may be used to reduce acid deposition resulting from the stationary sources (Box 5.1).

Policy-makers may apply a combination of instruments rather than a single instrument to reduce acid deposition from stationary sources for several reasons, such as addressing market inefficiencies; accounting for the spatial differentiation of impacts; encouraging enforcement by reducing compliance-costs uncertainty for firms; or reducing administrative costs of monitoring. In addition, a reason for the introduction of a mix of instruments (rather than a single instrument) may be to improve political acceptability (*e.g.* competitiveness issues).

An overview of the main instrument mixes in place in member countries suggests that two broad types of policy instruments are generally applied in combination: economic instruments and direct regulation (*e.g.* standards). Pollution taxes on emissions are for instance used with performance standards with emission limit values for NO_x and SO_2.

To reduce SO_2 emissions, a sulphur tax is combined in Sweden and Norway with administrative measures (*e.g.* individual emissions standards); tradable permits are used with performance standards in the United States (Acid Rain Programme) and in the Slovak Republic, as well as at the provincial level in Canada (*e.g.* in Ontario).

Box 5.1. **Main types of policy instruments used to reduce acid deposition from stationary sources**

Economic Instruments

- Input taxes or pollution taxes/charges:
  - For instance, taxes on specific fuels, tax differentiation based on sulphur content, taxes based on actual (or estimated) emission levels.
- Tradable permit systems:
  - For instance, an SO_2 emissions permit trading programme.

Direct regulation

- Performance standards:
  - For instance, restrictions on emission levels (whether expressed in absolute or relative terms), limits to the sulphur content of fuels.
- Technological standards:
  - For instance, the application of specific abatement equipment, restricted use of particular material inputs.

Information-based measures/Technical assistance

- Demonstration projects.
- Technical guidance on options for emission reductions and training programmes.

Financial support programmes

- Support for research and development.
- Targeted grants or preferential loans for the introduction of abatement equipment.

Voluntary Programmes

- Negotiated agreements developed at the sectoral level.
- Public programmes to which firms can volunteer to participate.

Examples of instrument packages adopted by member countries to address NO_x pollution include the combination of a tax and standards in France, of a NO_x charge and emission standards in Sweden and the use of standards (*e.g.* standards on new sources) together with a cap-and-trade programme in the United States.

When applying regulatory approaches, countries often combine different types of standards. For instance, the prescriptive component of the Swedish and the Norwegian package to reduce SO_2 emissions associates emission standards in the permits for major stationary sources with limits on the sulphur content of various fuels (product standards).

In addition, financial support programmes are sometimes introduced. This may negatively affect the efficiency and/or effectiveness of the policy if the instrument is not properly targeted.

5.2. "Aspects" of the environmental issue addressed and the choice of instruments

The nature of the environmental problems to be tackled should of course influence policy choices, including the decision to apply a combination of instruments rather than a single instrument. The issue of acid deposition arising from stationary sources is in particular characterised by a spatial differentiation of impacts.

5.2.1. The spatial dimension

Emissions of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) represent "multi-aspect" environmental problems that can raise concerns of both *local* and *transboundary pollution*.

The sensitivity of the eco-system impacts on the acidification of an area – in addition to the amount of deposition. In addition to monitoring the nationwide NO_x and SO_2 emissions levels, some countries exposed to acidification need to address problems raised at the local level to protect sensitive ecosystems. The notion of "critical load" is used as a measure of how much pollution an ecosystem can tolerate. The southern parts of Sweden are, for instance, quite sensitive to acid deposition. In a similar way, some provinces in Canada have more severe problems of acidification than others, and these geographical differences are reflected in the policy framework for achieving reductions in acid deposition: a cap on SO_2 emissions applies to seven easternmost provinces (Manitoba Eastward).

Acidification is also a *transboundary air pollution* issue. To address acidification in a given member country, reduction in NO_x and SO_2 emissions from neighbouring countries may be necessary (*e.g.* US emissions for Canada, UK emissions for Sweden). This geographical dimension is reflected in international conventions addressing acidification as well as in bi-lateral agreements.

Environmentally related taxes and tradable permits are efficient means of equalising marginal abatement costs across sources, thus minimising total abatement costs to achieve a specific level of emissions. While this is optimal for environmental impacts which are not differentiated according to the place or time of emission (*i.e.* CO_2), this is not necessarily true of regional air pollutants. In such cases, the undifferentiated application of instruments which do not reflect this variation will be sub-optimal.

It may be possible in theory to address spatial and temporal issues within the application of a single policy instrument (*i.e.* spatially differentiated pollution charges). However, in such cases, where one is in fact facing several separate targets, the use of two instruments in combination may be needed.[2]

Many governments use a mix of instruments to address the issue of spatial or temporal variation. For instance, market-based instruments and performance standards are used in combination to address the *total* amount of a certain type of emissions and the *differentiated* environmental impacts according to the place or time of the emission respectively.

Also to address the issue of localised areas of high pollution, or "hot spots", member countries apply a mix of instruments. In the United States, the emissions cap and allowance trading approach to reduce SO_2 emissions of the *Acid Rain Program* is used in combination with minimum performance standards which are differentiated according to place of emission. Specific constraints have also been imposed in some states.[3] A set of ambient and source-specific pollutant standards for SO_2 is enforced, under the *Clean Air Act* and State regulations. In Canada, Ontario's Emissions Trading Program (established in 2001) for trading NO_x and SO_2 is used together with site-specific emissions permits.

5.2.2. Scientific knowledge

Limits on the measurability of emissions and related environmental impacts have will influence policy design.

There are no major technical constraints on the choice of the instruments to reduce emissions from large stationary sources of NO_x and/or SO_2. Facilities may, for example, be

obliged to meet specific emissions rates – like in traditional permits issued at the facility level. Continuous emissions monitoring may also be used to ensure that the mandated reductions of SO_2 and NO_x are achieved – like in the *Acid Rain Program* in the United States.

The situation is different regarding transport-related emissions. The "ideal" tax would be to tax emissions from each vehicle (Feng *et al.*, 2005). The tax rate would vary depending on the level of emission, but would also take into account the location and timing of the emissions. However, the technology is not yet available to measure each car's emissions in a reliable and cost-effective manner. To address spatial/temporal aspects, in addition to introducing fuel taxes or emission standards (fuel, vehicle), governments often use complementary instruments that reduce local transport-related pollution (*e.g.* in large city centre). These measures include congestion charges (*e.g.* London Congestion Pricing), as well as regulatory measures, such as vehicle-use and parking restrictions (*e.g.* access restrictions in Rome).

Further, instrument mixes to reduce acidification arising from NO_x emissions from stationary sources are influenced by technical aspects, such as the nature of the process by which combustion gives rise to nitrogen oxides. In Sweden, for instance, the choice of a charge based on *measured emissions*, rather than a charge based on the characteristics of input fuels (as in the carbon case), has been motivated by the fact that NO_x emissions depend significantly on the combustion process applied.

While there is a good understanding of the environmental effects of SO_2 emissions, the same is not the case for NO_x emissions – due to the complexity of the nitrogen cycle. While sulphate critical loads are used as a yard-stick for assessing acid depositions, scientists are still developing critical loads for nitrogen.

5.3. Addressing acidification: international framework

5.3.1. *International Conventions and Protocols*

Acidification is, as mentioned earlier, a transboundary issue and the need to develop co-operation at the international level has resulted in the signature of a number of Conventions and Protocols.

The UN ECE *Convention on Long-range Transboundary Air Pollution* is the broad international framework of relevance. The Convention was signed in 1979 by 34 Governments and the European Community and it entered into force in 1983. It has since been extended by *eight specific protocols*, some of which are directly relevant for NO_x and SO_2 emissions, such as the *Oslo Protocol* and the *Gothenburg Protocol*.[4]

The 1994 *Oslo Protocol* on *Further Reduction of Sulphur Emissions* entered into force in 1998 and addressed acidification arising from fixed sources. The protocol adopted an effects-based approach, aiming at gradually attaining "critical loads".[5] It set long-term targets for reductions in sulphur emissions, with differentiated emission reduction obligations of the Parties to the Protocol.

The 1999 *Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone* was signed by 27 European countries and entered into force in May 2005. It sets emission ceilings for 2010 for sulphur and NO_x, as well as for VOCs and ammonia. Parties whose emissions have a more severe environmental or health impact – and whose emissions are relatively cheap to reduce – have to make the biggest cuts. Once the Protocol is fully implemented, Europe's sulphur emissions should be cut by at least 63% and its NO_x emissions by 41%, compared to 1990.

The Protocol also sets tight limit values for specific emission sources (*e.g.* combustion plants, electricity production) and requires Best Available Techniques (BAT) to be used. It has been estimated that once the Protocol is implemented, the area in Europe with excessive levels of acidification will shrink from 93 million hectares in 1990 to 15 million hectares. The area with excessive levels of eutrophication will fall from 165 million hectares in 1990 to 108 million hectares.

5.3.2. EU Directives addressing regional air pollution

At the EU level, a number of Directives applied in the area of regional air pollution have an impact on the instrument mixes adopted by member States to reduce NO_x and SO_2 emissions.

One key Directive is the revised *Large Combustion Plants Directive* (LCPD), which places limits on the emissions of SO_2 and NO_x[6] from the combustion of coal, oil and natural gas [see EU (2001)]. While the original LCPD [see CEC (1988)] applied only to *new* plants, the revised LCPD introduces for the first time emission limit values for *existing* coal-fired plants larger than 50 MWh from January 2008.[7]

Another important directive is the I*ntegrated Pollution Prevention and Control Directive* (IPPC) [see CEC (1996)]. This Directive covers (also) *existing* large combustion plants from March 2006 and requires that BAT are applied to the prevention and control of all emissions to air, water and land. The IPPC Directive applies an integrated environmental approach to the regulation of around 7 000 mainly large and industrial installations across many sectors, ranging from small textile units to large power plants. The Directive aims to conserve energy, prevent emissions and waste production and, where that is not practicable, reduce them to acceptable levels.

The *National Emissions Ceilings Directive* (NECD) is also an important Directive in the area of regional air pollution [see CEC (2001b)]. This directive, which came into force in 2001, seeks to reduce emissions of those pollutants that cause acidification as well as those that cause eutrophication and ground-level ozone in order to protect the environment and human health. Its long-term objective is to ensure that pollutant levels remain below their critical loads and critical levels these being the amounts of pollutants below which, significant adverse effects do not occur. For acidification, the interim environmental objective set for the year 2010 is that areas where critical loads are exceeded be reduced by at least 50% against a 1990 base. Towards that end, the Directive sets out national annual emission limits for a number of pollutants.[8]

5.3.3. Bilateral agreements: the Canada – United States Air Quality Agreement

Bilateral agreements are also signed between countries to address acidification problems. The *Canada-United States Air Quality Agreement* signed between Canada and the United States provides an example.

More than half of the acid deposition in eastern Canada originates in the United States. The two countries have co-operated on clean air initiatives since 1980, with the signing of a Memorandum of Intent on transboundary air pollution. This co-operation eventually led to the establishment of a framework for collaboration in 1991, with the Canada-United States Air Quality Agreement. Acid rain has been the primary focus of this agreement, which now also includes new areas of concern such as ground-level ozone (smog) and air toxics.[9]

Under this agreement, Canada and the United States are committed to reduce emissions of NO_x and SO_2 to address acid rain and to implement the acid rain control programs in each country.[10]

Building on the experience of the 1991 Air Quality Agreement, the countries strengthened their co-operation to reduce cross-border air pollution in 2003, under the *Canada-United States Border Air Quality Strategy*. Three joint projects of mutual concerns have been announced under this new framework, to be completed in co-operation with Provincial, State and other stakeholders, including a joint study to explore the feasibility of common emissions trading for NO_x and SO_2.

5.4. Instruments addressing regional air pollution in Sweden: NO_x and SO_2 emissions

5.4.1. *The issue of acidification in Sweden*[11]

Soil acidification has been a major issue in Sweden since the 1980s. Sweden is more sensitive to acid deposition than most other countries. While in many parts of Europe acid deposition is partially neutralised on reaching the ground through the process of weathering (chemical decomposition), even moderate depositions are sufficient to acidify the soil in Sweden. It is only in the interior of Northern Sweden that the acid deposition seems to have had practically no effect on the soil.

Only a minor part of the sulphur and nitrogen deposition occurring in Sweden emanates from pollution sources within the country itself (in a similar way as most of the depositions of mercury stem from sources abroad). The greater part derives from emissions in Central Europe and in the United Kingdom. Disparities exist at the national level, with greater acidity and pollutant content of precipitation in the southern parts of Sweden.

In terms of emissions, the energy sector is the dominant contributor of SO_2 and NO_x emissions. Energy-related sources contribute over 70% of national sulphur oxides (SO_x) emissions and around 95% of NO_x emissions [see OECD/IEA (2004)].

Air quality has improved in Sweden over the past 15 years, with concentrations for the major pollutants having dropped between 1986 and 2002 by 85% for SO_2, and by about 40% each for nitrogen oxides (NO_x) and particulates [see OECD/IEA (2004)].

However, in a recent assessment, the reduction in loadings to Sweden's acid sensitive natural environment will not be enough to repair the adverse effects of acidification within the defined timeframe [see Swedish Environmental Objectives Council (2006)].

New Swedish calculations of exceedance of critical loads – the maximum amounts of deposition which the natural environment can tolerate – indicate that some 13% of the country's lake area will still be receiving too high an input of acidifying pollutants in 2020, despite greatly reduced emissions [see Swedish Environmental Objectives Council (2006)].

Though progress have been made, projections suggest that the decrease of both sulphur and nitrogen oxides has slowed down. In recent years, the air quality index has shown no appreciable change for these emissions [see Swedish Environmental Objectives Council (2006)].

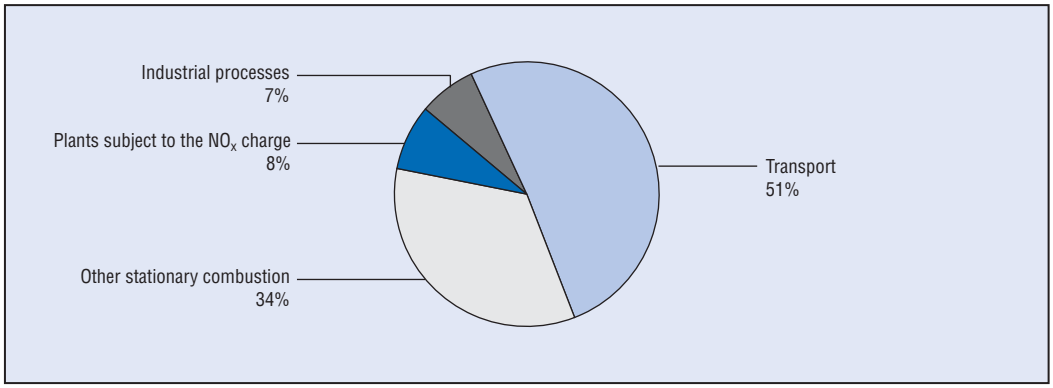
5.4.2. *Addressing NO_x emissions in Sweden*

Swedish emissions of nitrogen oxides are connected to several of the 15 national *Environmental Quality Objectives* adopted by Swedish Parliament in 1999 – such as "Natural

acidification only" and "Zero eutrophication". The acidification objective has an interim target, stating that emissions of NO_x are to be reduced to 148 000 tones per year by 2010. This is in line with the EC *Air Quality Directive* and would mean a 56% reduction compared to 1990 levels.

Emissions of NO_x have been far more difficult to reduce than SO_2 emissions, and Sweden failed to achieve an earlier target of a 30% reduction between 1980 and 1995. The emissions fell from 306 000 tonnes in 1990 to 197 000 tonnes in 2004; see Figure 7.16 in Chapter 7. Stationary sources are responsible for significant NO_x emissions, but half of the emissions derive from the transport sector (see Figure 5.1).

Figure 5.1. **Emissions of nitrogen oxides in Sweden by sector, 2003**



Source: Swedish EPA (2006b).

Swedish policy to address NO_x emissions was originally relying only on a command and control approach. Administrative guidelines were set up for a permit procedure, and permit conditions were tightened over time.

However, in order to achieve a more rapid reduction in emissions than was considered possible by relying only on the administrative guidelines – and to provide an incentive for cost-effective emission reductions – a *refunded charge* on NO_x emissions from energy generation at combustion plants was introduced in 1992.[12]

The charge is levied at a rate of SEK 40 per kg of emitted NO_x (about EUR 4.4). This rate level was based on engineering data on expected effectiveness and costs of abatement investments at electricity power stations and district heating plants. The abatement cost was found to range between SEK 3 and SEK 84 per kg reduced NO_x. A charge of SEK 40 per kg was therefore considered reasonable. The charge has remained constant in nominal terms since introduction.

The charge is based on actual measured NO_x emissions of nitrogen oxides, and is imposed irrespective of the fuel used. This design choice was governed by the nature of the process by which combustion gives rise to NO_x emissions. Nitrogen oxides are formed in two ways: *i)* as a result of a reaction between *atmospheric oxygen* and *nitrogen contained in the fuel*; and *ii)* trough a reaction between *atmospheric nitrogen* and *oxygen*. The significance of the second source of nitrogen oxides depends on the conditions under which combustion takes place, including the precise operating conditions. Direct emission measurement is thus likely to lead to a much more precisely focussed incentive than a charge based on fuel characteristics or other emissions proxies.

The refunding of revenues raised by the charge is linked to the high monitoring costs of NO_x emissions. The charge is applied to measured emissions, or to presumptive emissions levels of 250 mg NO_x per MJ for boilers and 600 mg NO_x per MJ for gas turbines. Plant operators may choose to pay the charge on the basis of presumptive emissions levels instead of installing measuring equipment, although in most cases the presumptive emissions levels are substantially higher than actual emissions, so measurement is generally preferred. The presumptive levels are also applied when the measuring equipment has been out of order, or does not comply with the specifications required [see Swedish EPA (2000)].

Initially, the charge was confined to about 124 combustion plants (182 boilers) producing at least 50 GWh of useful energy per boiler. As monitoring costs went down, the charge was extended to include all boilers producing at least 40 GWh, and in 1997 the limit was lowered to 25 GWh of useful energy per year. Another motivation to lower the threshold was to avoid that some energy producers supply just under the limit to avoid the charge.

Today about 260 plants (460 boilers) are subject to the charge, emitting about 15 000 tonnes of NO_x a year. Most of the liable plants are found in energy production (*e.g.* heating and power plants). The pulp and paper industry, the chemical industry and the metal industry also have combustion plants for energy production – and waste incinerators producing energy are similarly liable for the charge.

Different ways to expand the NO_x charge are being considered. One possible change is to extend the charge to include the emissions from a majority of industrial processes. However, this will require the charge and refund system to be divided into different categories of plants with similar competition conditions. The Swedish EPA proposed that the charge system could be extended to include recovery boilers and sulphide liquor boilers in the pulp and paper industry, and also combustion in industrial processes [see Swedish EPA (2004)].

Another possible development is to increase the charge per kg of NO_x. An increase from the current level to SEK 50 per kg would lead to a decrease in NO_x emissions of more than 5 000 tonnes, according to a 2004 investigation. This is equivalent to a decrease of about 30% in emissions from the boilers subject to the charge.

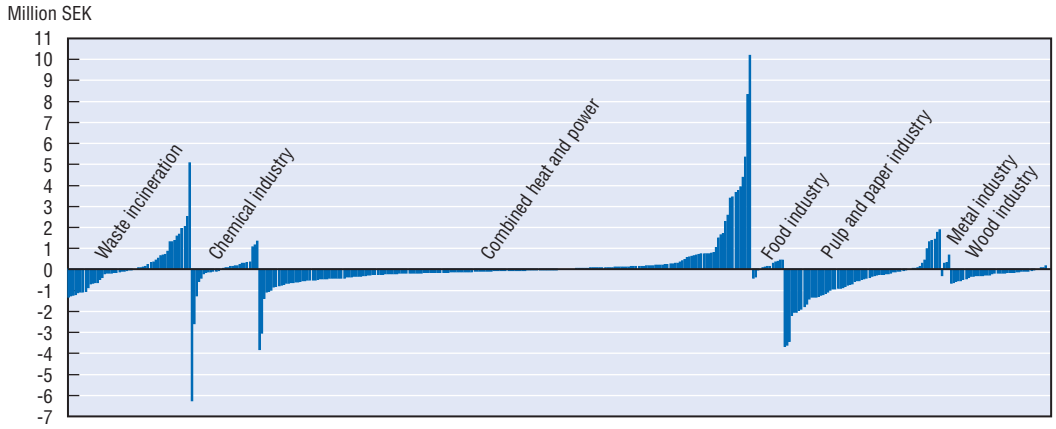
To avoid distorting the pattern of competition between the plants which are subject to the NO_x charge and those that are not, the system is designed so that all revenue (except the cost of administration) is returned to the participating plants, *in proportion to their production of useful energy*. This also avoids the incentives to replace existing equipment with inefficient, smaller, boilers that are not subject to the charge.

Boilers with high emissions relative to their energy output are net payers to the system, and sources with low emissions relative to energy output are net recipients (see Figure 5.2). This feature of the system encourages the targeted plants to reduce their NO_x emissions per unit of energy to the lowest possible level. In 2004, the refund was SEK 8.94 per MWh useful energy (about EUR 1 per MWh).

The total charge amount is returned to the participating plants (with the exception of only 0.7%, which is kept for administration costs). The refund to each plant is proportional to their production of useful energy. This encourages the targeted plants to reduce their emissions of nitrogen oxides per unit of energy to the lowest possible level.

Figure 5.2. **Net payers and receivers in the relation to the refunded NO_x charge in Sweden**

Production units in different industrial sectors, 2004



Source: Swedish Environmental Protection Agency.

5.4.3. *Addressing SO_2 emissions in Sweden*

Since 1970, when Swedish SO_2 emissions peaked at about 925 000 tonnes, the emissions have been reduced by more than 95%, to less than 58 500 tonnes in 2004 (see Figure 7.18 in Chapter 7).[13] Emissions of SO_2 have for the most part resulted from the combustion of coal and fuel oil. Combustion-related emissions have declined heavily in recent years, but still account for more than two-thirds of Sweden's total SO_2 emissions. Emissions from industrial processes (primarily from the pulp industry), is another major source of sulphur dioxide emissions. They have, however, been significantly reduced over the past decades.

Sweden long used a command and control approach to regulate sulphur emissions. The *Environmental Protection Act* (1969) set up guidelines to allocate individual permits to major stationary sources. For installations that entail a major environmental impact, the proponent must apply for a permit to the National Licensing Board for Environmental Protection, a central and independent authority. Limits for the sulphur content of various fuels have also been set up.

Environmental quality standards for SO_2 are based on EU Directives 1996/62/EC, 1999/30/EC, 2000/69/EC and 2002/03/EC, see Table 5.1.

Table 5.1. **Environmental quality standards for SO_2 in Sweden**

Substance	Concentration not to be exceeded (year of compliance)
Sulphur dioxide (SO_2)	
Hour[1]	200 μg per m^3 (2001)
24 h[2]	100 μg per m^3 (2001)
Year (eco systems)	20 μg per m^3 (2001)
Winter half-year (eco systems)	20 μg per m^3 (2001)

1. To be exceeded *not* more than 175 times per year (98 percentile, hour).
2. To be exceeded *not* more than 7 times per year (98 percentile, 24-h).

Source: Swedish EPA.

However, in 1991, a sulphur tax was introduced. It did not *replace* administrative regulation but was introduced as an *additional* measure, to comply with the national targets set up in a faster and more cost-effective way. When the tax was introduced, the maximum sulphur content of coal and fuel oils was already low by European standards. The sulphur tax was designed to achieve further reductions from these already low levels.

The tax is applied on the sulphur content of heavy fuel oils, coal and peat. The tax rate corresponds to SEK 30 per kilogram of sulphur content (approx EUR 3.32 per tonne).[14] The rate of the tax is relatively high compared to taxes levied on SO_2 in other OECD Countries.[15]

The implementation of the sulphur tax has also been related to the process of tax reform that has been taking place in Sweden since the beginning of the 1990s. In connection with this process, Sweden began a process of "green tax shift". Under this "tax shift", energy taxes were increased, while other taxes (such as income and payroll taxes) were decreased by similar amounts. Thus, the tax shift was intended to be revenue-neutral, while helping the country reach environmental goals.

The calculation of the rate took into account the estimated sulphur premium in the oil market (around SEK 10-55 per cubic meter for each 0.1 per cent by weight of the sulphur content) and the estimated costs of fuel desulphurisation (approximately between SEK 10 and SEK 40 of separated sulphur). The tax differentiation for light fuels-oils was set to approximate the extra cost for the cleaner fuels.

The annual estimated revenues were assumed to be around SEK 500 million for the sulphur tax levied of oil, and SEK 400 million for the tax on coal and peat. The revenues obtained with the sulphur tax are collected as general revenues in the national budget, not earmarked for environmental purposes.

The sulphur tax can be refunded if sulphur is removed from the exhaust gases and if emissions of SO_2 have been reduced through the use of desulphurisation techniques or other techniques. The National Tax Board refund SEK 30 per kg of sulphur in proportion to the reductions achieved. Tax reimbursements are mostly based on continuous monitoring systems.

In addition to applying a mix of direct regulation and economic instruments to reduce SO_2 emissions, the Swedish government also use financial support programmes to stimulate innovation in a similar way as to address NO_x emissions.

5.5. Instruments addressing regional air pollution in Canada

5.5.1. *The environmental issue*

The issue of acid rain started to raise some concerns in Canada in the late seventies, early eighties. As a result of the introduction of policy measures targeted at SO_2 and NO_x, the emissions have decreased, in particular for SO_2. Canada's total SO_2 emissions have decreased about 50% since 1980, to 2.4 million tonnes in 2002.

While SO_2 and NO_x emissions are projected to decrease significantly from 1995 levels, four Provinces in eastern Canada continue to have acidification problems and are committed to additional reductions: the provinces of Ontario, Quebec, New Brunswick, and Nova Scotia. In addition, transboundary flows from the United States continue to be an issue.

5.5.2. *The institutional context*

According to the Canadian Constitution, the responsibility for environmental protection lies with the provinces/territories. For instance, the provinces will have the responsibility of SO_x and NO_x emissions at the facility level. For power plants for instance it would be the Province that would set the standard.

However, while provinces have to deal with pollution within their borders, the Federal Government has the duty of addressing transboundary pollution. In the area of acidification, the Federal government addressed transboundary flows from the United States, where polluting sources are mainly power plants.

The Federal government also has jurisdiction over products (*e.g.* vehicles, fuels) and, as such, sets standards for new vehicles and fuel quality. For instance, Federal standards for sulphured gasoline or sulphured fuels will be introduced. Besides, with the revision of environmental acts some additional pollutants are now considered as toxic substances, which give new responsibilities to the Federal government to regulate at the national level.

5.5.3. *Instruments applied at the Federal level*

5.5.3.1. *Model Guidelines for NO_x and SO_2 emissions standards*

In the area of acidification, like in some other environmental areas where the responsibility lies with the Provinces/Territories, standards are set up through a negotiated process. They are not used in a prescriptive manner, but serve instead as a guideline. National guidelines exist for a variety of activities, such as power plants or industrial boilers, which are developed by the Federal government with the provinces and are published under the auspices of the Canadian Council of Ministers of the Environment (CCME). The standards set at the Federal level do not prescribe a certain technology, but reflect what is achievable with BAT.[16]

However, the role of agreement has evolved over time as the standards are now considered as an agreement to which the Provinces/Territories commit themselves. Now, the Canadian-wide standards are formal agreements signed between the Federal government and all of the provincial governments. They engage the different parties to respect the agreement and meet the standards within a certain period of time and put in place appropriate measures. More than guidance documents, these agreements tend to be considered as a form of public contract between the Federal government and the provinces, but they are not strictly legally binding. The model is that generally, the Federal government does not get involved in how the Provinces will meet a given standard.

Provinces use these model standards to develop their own regulations. For major industries (*e.g.* pulp and paper mills, smelters, refineries) provinces may have special regulations in place. Sometimes, it may be an operating permit at the level of the facility. It depends on how provinces regulate their industrial facilities. In most provinces, new facilities are not being built frequently enough to have a regulation and a permit basis approach may be used. For instance, in Nova Scotia they may build a power plant every ten years while a province like Ontario will have enough new facilities to develop standards. The guidelines will be used as a benchmark for issuing permits to new facilities (*e.g.* power plant, industrial facility).

A large number of air pollution standards were set up in Canada in the nineties, such as the standards dealing with VOCs and NO_x performance standards for polluting sources (*e.g.* power plants, boilers, gas turbines, stationary engines, refineries). Standards which

were negotiated about a decade ago are now updated. Some have already been revised while others are in the process of being updated like NO_x standards.

The Canadian model differs significantly from the approach developed in the United States where the standards are set at the Federal level for major emission sources. The implementation and enforcement is delegated to the states but the Federal government has the responsibility and the authority over major sources.

5.5.3.2. *Federal financial support to R&D*

At the Federal level, support programmes for the development of clean technologies have been developed similar to the US Clean Coal Technology Project to reduce SO_2 emissions. Funding for energy R&D programmes is administered primarily at Natural Resources Canada by the Office of Energy Research and Development (OERD) under the federal *Programme for Energy Research and Development* (PERD).

The *Clean and Efficient Combustion Technologies for Large Utility Electricity Generation* program is designed to help reduce emissions from combustion-based electric power generation systems by converting fossil fuels to electricity more efficiently. Activities focus mainly on: enhanced performance and efficiency, fuel switching and co-firing, and the accelerated demonstration of advanced fossil fuel-to-electricity cycles associated with "clean coal"; reduction of emissions of nitrogen oxides, sulphur oxides and particulate matter, through better power plant operation, design and control; and control of toxic emissions through a better understanding of fuel and combustion residue properties, and the development of capture processes.

5.5.3.3. *Fuel standards for stationary sources*

In 2001, the Canadian Government proposed measures to reduce the level of sulphur in both light and heavy fuel oils used in stationary sources (*Notice of Intent on Cleaner Vehicles, Engines and Fuels*) with a view to match the requirements set by the European Union (*i.e.*, 1% wt. for heavy fuel oil and 0.1% wt. for light fuel oil to be fully implemented by 2008). To this end, Environment Canada released a Discussion Paper in April 2003 and is currently assessing the path forward. The standards for fuel oils in stationary sources would come as a complement to the new regulations to reduce sulphur in gasoline and on-road diesel fuel.[17]

5.5.4. *Instruments applied at the Provincial/Territorial level*

In addition to the policy measures developed by the Federal government to address acidification concerns in Canada, instruments are specifically elaborated and implemented at the provincial/territorial level. Such measures include emissions permits at the facility level and emissions trading programmes on NO_x and SO_2.

Under the previous Federal-Provincial Agreements, the provinces agreed to meet specific emissions reductions.[18] As part of the Canada-wide *Acid Rain Strategy for Post-2000*, the provinces of Ontario, Quebec, New Brunswick, and Nova Scotia all committed to an additional 50% reduction in their SO_2 emissions beyond their 1985 Eastern Canada Acid rain Program targets by 2010.

Most provinces regulate their major sources with site-specific emission permits. Ontario for instance has its own specific regulations concerning NO_x and SO_x emissions for each of the major sources in the Province (*e.g.* smelters, Ontario Hydro). Each facility is

required to have a permit for all the environmental aspects of its operation, called certificates of approval. This site-specific approach is aimed at very local pollution effects.

In addition, some provinces are implementing emissions trading systems for NO_x and SO_2. The Province of Ontario has already introduced such a programme and Nova Scotia is considering introducing a tradable permit system to address acidification in the near future.

5.5.4.1. *Measures adopted in the Province of Ontario*

The Ontario Ministry of the Environment introduced net emissions caps on NO_x and SO_2 in 2002. The electricity sector is the first to have net NO_x and SO_2 emissions caps.[19]

As a follow-up step, Ontario envisions to expand the sector coverage of the emissions trading system to seven manufacturing industries in the province, including the iron and steel manufacturing, pulp and paper, nickel smelting and petroleum refineries. Capping other sectors in Ontario in a way that is consistent with the Ozone Annex would improve the market liquidity.

In addition to trading at the state level, opportunities for trading across national borders are being considered. Allowance trading between Ontario and the US is one of the possibilities in developing market liquidity. The development of an emission trading system that will mesh with the existing US trading system to provide the potential for Ontario electricity generators to participate in transboundary trading is a recommendation formulated by Environment Canada.

5.5.4.2. *Measures adopted in the Province of Alberta*

To manage NO_x and SO_2 emissions, the Government of Alberta is also applying a mix of instruments. Command-and-control are used in combination with economic instruments to target emissions from the electricity-generating sector.[20] New standards based on the Best Available Technology Economically Achievable (BATEA) have been implemented by Alberta Environment for coal and gas units.[21] The new annual NO_x and SO_2 emission limits for electricity generation units are effective since 2006 and apply to new generating units as well as generating units that have reached the end of their design life. The new BATEA standards are implemented together with an emission trading program for NO_x and SO_2 in the electricity sector (Alberta Environment, 2005).[22] The emission trading regulation was adopted in February 2006.

The emissions trading program has been created to provide industry with flexibility to meet new standards for nitrous oxides and sulphur dioxide, to encourage early emissions reductions and early shutdown of older units. If a unit emits less than its baseline emission rate,[23] it will generate credits. Unit operators can also receive credits for shutting down their unit or meeting the annual emission limit earlier than required.

The emissions trading program allows a unit operator to use credits to meet its new annual emission limits when a unit reaches the end of its Design life. A unit's operator can use credits for a period of no more than 10 years, thereby enabling the unit to operate for a maximum of 40 years from commissioning for gas, and 50 years from commissioning for coal. To ensure that regional air quality guidelines are met and prevent any "hot spots", unit operators cannot use credits to meet their hourly emission limits.

Notes

1. For further details on the LRTAP Convention, see: *www.unece.org/env/lrtap/welcome.html*
2. See Baumol and Oates (1988) for a discussion of the combination of taxes and bans to address temporally differentiated impacts.
3. In Illinois, the use of scrubbers was mandated. In New York, the Department of Environmental Conservation filed a suit to force the EPA to use "deposition standards" to restrict the use of permits in environmentally sensitive areas [see OECD (2003b)].
4. Previous relevant Protocols include the 1985 *Protocol on the Reduction of Sulphur Emissions or their Transboundary Fluxes by at least 30%*, the 1988 *Protocol Concerning the Control of Nitrogen Oxides* or *their Transboundary Fluxes* [see *www.unece.org/env/lrtap/status/lrtap_s.htm*].
5. They are the maximum amount of acidifying deposition an ecosystem can tolerate in the long term, without being damaged.
6. As well as particulates (dust).
7. Excluding cement kilns and coke ovens.
8. Preparatory work has started to revise the National Emission Ceilings Directive. For further details, see *http://ec.europa.eu/environment/air/rev_nec_dir.htm*.
9. For further details, see *www.epa.gov/airmarkets/usca/airus04.pdf*.
10. See Environment Canada (2006) for further details and for a progress report.
11. More information on acidification in Sweden is available at the website of the Swedish Environmental Protection Agency, at *www.internat.naturvardsverket.se/*.
12. For further details, see *www.internat.naturvardsverket.se/* and OECD (2006b).
13. That figure does not include emissions from fuel sold in Sweden for use in international shipping and aviation.
14. For oil, the tax rate amounts to SEK 27 (approximately EUR 2.9) per cubic meter for every 0.1% by weight of the sulphur content of such oil. For liquid fuels containing less than 0.1% sulphur by weight, no sulphur tax is charged for practical reasons.
15. Details on sulphur taxes in OECD member countries can be found at *www.oecd.org/env/policies/database*. A comparison of sulphur tax rates can be found in OECD (2001).
16. For example, the standards would typically reflect a very low-temperature combustion system for NO_x emissions and scrubbing for SO_2.
17. The Federal regulations which were in effect from 1998 to 2002, required all on-road diesel fuel to have a sulphur level not exceeding 0.05% by weight (500 mg/kg). Those regulations were revoked and replaced in 2003 by the *Sulphur in Diesel Fuel Regulations*.
18. Targets for SO_2 emissions were approximately a 50% reduction from 1980 levels by 1994 for large plants.
19. The NO_x cap was 36 kilotonnes (kt) of NO (55.1 kt of NO_2) in 2001 for power plants fuelled by coal and oil, falling to 28 kt of NO (42.8 kt NO_2) in 2007 for all electricity generating stations. The SO_2 cap was 157.5 kt in 2001, falling to 131 kt in 2007.
20. Alberta Environment is implementing the recommendations of the Clean Air Strategic Alliance's Electricity Project Team for the management of air emissions from the electricity-generating sector [see Clean Air Strategic Alliance (2003)].
21. See *www3.gov.ab.ca/env/air/Documents/2006_Power_Plant_Emission_Standards.pdf*.
22. See *www3.gov.ab.ca/env/air/pubs/AtoZTrading.pdf*.
23. The baseline emission rate is an emission rate below a unit's currently allowed rate of emissions.

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Chapter 6

Instrument Mixes Addressing Emissions to Air of Mercury

This chapter reviews instrument mixes dealing with emissions to air of mercury in Norway, Sweden and in the United States. In the case of the United States, the focus is on instruments applied at the Federal level, and on instruments in use in the States of Massachusetts and Michigan.

6.1. Introduction[1]

As a chemical, mercury has both typical and atypical implications for environmental policy instruments:

- Mercury has aspects similar to other hazardous chemicals which should be addressed with regard to workplace environment, consumer safety and other direct – and often complex – exposure risks.
- For a chemical, mercury has atypical aspects similar to bulk atmospheric pollutants such as SO_x, NO_x, etc., because mercury cycles long distances in the atmosphere and has certain major, well defined, point-source categories.
- Like many other chemicals, mercury also has many smaller, diffuse, source categories, which are much more complex to address than the major, well-defined, sources.
- As a heavy metal, mercury's toxicity is persistent. It cannot be destroyed and it remains harmful even if its chemical form changes; as such it flows through societal sectors and environmental compartments with associated risks over a long time-span.

6.1.1. Basics of mercury releases and cycling

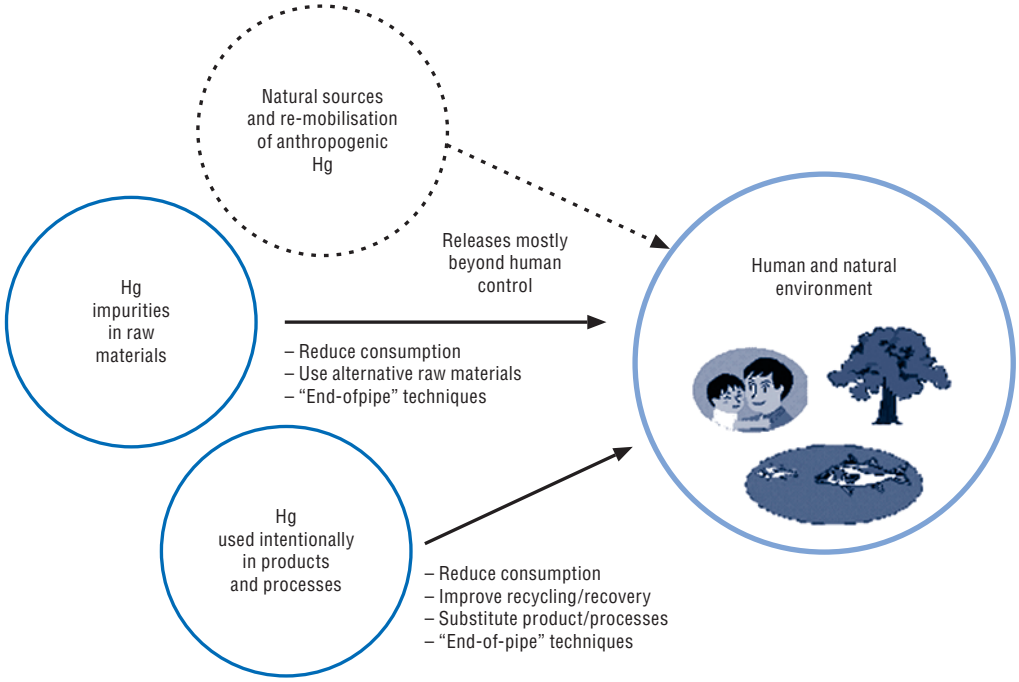
The recipients of mercury releases to the environment include the atmosphere, water environments (aquatic) and soil environments (terrestrial). There are continuing interactions – fluxes of mercury – between these media. Current releases add to the global pool of mercury in the biosphere – mercury that is continuously mobilised, deposited on land and water surfaces, and re-mobilised. Being an element, mercury is persistent – it cannot be broken down to less toxic substances in the environment. However, mercury does exhibit complex cycling in the environment where it undergoes various reactions and can change forms. Nevertheless, the only known long-term "sinks" for removal of mercury from the biosphere are deep-sea sediments and, to a certain extent, controlled landfills. This also implies that even as the anthropogenic releases of mercury are gradually eliminated, decreases in some mercury concentrations – and related environmental improvements – will occur only slowly, most likely over several decades or longer. However, improvements may occur more quickly in specific locations or regions that are primarily impacted by local or regional sources [see UNEP (2002)].

Releases of mercury to the biosphere can be grouped in four categories [UNEP (2002)]:

- *Natural sources* – releases due to natural mobilisation of naturally occurring mercury from the Earth's crust, such as volcanic activity and weathering of rocks.
- Current anthropogenic (associated with human activity) releases from the *mobilisation of mercury impurities in raw materials* such as fossil fuels – particularly coal, and to a lesser extent gas and oil – and other extracted, treated and recycled minerals.
- Current anthropogenic releases resulting from *mercury used intentionally* in products and processes, due to releases during manufacturing, leaks, disposal or incineration of spent products or other re-leases.
- *Re-mobilisation* of historic anthropogenic mercury releases previously deposited in soils, sediments, water bodies, landfills and waste/tailings piles.

Figure 6.1 illustrates these release categories and main types of possible control mechanisms.

Figure 6.1. **Sources of mercury releases to the environment and main control options**



Source: UNEP (2002).

6.1.2. Mercury releases and their exchange between media

This chapter deals with "instruments used to address mercury emissions to air", which includes all legislation, regulation and other measures intended to control or reduce anthropogenic, atmospheric mercury emissions. All types of measures addressing *any* phase in the life-cycle of mercury which may ultimately lead to atmospheric mercury emissions are included here. Examples include bans on mercury use in specific consumer products (*e.g.* batteries) which end up in incinerated household waste, as well as specific emission limits for mercury in emitted flue-gas in waste incinerators. A special feature of mercury is its *liquid* form and its *volatility* at ambient temperatures. Due to volatility, most forms of mercury released or discarded as waste – as vapour, as liquid elemental mercury, as compounds, as integrated in products, consumer waste or flue-gas cleaning residues – may lead to emissions to air. Therefore, the instruments addressing "ultimate" mercury emissions to air may need to be quite numerous and diverse. Instruments addressing mercury exposure in the workplace, and instruments that aim at reducing exposure from mercury already in the environment (such as fish consumption advisories) are *not* included systematically in this study.

In addressing pollutants such as mercury, a distinction is often made between *reduction measures* and *prevention measures*. Reduction measures aim at reducing releases to the environment through the use of emission control systems (for example filters retaining the pollutant from exhaust gasses). Prevention measures aim at cutting the input or

generation of the polluting substances at the original source by using alternative materials or technologies (for example by banning the sale of products containing mercury or by substituting a mercury-containing fuel in energy production).

An overview of the types of instruments used to address mercury in its different life-cycle phases is given in Table 6.1, along with indications of the current state of implementation.

Table 6.1. **General overview of the types of instruments used to address mercury**

Type and aim of measure		State of implementation
Production and use phases of life-cycle		
POINT SOURCES	Prevent or limit the intentional use of mercury in processes	General bans implemented in very few countries
	Prevent or limit mercury from industrial processes (such as chlor-alkali and metallurgic industry) from being released directly to the environment	Implemented in many countries, especially OECD countries
	Apply emission control technologies to limit emissions of mercury from combustion of fossil fuels and processing of mineral materials	Implemented in some OECD countries
	Prevent or limit the release of mercury from processes to the wastewater treatment system	Implemented in some OECD countries
	Prevent or limit use of obsolete technology and/or require use of best available technology to reduce or prevent mercury releases	Implemented in some countries, especially OECD countries
PRODUCTS	Prevent or limit products containing mercury from being marketed nationally	General bans implemented in a few countries only. Bans or limits on specific products are more widespread, such as batteries, lighting, clinical thermometers
	Prevent products containing mercury from being exported	Only implemented in a few countries
	Prevent or limit the use of already purchased mercury and mercury-containing products	Only implemented in a few countries
	Limit the allowable content of mercury present as impurities in high-volume materials	Only implemented in a few countries
	Limit the allowed contents of mercury in commercial foodstuffs, particularly fish, and provide guidance (based on same or other limits values) regarding consumption of contaminated fish	Implemented in some countries, especially OECD countries. WHO guidelines used by some countries.
Disposal phase of life-cycle		
Prevent mercury in products and process waste from being released directly to the environment, by efficient waste collection		Implemented in many countries, especially OECD countries
Prevent mercury in products and process waste from being mixed with less hazardous waste in the general waste stream, by separate collection and treatment		Implemented in many countries, especially OECD countries
Prevent or limit mercury releases to the environment from treatment of household waste, hazardous waste and medical waste by emission control technologies		Implemented or implementation ongoing in some countries, especially OECD countries
Set limit values for allowable mercury contents in sewage sludge spread on agricultural land		Implemented in a number of countries
Restrict the use of solid incineration residues in road-building, construction and other applications		Implemented in some OECD countries
Prevent the re-marketing of used, recycled mercury		Only implemented in a few countries

Source: UNEP (2002).

6.1.3. Historical administration of mercury sources

Instruments used nationally today to address mercury have often been developed in the light of an empirically based evolution of observations, motivations and policy reactions:

- Specific environment or health impacts are first observed at specific local point sources, *e.g.* paper mills using mercury biocides, as was the case in Sweden (Swedish EPA, 2006). Instruments are then introduced to address the specific point source problems.
- Increased awareness of the pollutant triggers surveys of the presence and impacts of the pollutant in other natural recipients, foods, ground water, etc. Major point source categories releasing mercury to the local environment are identified and their releases quantified through on-site measurements of mercury releases. Instruments are then introduced to address these point source categories (regionally or nationally).
- The knowledge and understanding with regard to impacts, exposures, mercury-releasing source categories and released quantities increase, and current risks of exposure above estimated threshold levels are still observed or modelled. Instruments are then focused to address remaining source categories, such as diffuse sources (*e.g.* consumer products and mobile sources) and smaller point sources. An integrated approach is sometimes introduced to understand (map) and address release sources throughout the life-cycle of mercury in products and processes. This is done, for example, by introducing separate collection and treatment of mercury-containing waste, or by banning sales of mercury-containing products to minimise releases and exposure throughout the life-cycle, and reduce costs of measures aimed at minimizing downstream releases.

The first two bullets above focus primarily on industrial and service sectors, which generally comprise relatively few, easily identifiable, actors on a national scale. The integrated approach and the smaller sources mentioned in the third bullet, make it necessary to address a generally much larger, and more complex, group of actors on a national scale (*e.g.* a large number of import/export trade companies, and retailer shops with their own imports, who may in many cases be unaware that their activities cause mercury releases and exposures).

As a consequence of the historical sequence of events described above, because of the practical aspects of environmental management and enforcement work, and perhaps because of differences among the sectors involved, the various instruments addressing mercury releases to the environment (and the enforcement procedures of the environmental authorities) tend to be divided according to the major groups of "point sources", "diffuse/other sources" and "waste management".

Table 6.2 gives examples of mercury release source categories grouped according to the original source of the mercury (intentional use *versus* mobilization of mercury impurities), and by group of instruments/enforcement often applied by environmental authorities. As can be seen, some source types naturally belong to more than one of the groups. For convenience, they are described under one heading only in this report. Note also that direct emissions from some of the source categories are primarily released to environmental media (*e.g.* water, land) other than the atmosphere.

Table 6.2. **Examples of mercury release source categories**

Grouped according to the original source of the mercury, and by group of instruments/enforcement often applied by environmental authorities

Origin of mercury released	Point sources	Diffuse/other sources	Waste management and similar end-of-use aspects
Intentional mercury use	Chlor-alkali production with Hg cells Mercury mining[1] Small-scale gold and silver mining (amalgamation process) Manufacture of Hg containing-products	Mercury-containing products: Dental amalgam, thermometers, switches, batteries, manometers, etc. Dental clinics	Hazardous/medical waste incineration Waste water facilities and sludge incineration Special Hg waste treatment: Recycling, deposition, storage Cremation Cemeteries
Mobilisation of mercury impurities	Coal combustion Other fossil carbon fuels combustion Non-ferrous metal extraction (zinc, gold, etc.)	Combustion of oil and gas Combustion of wood and other bio-fuels	
A combination of the above	Waste incineration Cement production (mercury in lime and waste used as fuel) Landfills		Waste incineration Landfills

1. Only a handful of dedicated mercury mines remain in operation in the World, but mercury is produced as a by-product in certain zinc, gold, lead and copper mines.

Source: UNEP (2002).

6.2. Instruments addressing mercury emissions to air in Norway[2]

6.2.1. Introduction

Norway is among the countries with an ambitious mercury regulation, and mercury release-reductions have been a national priority for the last two decades. The country has set ambitious targets for mercury releases, which are to be substantially reduced from the 1995 level by 2010, and the use and releases of mercury are to be eliminated within one generation (*i.e.* by 2020).

By 2003, Norway's releases had been reduced by 60% since 1995. Further action will be required to ensure that national targets are achieved. Mercury pollution is caused both by national emissions and by long-range transport of pollution. Thus, to reduce the total pollution in Norway, both international efforts and steps to reduce Norwegian releases are necessary.

The Norwegian Ministry of the Environment has developed an Action Plan to intensify actions to deal with mercury pollution. The *Action Plan* identifies important steps to eliminate mercury releases as far as possible, both in Norway and internationally [see Ministry of Environment (2004)].

Although Norway is not a member of the EU, it must meet EU environmental regulation as a consequence of the *European Economic Area* agreement with the EU. Much of the Norwegian environmental regulation therefore implements EU legislation, but in several cases Norway applies stricter regulation than the EU does.

6.2.2. Overview of instruments addressing mercury

6.2.2.1. Point sources

Emission limits in discharge permits issued pursuant to Norway's *Pollution Control Act* are the most important tool for reducing mercury releases from the manufacturing sector. This legislation prescribes maximum allowable releases of mercury (and other pollutants) from industrial facilities to the environment (air, water and soil/groundwater). The releases

from such point sources are regulated individually on a basis of national standards or guidelines. In addition, several industrial plants are covered by BAT requirements of the EU IPPC directive (CEU, 1996).

Mercury is a by-product in the *production of zinc*. There is one zinc production site in Norway. The mercury residue from this site is considered as hazardous waste, and is cemented in sarcophagi and placed in bedrock at the production site. There are no emissions of mercury from this activity.

Mercury emissions also come from the production of *ferromanganese and other manganese alloys*. There are four such production sites in Norway. All manganese ores have traces of mercury, but one of the commercial types of ore is substantially more contaminated than the others. The members of this Norwegian industry must apply for discharge permits for mercury. Conditions set in discharge permits for ferro-manganese producers require them to install equipment to control mercury emissions if the ore they use has too high a content of mercury.

Strict mercury emissions limits have been set for secondary steel manufacturing. However, in order to ensure equal conditions for competition with EU actors in the same industry, these requirements will only enter into force when EU countries adopt the regulation for the sector (SFT, 2006).

Norwegian *chlor-alkali* plants have changed from mercury-cell technology to diaphragm and membrane technology, thereby eliminating mercury releases from this sector. This is a general trend in OECD countries, but Norway is ahead of many countries in this aspect.

Crematoria have purification requirements on the emissions of mercury since 1 January 2003. For units existing before this date, the regulation entered into force from 1 January 2007.

6.2.2.2. Products and other diffuse sources

Legislation is in place preventing certain products containing mercury from being marketed nationally:

- *Thermometers:* The trade of mercury-containing thermometers for both consumer and professional use was banned in 1998.
- Regulations on scrapped *electrical and electronic products* came into force in 1998. Similar regulations are now included in the EU *Directives* on *Restriction of the use of Certain Hazardous Substances in Electrical and Electronic Equipment* [RoHS, *cf.* EU (2002c)] and on *Waste Electrical and Electronic Equipment* [WEEE, *cf.* EU (2002b)]. The RoHS Directive sets limits to allowable mercury contents in certain electrical and electronic devises, and the WEEE Directive prescribes how the wastes of such products should be handled, due to environmental considerations.
- Regulation of *batteries* came into force in 1990. Similar regulations are now included in the EU *Battery Directive* (1991) with later revisions, setting limits for allowable mercury contents in batteries and requiring the labelling and separate collection of the same.
- There is no ban of mercury in *paints*. However, no amounts are registered for use in the Norwegian Product Register. There is a ban on use of mercury compounds in antifouling preparations and wood preservatives (since 1994).

- *Pesticides* must be authorized for use in Norway. Although there is no general ban on mercury, mercury-containing products for pesticide use have not been authorised since 1992.
- *Dental amalgam:* Dental clinics must (since 1998) collect remaining amalgam and amalgam as waste by a filter device before the wastewater enters into municipal sewage systems. The waste must be delivered to reception-stations for hazardous waste. Filter devices must since 2006 retain 95% of the amalgam. National guidelines have been developed for the use and disposal of dental amalgam.[3]
- Due to the use of dental amalgam, Norway has regulated mercury emissions from new crematoria constructed since 2003. For units existing before this date, the regulation entered into force from 2007. The regulation is aiming at reducing mercury emissions from crematoria through emission limits (above a certain activity rate).
- Legislation also sets limits on allowed contents of mercury in *packaging materials* as a result of the EU *Packaging Directive* (see Section 2.3.3), which is aimed at enhancing packaging materials recycling/energy recovery and limiting the flow of heavy metals to waste treatment and resulting environmental releases among other aims [see ACAP (2004)].
- *General instruments for products and materials* containing chemicals (including mercury) in Norway include a requirement to substitute with less hazardous substances whenever possible, general labelling requirements (for contents of hazardous substances) and various information activities targeting sectors or the broad public (SFT, 2006).
- Norway has proposed a national general prohibition on the production, import, export, marketing and use of mercury and mercury-containing products. The proposal has been notified to the European Commission and the WTO. The Norwegian Ministry of Environment expects such a ban to enter into force in 2007.

6.2.2.3. Waste management

Norwegian legislation prescribes separate collection and environmentally sound waste treatment of products and process-waste containing mercury – for instance for batteries, electric articles, fluorescent light tubes and dental amalgam filter residues.

Collection of scrapped electrical and electronic products has been covered by national regulation since 1999. This was later implemented with the EU RoHS and WEEE Directives. Norwegian national regulation on waste comprises more products than the WEEE Directive (*e.g.* Norwegian regulation also includes industrial electrical and electronic equipment waste).

Treatment facilities for end-of-life vehicles are required to remove components that contain mercury from all vehicles. Until 2007, the collectors receive subsidies contributing to this activity from the Norwegian Pollution Authority (SFT). From 2007, this obligation is part of the obligations of the take-back system set up in accordance with the EU Directive on End-of-Life Vehicles, *cf.* EU (2000c). The components are delivered to a plant for hazardous waste deposition.

Producer take-back requirements. An overview of take-back requirements for products that may contain mercury is given in Table 6.3.

Sewage sludge. There is also legislation prescribing maximum allowable concentrations of mercury in wastewater sludge used as fertiliser on agricultural land (3 mg per kg total residue) and on other areas (5 mg per kg total residue). Using wastewater sludge on agricultural land with soil containing more than 1 mg per kg total residue is not allowed (ACAP, 2004).

Table 6.3. **Overview of take-back requirements for products which may contain mercury**

Product category	Share required to be taken back			Organisation of the take-back system			Financing of the take-back system					
	> 80%	50-80%	< 50%	Individual collection	Voluntary participation in a PRO	Obligatory participation in a PRO	No separate financing	Fee on all products within the product category	PRO fees based on the market share of each firm	PRO fees based on the recycling costs of each product	Public subsidies	Other
EE-waste (incl. fluorescent lamps)	X			X[2]	X[2]			X				
Batteries		X				X		X				
Vehicles	X					X[1]		X				

1. Partly from 1 July 2002. Completely from 1 January 2007 (End-of-life vehicles Directive).
2. Individual collection or participation in Producer Responsibility Organisation (PRO) for EE-waste.

Source: Norwegian questionnaire response.

Hazardous waste treatment including incineration. Waste containing more than 0.25% mercury is treated in accordance with the legislation for hazardous waste. Generally hazardous waste containing mercury is disposed of in an authorized landfill for *inorganic* hazardous waste.

The release of mercury from incineration plants for medical and household wastes is restricted by specified emission limits in the relevant permits. All municipal incineration plants with permits newer than 1994 have 0.03 mg mercury per Nm^3 flue gas as emission limit. This stringent limit has been in force from 2003 for new facilities incinerating medical and hazardous wastes, but is now in force for all existing incinerators. The new limits are more stringent than in EU. From 2003, ashes and slag will be on the European Union waste list and be considered hazardous waste if it contains hazardous substances. For mercury, the limit is as mentioned earlier 0.25% (see ACAP, 2004).

Incineration of municipal waste. Norwegian atmospheric emission limit values for mercury from municipal waste incineration (0.03 mg mercury per Nm^3 flue gas) are lower than required by the EU Waste Incineration Directive 2000/76/EC (0.05 mg mercury per Nm^3 flue gas).

Norway has also introduced a tax on atmospheric emissions from incineration of municipal waste in order to give operators of the plants economic incentives to reduce emissions of mercury and a number of other pollutants beyond the already strict limit values (Norwegian questionnaire response, 2006; SFT, 2006).

In 1999, Norway introduced general taxes on final treatment of municipal waste based on tonnes of waste treated by incineration, deposition or other methods. The tax combined a *basic tax* (NOK 75 per tonne) for all waste incineration, and an *additional tax*, depending on the level of energy generation from the incineration (NOK 0-225 per tonne of waste). This tax resulted in reduced waste generation, increased materials recycling, decreased waste deposition, and somewhat better energy recovery. This tax may also have had indirect effects on atmospheric mercury emissions.

From 1 July 2004, the tax on incineration of waste was changed into a tax on measured or estimated emissions of a number of substances, including mercury. The tax rate for atmospheric mercury emissions were set at NOK 27 per g mercury emitted, based on assessments of the costs of mercury's adverse environmental impacts [SFT (2006);

assessment by ECON (2000) and (2001)]. The tax rate is being price adjusted annually. In 2006, the national environmental authority (SFT) reviewed the tax on all substances and found no relevant, available material that suggested that a change in the level of the tax was needed. The tax rate on mercury was introduced to promote implementation of release-reduction equipment on the waste incineration plants earlier than required by the EU *Waste Incineration Directive*. The Directive had a relatively long transition period for existing incineration plants. In economic terms, the tax internalises the environmental costs of mercury-releases from the incineration plants.

According to sources interviewed during the case study, there have been some considerations concerning the exemptions to the tax. At present, the tax principally includes incineration of *municipal waste*. The disadvantages of including the incineration of for instance *hazardous and other waste* were considered too large. Due to concern for negative sectoral competitiveness impacts, it has not been considered to include mercury emissions to air from, for instance, industry in the tax. It was also stated that existing pollution permits for mercury already cover these sources well.

6.3. Instruments addressing mercury emissions to air in Sweden

6.3.1. Introduction

Sweden is among the countries with the most ambitious mercury regulation, and Sweden was one of the countries which initiated international co-operation on mercury release-reductions several decades ago. Mercury has been a target for political decisions in Sweden for a long time due to findings of high levels of mercury in the environment. Thousands of Swedish lakes contain fish with mercury-levels above the limit-value set by the World Health Organisation (0.5 mg mercury per kg fish). The overall aim of Swedish policy is to reduce the levels of mercury in the environment to normal background levels. In the 1960s, most of the work on this issue was related to reducing emissions from point-sources. This was not enough to achieve this target, and it was concluded that stopping the use of products and processes, and international measures, were needed, as most of the mercury deposited in Sweden is coming from other countries.

The Swedish Parliament therefore decided in the early 1990s to reduce the use of mercury in products, mainly by regulatory means, but also supported by voluntary measures, and with the *ultimate aim of a total phase-out*. The official policy on mercury has led to increased awareness (and acceptance) by Swedish industry and society of the need to avoid mercury. Between 1992 and 2003, approximately 95% of the sold amounts of mercury in products were phased out. A national goal adopted in 2000 stated that mercury should be phased out in products and industrial processes as far as possible by 2003 (changed in 2004 to "as soon as possible, but latest by 2007"). According to present goals, remaining mercury use in the chlor-alkali industry will be allowed until the end of 2009.

The Swedish strategy to achieve a "mercury-free environment" consists of measures to reduce the input of mercury into society by imposing a ban on it in different products and processes; to collect mercury from society's so-called "hidden mercury store"; and to implement a terminal disposal of mercury. Mercury should not be recycled, but should be disposed of terminally in a safe and environmentally sound way. Substantive actions are also taken to restrict emissions of mercury to air, land and water. Guiding principles for the policy on mercury, and for hazardous substances in general, are the notions of precaution and substitution, which are also legal requirements according to the Swedish Environmental Code.

In spite of substantial reductions in domestic mercury releases (and examples of reduced mercury concentrations in some Swedish lakes), an accumulation of mercury in top soil is still being observed. An important reason for this is that mercury released from sources abroad is still deposited on Swedish territory. This is the background for the continued efforts made by Sweden to reduce mercury releases internationally (Swedish EPA, 2006).

Preservation of nature has been a major motivation for controlling mercury pollution in Sweden, and it has had a major influence on how mercury has been prioritised and targeted with instruments over the past decades. Nature is an important aspect of peoples' life in Sweden. For example, many people appreciate being able to catch and eat their own fish, and this may have helped in creating political support for protection of the environment, including for abating mercury pollution.

The Parliament decides on the *Environmental Quality Objectives* in Sweden based on proposals from the Environmental Objectives Council. The overall political targets regarding environmental protection are reviewed and communicated in reports and pamphlets dedicated for this purpose, and in other media, at least every four years (once in every Government period) by the Environmental Objectives Council. Presently, the Council have defined 16 main quality objectives for environmental protection. Interim targets are set for each quality objective, and there are 9 interim targets concerning hazardous substances, for example mercury (Swedish EPA, 2006).[4]

Sweden is a member of the EU since 1995. However, Sweden has stricter legislation on mercury than the EU since the beginnings of the 1990s, when a national regulation entered into force which prohibits certain mercury-containing products. As of 1 July 1997, mercury and chemical compounds and preparations containing mercury may not be commercially exported from Sweden. The EU mercury strategy (EC, 2005) – discussing most of the mercury issues at stake here – propose certain instruments which are in line with Swedish priorities on mercury, or are otherwise likely to affect Swedish policies on mercury in the coming years.

6.3.2. Overview of instruments addressing mercury[5]

6.3.2.1. Point sources

Point sources are regulated by the Swedish *Environmental Code* [1998:808; replacing the earlier the Environmental Protection Act (1969:387)]. Specific point sources are regulated individually on the basis of national and European Union standards/guidelines. Discharge of pollutants from point sources to air, water and soil may in general not be undertaken without prior permission from a competent authority. In the permission procedure, firms are generally required to use "Best Available Techniques" (BAT). This Code implements the requirements of the EU IPPC Directive. Reference documents on Best Available Techniques have been developed in the EU context for a number of source categories relevant for mercury, (for example, chlor-alkali production (BAT is mercury-free techniques only) and non-ferrous metal extraction).

There is currently one *chlor-alkali plant* in Sweden that still uses the mercury cell process. The more environmentally friendly membrane process is also used at one site. In line with the OSPAR[6] Decision 90/3 on phase-out of mercury cells in chlor-alkali plants, the Swedish Government has in several Bills stated that the amalgam process should be eliminated by the end of 2009 [see ACAP (2004) and UNEP (2002)].

Due to the use of dental amalgam, *crematoria* are the largest sources of mercury releases in Sweden today (about 110 kg per year). All larger crematoria have mercury-specific emission filters – mainly carbon filters in a number of different configurations. A guidance document on pollution abatement on crematoria describes the different emission reduction options and presents experience with achieved reductions. The guidance document builds on an OSPAR background document on crematoria, in combination with Swedish experience.

6.3.2.2. Products and other diffuse sources

Legislation prohibiting certain products containing mercury from being professionally manufactured, sold, exported or imported is implemented through Government Ordinance 1998:944 and includes thermometers and other measuring instruments, and a general ban on exports of mercury from Sweden. There are also restrictions regarding batteries, packaging and end-of-life vehicles. Swedish product regulation for mercury has so far been stricter than common EU regulation for most mercury uses.

Besides limits/bans for *mercury content in batteries*, an import fee is charged on certain mercury-containing batteries. The fee provides funding for separate collection and end treatment of spent batteries containing mercury (Swedish EPA, 2006).

Mercury-containing *chemicals* for analysis and reagents are mainly used in the environmental control of mercury sulphate in COD (chemical oxygen consumption) analyses. Information activities have not been effective to phase-out this particular use [see ACAP (2004) and UNEP (2002)].

For *dental amalgam*, a voluntary instrument was introduced in 1995 not to use amalgam for fillings in children's teeth up to the age of 19. In 1999, public dental care compensation ceased to pay for amalgam fillings (but not for mercury-free alternative fillings). In 2003, less than 2% of the total number of fillings were made with amalgam.

Sweden has proposed a *general national ban* on the marketing, use exports and imports of mercury and mercury-containing products. The European Commission and the WTO has been notified of this proposal. This procedure is a formal requirement for members of the EU who wish to adopt national regulation stricter than the EU regulation of the same issue. Sweden is currently in the process to respond to the Commissions statements on the proposal (Swedish EPA, 2006).

Table 6.4 presents some of the major initiatives on mercury use that have been implemented in Sweden in order to achieve the 1990 overall goal of total phase-out of mercury.

6.3.2.3. Waste management

Municipalities have the responsibility for setting up collection schemes for hazardous household waste. Hazardous (mercury-containing) waste should not be mixed, but should be collected separately and treated according to its properties. There are therefore separate collection systems for batteries, fluorescent lamps, amalgam waste, waste electronic and electrical equipment and mercury devices in end-of-life vehicles [see Nordic Council of Ministers (2001)].

Table 6.4. **Major initiatives on mercury use in Sweden**

	Source	Major initiatives on mercury use in Sweden
1979	Dental releases	A voluntary agreement was made with dentists associations in Sweden 1979, which requires that all dental clinics are equipped with amalgam separators.
1979	Seed dressings	The use of mercury-containing seed dressings is banned (SFS 1979:349).
1985	Biocides	Import, sale, transfer and use of mercury and mercury compounds as biocides are not approved (SFS 1985:836).
1990	Goal setting – phase-out	Government Bill 1990:91/90 proposed a numerous set of legislative and voluntary actions, with the ultimate aim of a total phase-out of mercury use.
1991	Dental amalgam	The overall goal of a phase-out of mercury also included dental amalgam. This lead to a voluntary reduction of new amalgam fillings in children's teeth from 30 to 1.5 % between 1991 and 1995. The use in adult's teeth decreased from 32 to 15 %. Since then, total annual sale of mercury in amalgam has levelled off.
1992	Clinical thermometers	The import, professional manufacture and sale of clinical mercury thermometers were prohibited from 1 January 1992.
1992	Export of waste	Export of mercury waste and products containing mercury was banned.
1993	Thermometers, measuring instruments and electrical equipment	Professional manufacture, import and sale of thermometers, level switches, pressure switches, thermostats, relays, electrical contacts and other measuring instruments has been banned since 1993. (Ordinance 1991:1290) Some exemptions, mainly for spare parts, still exist. (Ordinance 1998:944) Time-schedule for phase-out is stipulated for each exemption. (Regulation 1998:8)
1993	Goal setting – timing of phase-out	Government Bill 1993/94:163 set a goal of phase-out of mercury and mercury-containing products by the year 2000. After that date, mercury should be offered for sale only in vital products and for uses to which no alternative techniques are known or fully developed.
1995	Dental amalgam	An agreement was reached between the state and the county councils that amalgam should be phased out from children's and young people's dentistry as a precautionary measure to minimise the exposure of these groups to metal mercury vapour.
1998	Batteries	The EU Battery Directive that also applies in Sweden was amended in 1998. Batteries with mercury content in excess of 0.0005 % by weight are defined as dangerous for the environment and may not be marketed as such or incorporated into appliances. Button cells with a mercury content of no more than 2 % by weight are exempted. The new rules mean that mercury oxide batteries may no longer be sold – such batteries accounted for 700 of the 800 kg of mercury in batteries in 1997. The new rules have led to a sharp reduction in sold quantities of mercury in batteries – in 1999, the amount of mercury in batteries sold was estimated to approximately 100 kg.
1998	Sewage sludge	In Ordinance 1998:944 the contents of heavy metals in sewage is regulated in cases where sewage sludge is sold or conveyed for agricultural purposes. Regulations for when, where and how much sludge may be used in agriculture are found in SNFS 1994:2 (changed SNFS 1998:4). At present the maximum content of mercury allowed in sludge is 2.5 mg per kg dry matter and the maximum application is 1.5 g per hectare and year.
1997	Export of mercury and mercury – containing products	As of 1 July 1997 mercury and chemical compounds and preparations containing mercury may not be commercially exported from Sweden. Also, mercury – containing measuring instruments and electrical components may not be exported (Ordinance 1998:944).
1999	Dental amalgam	Dental care compensation ceased to be paid for amalgam fillings in 1999, which has made the cost of a composite filling and amalgam about the same for the patient. In 2003, amalgam was used for 0.05% of the fillings in children and young people and 1.8% of the fillings in adults in Sweden [see KemI (2004)].
2000	New products containing mercury	Bill 2000/01:65, Chemical Strategy for a Non-Toxic Environment requires that new goods put on the market should be, as far as possible, free from mercury by 2003, at the latest.
	Production processes	Also, mercury should not be used in production processes, unless the producer can prove that neither human health nor the environment would be harmed.
2006	All product uses and production processes	Sweden has proposed a national general ban on the marketing and use of mercury and mercury-containing products, export and import. The proposal has been notified to the European Commission and to the WTO.

Source: Based on ACAP (2004) and UNEP (2002).

For waste incineration, legislation prescribes maximum allowable releases of a number of pollutants to the atmosphere and wastewater, as well as specification for the depositing of solid incineration residues. Ordinance (2002: 1060) and Swedish EPA Regulation (2002:28) implement the EU Waste Incineration Directive. There were other laws and regulations in place before as well.

Between 1995 and 1999, the Government allocated almost SEK 30 million in subsidies to an Action Programme for the collection of mercury. This Programme was implemented by the Swedish Environmental Protection Agency, in co-operation with Swedish County Administrative Boards and municipalities. The Programme included projects for the inventory and collection of clinical thermometers, mercury in technical goods and products, and metallic mercury on shelves and in storerooms. Schools, universities and colleges were purged of mercury and a project to identify hidden mercury in industry was also carried out. The world's first mercury-tracking dogs were used in these efforts.[7] In addition, previously unemployed electricians were hired as "mercury hunters", with the task of identifying mercury in goods and products. The project included many awareness-raising activities, which were considered essential to reach a good result. As of 2003, 10-11 tonnes of mercury were identified; 6-7 tonnes of which were collected. At that time, the Swedish EPA estimated that there were about 30 tonnes of mercury still left in technical products and goods that are either in use or in storage, and more in households and agriculture [see Swedish EPA (2006a)].

Producer take-back schemes. Batteries, electronic and electrical waste (WEEE) are subject to producer take-back obligations. For small batteries it is not a real *producer* responsibility. Municipalities have the responsibility to set up collection schemes and producers are obliged to pay the cost of collection and treatment and other cost arising connected to the handling of discarded small batteries.

6.3.3. Instruments addressing final storage of mercury waste

The Swedish position is that mercury is a substance that remains a threat to human health and the environment in perpetuity, and for this reason it *should not be recycled*. Instead, mercury-containing waste should be dealt with in a permanently safe and environmentally acceptable way. Therefore, the Swedish Government has decided that waste containing more than 0.1% mercury shall be finally stored in a deep bedrock repository, at the latest by 2015 [Swedish Ordinances (2001:1063) and (1998:899)].

The Swedish decision to ban mercury export was based on a conviction that countries should not export an environmental problem. Very extensive investigations have been performed to identify the technically, environmentally and economically most feasible final storage depot, and several options have been under consideration. A deep rock depot in granite rock has been deemed the safest, and the costs for this approach were found to be defendable due to the political priority given to this project. However, detailed economic analyses of the investment and operation costs *versus* the environment and health benefits have not been performed. The deep rock mercury depot will be run by a private company, under the responsibility, and with the funding, of the mercury waste owners (primarily a few large enterprises). It will be constructed and run according to requirements defined by the Swedish Environmental Court, and will be controlled by the environmental authorities (Swedish EPA, 2006).

The possibility of establishing a common Nordic strategy for mercury containing was discussed in a working group under the Nordic Council of Ministers [see Endre *et al.* (1999)]. Issues such as retirement of mercury *versus* re-marketing, and making common use of Nordic waste treatment facilities, were discussed. It was, however, concluded that common solutions were premature at that time, due to difference in short-term priorities.

Establishment of the Swedish deep rock depot could potentially be reviewed depending of the outcome of current (spring 2007) EU negotiations about establishing temporary or final storage for excess mercury in conjunction with an EU-wide export ban for metallic mercury [see European Commission (2006a) and (2006b)].

Such an export ban (by 2011), including storage requirements, was proposed by the European Commission in October 2006, and the proposal will now undergo negotiations in the Council and the European Parliament. An EU-wide mercury retirement and storage programme could perhaps result in common solutions between several countries.

6.4. Instruments addressing mercury emissions to air in the United States

6.4.1. Introduction

The United States has been actively addressing the risks posed by exposure to mercury for many years, both through implementation of regulatory activities and voluntary reduction programmes [see ACAP (2004)]. Already in 1991, the US EPA initiated the "*33/50 Program*", aimed at helping to reduce releases of mercury and 16 other toxic substances into the environment. The goal of the Programme was to encourage companies to voluntarily reduce their releases of some or all of these toxics by 33% by 1992, and 50% by 1995. As a result, between 1988 and 1991, environmental releases of mercury were reduced by 38% and transfers of mercury for off-site treatment or disposal were reduced by 30% (OECD, 1995). However, while the 33/50 Program may have served as an initiation of later activities, the quantitative results were found to be moderate when corrected for other factors contributing to the observed reductions, according to studies by Khanna and Damon (1999) and Vidovic and Khanna (2007).

Also, the *National Partnership for Environmental Priorities Program* (NPEP) encourages public and private organisations to form voluntary partnerships with EPA that reduce the use or release of any of 31 priority chemicals, including mercury. To date, 26 partners have committed reducing their use of 38 375 pounds of mercury (US EPA, 2006b).

Understanding the characteristics and magnitude of mercury releases is critical to the design of effective risk management strategies. The *Clean Air Act*, amended in 1990, requires US EPA to prepare an assessment of the magnitude of USA mercury emissions by source, the health and environmental effects of the emissions, and the cost and availability of control technologies. The resulting report, *Mercury Study Report to Congress* was published in December 1997 [see US EPA (1997)]. As the state-of-the-science for mercury is continuously and rapidly evolving, the report represents a "snapshot" of the mid-1990s understanding of mercury in the United States.

The US EPA Office of Research and Development (ORD) published in 2000 its *Mercury Research Strategy*, which was intended to guide the mercury research programme through 2005. That Strategy identified the key scientific questions of greatest importance to the Agency, and then described a research programme to answer those questions. The goal was to reduce scientific uncertainties limiting US EPA's ability to assess and manage mercury and methylmercury risks. An integral part of the strategy involved a study of atmospheric mercury transport, transformation and fates.

In July 2006, US EPA issued a report titled *EPA's Roadmap for Mercury*. This report describes the Agency's progress to date in addressing mercury issues domestically and internationally, and outlines EPA's major ongoing and planned actions to address risks associated with mercury [see US EPA (2006c)].

6.4.2. Overview of instruments addressing mercury at the Federal level

The US approach to designing effective risk management strategies for mercury comprises both specific regulatory limits on releases and voluntary efforts with industry to reduce mercury use, implemented by a number of agencies at both the Federal and State levels. The most important initiatives at the Federal level are summarized below, based primarily on a review prepared by the US EPA for an Arctic Council report (ACAP, 2004).

In the US, individual States can define their own measures to reduce environmental impacts from mercury. There is a distinction between Federal environmental statutes, which are laws passed by the US Congress, and agency regulations, which are written by EPA and other agencies to implement the statutes. Statutes define some aspects of environmental programmes in detail and leave other aspects up to the discretion of the EPA administrator. State environmental programmes must conform to the statutes and regulations, but are generally allowed to be more stringent than Federal standards and, in some cases, to diverge from the detail of the regulations if such programmes can demonstrate that they meet or exceed the expectations of the regulations. Summary information on instruments used to address mercury at the State level is given in Section 6.4.3.

6.4.2.1. Point sources

Air point sources. Mercury and mercury compounds are considered hazardous air pollutants (HAPs) under the *Clean Air Act*. US EPA established *National Emission Standards for Hazardous Air Pollutants* (NESHAPs) for mercury emissions, based on risk under the pre-1990 version of the Clean Air Act. Under the Clean Air Act Amendments of 1990, US EPA regulates Hazardous Air Pollutant Emissions by source categories using *Maximum Achievable Control Technology* (MACT) standards for each "major source" in any listed source category. The Maximum Achievable Control Technology level for new sources is the level of HAP emissions control currently achieved by the best-controlled similar source. The MACT level for existing sources is the average level of mercury emissions control achieved by the top 12% of the currently operating sources. See also the description of the cap-and-trade programme for coal-fired power plants below, a major mercury point source type.

Chlor-alkali industry. In August 2003, EPA promulgated a rule that limits mercury emissions from plants that produce chlorine using the mercury-cell method. The rule includes emissions limits based on MACT and on stringent management practices. EPA estimates that this regulation will reduce stack emissions by 1 500 pounds (74% from previous levels), in addition to unquantifiable reductions in fugitive emissions expected as a result of improved work practice standards. This standard does not allow any new chlor-alkali mercury cell facilities to be built.[8] The last USA mercury-cell based factory was built in 1970.

In addition, as a voluntary measure, the Chlorine Institute, on behalf of USA mercury cell chlor-alkali facilities, committed in 1997 to reduce mercury use 50% by 2005 and to report annually on progress. In July 2004, the Chlorine Institute reported that mercury consumption by US chlor-alkali factories had declined by 76% over an eight year period, or a 69% reduction after adjusting for shut-down facilities. This is a decline from 160 tons per year (during a baseline period of 1990-95) to 30 tons in 2001.[9]

Energy production. The largest anthropogenic source of atmospheric mercury emissions in the US is currently coal-fired power plants. After considerations of several options for addressing this sector, the US EPA issued the *Clean Air Mercury Rule* in 2005, which

addresses mercury reductions with a "cap-and-trade" system for atmospheric mercury releases from this sector. Facilities facing high emission reduction costs, due to their state of technology or other factors, can buy emission allowances from facilities that can reduce mercury releases at lower cost. The rule works with two phases. The total mercury cap in Phase 1 is 38 short tons per year (to be achieved from 2010), and in Phase 2 (by 2018), the cap is set at 15 short tons per year.

Information and reporting requirements. Under the US Toxics Release Inventory (TRI), starting with the 2000 reporting year, the reporting threshold for mercury and its compounds has been lowered to 10 pounds (about 4.5 kg) per year. The previous thresholds were 25 000 pounds (about 11 300 kg) for manufacture or processing and 10 000 pounds (about 4 500 kg) for otherwise use. Through this action, the US got a much more comprehensive picture of the amounts of mercury and its compounds that are released to air, water, land, transferred off-site for disposal, transferred off-site for recycling or recycled on-site within industrial facilities.

6.4.2.2. Products and other diffuse sources

Mercury in products. Mercury-containing products are regulated in several ways. At the Federal level, mercury product regulation has generally centred on health-based reasons to eliminate mercury from products, using the *Federal Insecticide, Fungicide, and Rodenticide Act* (FIFRA) and the *Federal Food, Drug, and Cosmetic Act* (FFDCA). Restrictions on mercury-containing products, once used sparingly by the Federal Government, are increasing rapidly at the State level (see Section 6.4.3).

Batteries. Between late 1989 and early 1991, all US manufacturers converted production so that mercury content, except in button and "coin" cells, did not exceed 0.025% mercury by weight. A Federal law (the *Mercury-Containing and Rechargeable Battery Management Act*) went into effect in 1996. This Act prohibits the sale of:

- alkaline-manganese batteries containing mercury (alkaline-manganese button cell batteries are limited to 25 mg mercury per button cell);
- zinc carbon batteries containing mercury;
- button cell mercuric-oxide batteries for use in the USA; and
- any mercuric-oxide battery unless the manufacturer identifies a collection site that has all required Federal, State, and local government approvals, to which persons may send batteries for recycling and disposal.

The Act contains labelling requirements and encourages voluntary industry programmes, by eliminating barriers to funding the collection and recycling or proper disposal of used rechargeable batteries. The Act also grants States the authority to add other batteries to the recycling programme. This (Federal) law followed the lead of several States that passed legislation in the early 1990s limiting the mercury content of batteries.

Auto switches. In July 2006, EPA proposed a *Significant New Use Rule* (SNUR) to require notification to EPA 90 days prior to US manufacture, import or processing of elemental mercury for use in certain convenience light switches, anti-lock brake system (ABS) switches and active ride control system switches in certain motor vehicles. In 2003, American automakers voluntarily discontinued the use of mercury switches in convenience lights, ABS systems, and ride control systems. Foreign automakers had

previously discontinued use of such switches. The proposed rule does not include mercury switches used as aftermarket replacement parts for ABS and ride control systems in pre-2003 vehicles, because there are currently no suitable non-mercury substitutes for these replacement parts and the remaining market for these products is very limited and declining. The required notice would provide EPA with the opportunity to evaluate any resumption of use of mercury in these switches, and, if necessary, to prohibit or limit such activity before it occurs, in order to prevent unreasonable risk of injury to human health or the environment [see US EPA (2006d)].

Cosmetics. According to FFDCA, mercury use as a preservative or anti-microbial is limited to eye-area cosmetics or ointments in concentrations below 60 ppm. Yellow mercuric oxide is not recognized as a safe and effective ophthalmic anti-infective ingredient.

Dental amalgam. The Food and Drug Administration (FDA) also regulates dental amalgam under FFDCA. Dental mercury is classified as a Class I medical device, with extensive safety regulations on its use. Dental amalgam alloy is classified as a Class II device, subject to additional special controls.

Paints. As of 1991, all registrations for mercury biocides used in paints were voluntarily cancelled by the registrants, leading to a drastic decrease in the use of mercury in paint. In addition to the paint industry reformulating its paints to eliminate mercury, US EPA banned the use of mercury in interior paint in 1990 and in exterior paint in 1991.

Pesticides. The *Federal Insecticide, Fungicide and Rodenticide Act* covers the sale and use of pesticides, including registration of chemicals that meet health and safety tests. Several mercury compounds had previously been registered as pesticides, bactericides, and fungicides, but registrations of the last mercury-based pesticides for use to control pink and grey snow mold were voluntarily cancelled by the manufacturer in November 1993.

Vaccines. Under the *Food and Drug Administration Modernization Act* of 1997, FDA is required to assess the risk of all mercury-containing food and drugs. Under this provision, FDA asked vaccine manufacturers to provide information about thimerisol (a mercury-based biocide) content of vaccines. Based on this information, the Public Health Service, the American Academy of Pediatrics, and vaccine manufacturers agreed that thimerisol-containing vaccines should be removed as soon as possible. Manufacturers have been asked for a clear commitment to eliminate mercury from vaccines, and FDA will do expedited reviews of resulting revisions to product license applications.

Thermostats. As a voluntary measure, the industry-funded Thermostat Recycling Corporation (TRC) launched a programme in 1998 to recycle mercury-switch thermostats in nine States.[10] It has since been expanded to 48 States in the US, and in 2005 the programme collected 87 900 thermostats and 820 pounds of mercury. Between its inception in January 1998 and June 2006, it collected over 450 000 thermostats and removed more than 3 900 pounds of mercury from the waste stream. The mercury is purified for re-use.

US EPA is exploring other avenues, such as voluntary waste minimization partnerships focusing on mercury applications, to reach its environmental goals quickly and efficiently without imposing unfair burdens on particular industries, small businesses, or other groups.

Recreational mining. There is no active mercury mining in the United States. There is also no use of mercury in large-scale gold mining. There has been minor recovery of mercury by recreational miners in California, but the mercury is recovered as elemental free mercury in stream bottoms where mercury was used in the distant past for gold extraction. The mercury is incidentally recovered on the sluices of recreational portable dredge operators. US EPA and the State of California are working on ways to set up collection points for waste mercury to ensure that recreational miners do not dump their waste mercury in streams.

Transportation. The Federal Department of Transportation regulates hazardous materials transport under the *Hazardous Materials Transportation Act*. Mercury and mercury compounds are hazardous substances subject to packaging, shipping and transportation rules for hazardous materials.

6.4.2.3. Waste management

Waste treatment, including incineration. Prior to 1995, municipal waste combustors and medical waste incinerators were the largest identifiable sources of man-made mercury emissions to the atmosphere. Regulations that have been finalized for municipal waste combustors and medical waste incinerators will, when fully implemented, reduce emissions from these source categories by an additional 90% over 1995 levels.

In December 1995, the US EPA finalized New Source Performance Standards (NSPSs) and Emission Guidelines (EGs) applicable to *municipal waste combustor (MWC)* units with a capacity greater than 227 metric tonnes of waste per day (*i.e.* large MWCs). All 167 large MWCs that are subject to the regulations came into compliance by 2000 and mercury emissions (based on year 2000 stack test compliance data) from this source category have been reduced by about 95% from 1990 levels. A companion rule (NSPSs and EGs) for a small MWC unit (with a capacity of 32 to 227 metric tonnes per day) was adopted in 2000, with retrofit required by 2005. The same mercury emissions limits apply, and the same control technology was expected to be used.

Since 1997, mercury emissions from medical waste incinerators have been limited by a US EPA regulation that sets strict standards for new sources and requires existing sources to reduce emissions by 93-95%. The regulation also requires training and qualification of operators, incorporate siting requirements, specify testing and monitoring requirements to demonstrate compliance with the emission limits, and establish reporting and record keeping requirements.

As a voluntary measure, US EPA and the American Hospital Association in 1998 signed a Memorandum of Understanding (MoU) committing to work together to significantly cut hospital wastes by 2005. The MoU envisioned the virtual elimination of mercury-containing hospital wastes and a one-third reduction in total hospital wastes by 2005.

Hazardous waste incinerators. In 2002, US EPA promulgated interim emission standards for hazardous waste incinerators, hazardous waste burning cement kilns, and hazardous waste burning lightweight aggregate kilns under joint authority of the *Clean Air Act* and the *Resource Conservation and Recovery Act* (RCRA). The standards limit emissions of chlorinated dioxins and furans, other toxic organic compounds, toxic metals (including mercury), hydrochloric acid, chlorine gas and particulate matter.

Waste disposal. The RCRA hazardous waste regulations identify those solid wastes that should be managed as hazardous waste and specifies treatment, storage, and disposal requirements. They describe specific disposal requirements for specific wastes. Generally, mercury-bearing wastes are subject to land disposal restrictions. That is, the mercury concentration in these wastes must be below the regulatory concentration level before the wastes may be landfilled. For some types of waste, the regulations require a specific treatment, such as recovery of the mercury or incineration.

RCRA regulations also influence product disposal and recycling options for mercury-containing products. Discarded products considered to be hazardous wastes are potentially subject to transportation requirements, and permitting requirements for storage, treatment and disposal of the waste. Currently, certain spent mercury-containing equipment and other types of hazardous wastes are included in a *Universal Waste Rule* (UWR) that, since 1995, eases RCRA hazardous waste management requirements for those wastes and enables States to set up special collection programmes. "Universal wastes" are items commonly thrown into the trash by entities including households and small businesses. UWR is designed to reduce the amount of hazardous waste in the municipal solid waste stream, to encourage the recycling and proper disposal of some common hazardous wastes, and to reduce the regulatory burden on businesses that wish to collect or generate these wastes. Although handlers of universal wastes face less stringent standards for storing and collecting wastes, and transporters for moving wastes, than handlers and transporters of hazardous waste, the waste must comply with full hazardous waste requirements for final recycling, treatment, or disposal. This management structure helps remove these wastes from municipal landfills and incinerators. In July 1999, EPA added hazardous waste lamps to the *Universal Waste Rule*, which already covered some batteries, thermostats, and pesticides. In 2005, EPA added other mercury-containing wastes to the UWR.

End-of-life vehicles. In an effort to reduce mercury emitted from electric arc furnaces that consume scrap from recycled automobiles – which US EPA estimates emit about 8-12 tons of mercury per year – US EPA has pursued multiple programme efforts to encourage the removal of mercury switches from scrap automobiles prior to recycling. Nearly all obsolete automobiles in the US are dismantled and shredded to recycle the metal (ACAP, 2004). The US *Clean Air Act* gives the US EPA authority to regulate the steel mills – but not the car dismantlers. However, there is little the steel producers can do about the problem, other than installing expensive end-of-pipe cleaning equipment. The car dismantlers can relatively easily take out the switches before the cars are flattened – but they have by themselves no incentive to do so. Hence, the *National Vehicle Mercury Switch Removal Program* (NVMSRP) Memorandum of Understanding (MoU) was signed in August 2006. The MoU is with automobile and steel manufacturers, scrap recycling, vehicle dismantling, environmental groups, and the States. This is a voluntary programme to reduce the presence of mercury switches originating from automobile convenience lighting and anti-lock brake systems from the scrap metal supply chain. Removing switches before automobiles are crushed and sent to electric arc furnaces will prevent up to 75 tons of mercury emissions over the next 15 years [see US EPA (2006b)].

Stockpiles of mercury. The US Government maintains a supply of mercury as part of the *National Defense Stockpile*, established at the end of World War I to maintain adequate supplies of materials deemed critical to national defense. The Defense Logistics Agency (DLA), a unit of the Department of Defense, manages the stockpile.[11] The *Strategic and*

Critical Materials Stockpile Act regulates mercury that the DLA sells from this national stockpile. In July 1994, DLA suspended future mercury sales, pending analysis of the environmental consequences (such as national or global mercury releases resulting from the use of sold mercury). An Environmental Impact Statement to determine the disposition of the stockpile was completed in 2004. In the meantime, a complete review of the four facilities across the US currently storing its mercury and inspection of all the mercury-containing flasks to ensure proper and safe storage was undertaken. The US Department of Defense announced that its "preferred option" is consolidated storage of its mercury at one location for at least 40 years, and this option has now been chosen for implementation.

Also, US EPA (2006c) commits the Agency to establishing an inter-agency process in 2006 to address how to best manage domestic supplies of commodity-grade mercury, in light of an emerging global mercury surplus that is expected over the next ten years.

6.4.3. State-level instruments addressing mercury

Restrictions on mercury-containing products, once used sparingly by the Federal Government, are increasing rapidly at the State level. Certain US States have recently initiated a variety of initiatives aimed at reducing mercury releases from the use and disposal of products. These initiatives include notification and labelling requirements to gain information on the mercury content of particular products, and to inform purchasers that products contain mercury; prohibitions on the sale of a variety of products for which alternatives were deemed readily available, such as fever thermometers, dairy manometers, novelty items (toys, shoes), switches in automobiles, and thermostats in residential and commercial applications; concentration limits on other products such as batteries and packaging; restrictions on product disposal so that the products must be segregated from the solid waste stream and ultimately recycled; and State-sponsored collection programmes for items such as fever thermometers, historic dental inventories, and products found in schools [see ACAP (2004)].

A recent overview of State instruments and activities on mercury is provided in ECOS (2005).[12] This overview is based on a questionnaire to States which was answered by 45 States. It provides summary text of the States' mercury-related activities as of 2005. New activities have also been introduced since 2005; those pertaining to coal combustion in power production are described in Section 6.4.6.

According to ECOS (2005), many States are co-ordinating their mercury activities via an action plan or strategy or through task forces and workgroups; even more States are addressing mercury in consumer products. More than 40% of the states responding to the survey have taken steps to curb mercury use in products and to prevent improper disposal. States' activities range from phase-outs and bans to labelling and collecting mercury and mercury-containing products. The most common activity related to mercury-containing products is voluntary mercury take-back programs – 80% of the States reported sponsoring such activities and over one-third of the States also reported having phase-outs and bans on the sale of some products.

Many States are also taking action to address mercury-containing switches in vehicles. Nearly half of the States reported proposed or enacted vehicle-switch legislation. Although only a handful has mandatory switch removal, over a quarter of the States had activities related to voluntary switch removal from end-of-life vehicles. While States reported facing many challenges in dealing with the issue of vehicle switches, the most frequently cited was the lack of funding for removal programmes.

Two additional mercury sources that States are beginning to address are electric utilities and the dental sector. Emissions from coal-fired electric power plants are a widely recognised problem. Twenty-four States report that these are the largest sources of mercury emitted within their jurisdiction. States are also beginning to look for ways to reduce mercury from dentists. Four States reported having regulations on mercury-releases from dental separators; eleven States take into account the number of dentists with amalgam separators when quantifying their mercury-reduction progress.

All States responding to the conducted survey were engaged in some mercury-related outreach or education activity. Nearly all also perform some form of mercury monitoring [see ECOS (2005)].

The Northeast Waste Management Officials' Association (NEWMOA) has been instrumental in establishing mercury-reduction efforts in the northeast part of the US, including the development and implementation of the *Interstate Mercury Education and Reduction Clearinghouse*.[13]

While all States in the US address mercury emissions, the subsequent two sections present, as examples, more information on the instrument mixes applied in the States of Massachusetts and Michigan.

6.4.4. Instruments addressing mercury in the State of Massachusetts[14]

Massachusetts has an overall mercury action plan with the goal of virtually eliminating all anthropogenic sources of mercury and a milestone goal of 75% reduction by 2010. Major elements of the plan include: small business, household, medical and dental mercury waste management; strict mercury emission limits; limiting mercury discharges to water; reduction of mercury-use in products; recycling; and outreach and education. The State has various policies, statutes, and regulations relating to the mercury action plan.

Massachusetts has strict State regulations on mercury releases from coal-fired power plants (85% reduction efficiency by 2008 and 95% by 2010), for municipal waste incinerators (three-fold more stringent than Federal EPA requirements), for dental offices, and for wastewater treatment. Massachusetts also has a voluntary programme that provides dentists with an incentive to install amalgam separators and adopted regulations in 2006 requiring dentists to install amalgam separators.

Massachusetts has banned the sale of mercury thermometers and completed a State-wide collection program that recycled more than 95 000 thermometers. Massachusetts' waste incinerator rules require facilities to implement mercury source separation plans (separate collection) in their respective geographical areas. Mercury products are managed as hazardous wastes and recycled. Massachusetts monitors collection activities through audits, inspections, and self-certification provisions. The Massachusetts Department for Environmental Protection supports mercury cleanouts and education programmes in schools as well as other community recycling efforts.

According to the Massachusetts Department of Environmental Protection, a 1996 report on mercury, and subsequent work, found that there are significant local impacts of mercury emissions to air in North-eastern parts of the State. This is part of the reason that Massachusetts has decided to opt out of the Federal mercury rule for coal-fired power plants – in order to avoid contributing to or creating local "hot spots" or disproportionate impacts. The State has also found that it may be impossible to meet *Total Maximum Daily*

Loads for mercury for their freshwaters (so-called "TMDLs" – see Section 3.7.1), as required under the *Clean Water Act*, with the US EPA approach to mercury emission management. Massachusetts Department of Environmental Protection also concluded that emissions could be reduced from coal-fired energy-generating utilities to a much greater degree and on a faster timeline (MA DEP, 2006).

Again according to the Massachusetts Department of Environmental Protection, an integrated approach taking into account all mercury-release sources (including products, etc.) and all environmental media is necessary to achieve acceptable levels of mercury exposure.

The Massachusetts Department of Environmental Protection has undertaken substantial monitoring in the State and has already seen reductions in mercury levels in fish of some 20-30%, after some local sources reduced their emissions 80-90% around 2000/01 (MA DEP, 2006).

6.4.5. *Instruments addressing mercury in the State of Michigan*[15]

Michigan has an overall mercury action plan. Major elements of the plan include: medical/dental mercury waste management; limits on mercury discharges into water; reduction of mercury-use in consumer products; technical assistance for industries; mercury recycling; and public outreach and education.

State regulations also exist for mercury-releases from wastewater treatment and industrial facilities, municipal waste incinerators, and medical waste incinerators. Air permits for certain sources (such as shredders and sewage-sludge incinerators) include mercury-specific limits or Best Management Practices (BMP) requirements.

Michigan has a mercury-free purchasing policy for State procurement. The State has phased out the sale of mercury-containing thermometers, and the use of mercury-containing devices in certain schools, and the sale of mercury-containing batteries that exceed allowable mercury amounts. Separate collection programmes also exist for elemental mercury, mercury waste, and mercury-containing products.

There is also a programme for voluntary removal of mercury switches (hood lights, trunk lights, and ABS brake sensors) from end-of-life non-commercial vehicles. Vehicle manufacturers provide training materials, buckets, pick-up and transport of collected switches. The mercury collected in this initiative will be put into permanent storage, instead of being recycled to the international market. Dismantlers receive payments for switches that are collected separately.

Michigan is not going to participate in the trading program under the 2006 Federal *Clean Air Mercury Rule* for mercury emissions from power plants. Instead, they require a 90% reduction from input mercury levels in coal by 2015 or an emission standard (MDEQ, 2007).

An industrial-scale pilot experiment was started at a 270 MW coal-fired power plant in Michigan in 2006, using active carbon technology to reduce mercury emissions. Preliminary parametric testing has demonstrated that a 90% reduction in mercury emissions is possible (MDEQ, 2007).

6.4.6. *More information on instruments addressing mercury emissions from coal combustion*

The largest anthropogenic source of mercury emissions in the US is currently coal-fired power plants. Utility steam generating sources were subject to special study and required a determination by the US EPA as to whether it would be "appropriate and

necessary" to regulate them under Section 112 of the *Clean Air Act*.[16] In 2000, US EPA released its *Regulatory Finding on the Emissions of Hazardous Air Pollutants from Electric Utility Steam Generating Units*. The Agency concluded that regulation of hazardous air pollutants under Section 112 from coal- and oil-fired electric (but not natural gas-fired) utility steam generating units was "appropriate and necessary", and that mercury was the air toxic of most serious concern. In 2004, the US EPA proposed alternative approaches to regulating mercury from coal-fired power plants. Under one approach, the Agency would continue along its original path and establish MACT standards for hazardous air pollutants. Under another approach, the Agency would revise its 2000 finding that it is "appropriate and necessary" to regulate utility hazardous air emissions using the MACT standards provisions (Section 112) of the *Clean Air Act*. This would give EPA the flexibility to consider a more efficient and more cost-effective way to control mercury emissions, based on a "cap-and-trade" system (US EPA, 2006).

As mentioned above, US EPA issued the *Clean Air Mercury Rule* (CAMR) in 2005, which is based on the cap-and-trade system and new source performance standards for new units. A total national "cap" on annual mercury releases from coal-fired power-plants is shared among the participating States and Tribes, who decide on how their total number of allowances should be allocated to individual power plants.[17, 18] Facilities facing high emission reduction costs, due to their state of technology or other factors, can buy allowances from facilities that can reduce mercury releases at lower cost. Facilities that reduce their emissions below the number of allowances they hold for a given year can sell the "surplus" allowances in the market, or "bank" them for use in a later year.

The *Clean Air Mercury Rule* works with two phases. The total mercury cap in Phase 1 is 38 short tons per year (to be achieved by 2010), and in Phase 2 (by 2018) the cap is set at 15 short tons per year. The 38 tons cap is expected to be fulfilled solely by co-benefits of the 2005 US EPA *Clean Air Interstate Rule* that significantly tightens the caps on SO_2 and NO_x emissions from power plants. That is, no mercury-specific reduction equipment will be required. Some power plants would, however, probably undertake some such measures – which would generate bankable permits that could be used, once the total cap is reduced to 15 tons in 2018.

A number of US States are concerned that the cap-and-trade system will not secure sufficient reductions on the local or regional level where specific facilities are major sources of mercury pollution in the local or regional environment. In addition, some States (and other opponents) claim the *Clean Air Mercury Rule* does not meet their assessment of the needs for mercury reductions.

For example STAPPA, the "State and Territorial Air Pollution Program Administrators", has suggested a model rule for States and localities' that would aim for a 90% reduction by around 2015, whereas US EPA aims for a 70% reduction target of mercury emissions from power plants by 2018. One reason for setting a more ambitious target than US EPA is, according to STAPPA, that the costs of using activated carbon to clean the end-of-pipe emissions are rapidly falling (STAPPA, 2006).

Concerns about "hot spots" have been raised by some,[19] despite the success and growing use of cap-and-trade programmes. The US EPA believes that a trading approach will help to address this problem. In addition to the reductions required by the cap, all States would have the ability to address local health-based concerns separate from the mercury cap-and-trade programme requirements (US EPA, 2006b).

All in all, out of the 48 States that have coal-fired power utilities, about two-thirds are planning to introduce mercury permit trading, according to US EPA (2007). Of these, three-fifths will implement the CAMR "as-is". About one-fifth of the States with trading will have more stringent, direct (*e.g.*, permit limits) local control requirements in addition to trading, whereas about one-fifth of these States will introduce trading, but will hold back part of their State-wide emissions quota. About one-third of the 48 states with coal-fired power utilities will either not participate in the trading programme or had not as of March 2007 provided clear indication of their plans. Those that have indicated that they will not participate in the trading programme will likely introduce more stringent direct controls.

6.4.6.1. Differentiation depending on domestic coal types

The 2005 *Clean Air Mercury Rule* also contains emission limits on new coal-fired power plants that vary with the type of coal being used (because of differences in mercury concentrations and other chemical characteristics that influence mercury retention efficiency in control equipment). In principle, such variations in the emission limits are unfortunate – since they tend to facilitate the use of coal with high mercury content.[20] It was a priority for the EPA *not* to change the fuel choices of the power plants; the fuel choice could in any case be more determined by the price and the sulphur content than by the mercury content of each fuel. It was important to keep all types of domestic coal reserves viable for power production to provide national energy reliability and security, according to US EPA (2006b).

6.4.6.2. Potential for inclusion of other source categories in the trade-and-cap system

US EPA did not see any engineering or economic rationale for allowing other point sources to take part in the trading scheme that is being prepared for controlling mercury emissions from the coal-fired power plants; however, under the *Clean Air Interstate Rule* (CAIR), there are opt-in provisions for SO_2 and NO_x control from non-utility sources. This was reasoned partly by administrative factors – different sources are regulated through different sections of the *Clean Air Act* and a revision of the Act could not be envisaged at present – partly by technical differences between mercury source categories. For example, according to the US EPA (2006a), there would be few benefits by including waste incinerators in the same trading scheme as the power plants, because all the incinerators in the US are relatively similar and can be governed efficiently and economically by standardised emission limit schemes. However, for coal-fired power plants there are significant engineering differences that render traditional command-and-control approaches less effective. In a market-based approach, such as CAIR and CAMR, the large plants are expected to invest in mercury-specific abatement measures early in the transition period, while the small power plants would tend to buy pollution allowances instead.

6.4.6.3. "Safety valve" mechanism

In the original proposal for the *Clean Air Mercury Rule*, where the total cap in the first period was set at 34 tons per year, a safety valve provision was included in order to minimize unanticipated market volatility and provide more market information that industry could rely upon for compliance decisions. The safety valve mechanism should ensure that costs of controls did not exceed a certain level, but also ensures that emissions reductions were achieved. The final *Clean Air Mercury Rule* did not contain a safety valve provision. The total cap of the trading system was, however – for unrelated reasons – increased from 34 to 38 tons per year.

Notes

1. Jakob Maag, COWI A/S, Denmark, prepared the 1st draft of the case study on which this chapter is based.
2. In addition to the directly referenced information provided in this section, substantial information provided in the Norwegian questionnaire response to this study was used in this section.
3. See *www.shdir.no/vp/multimedia/archive/00001/IS-1136_1661a.pdf*.
4. Under the Objective "A Non-Toxic Environment", see *www.miljomal.nu/english/english.php*.
5. In addition to the directly referenced information, substantial information provided in the Swedish questionnaire response to this study was used in the description here.
6. OSPAR: The Convention for the Protection of the Marine Environment of the North-East Atlantic.
7. A mercury-detecting dog has also been used in the State of Minnesota in the US, helping to collect 2 tons of mercury from schools.
8. See *www.epa.gov/ttn/atw/hgcellcl/hgcellclpg.html*.
9. See *www.epa.gov/Region5/air/mercury/reducing.html#heavy%20industry*.
10. See *www.nema.org/gov/ehs/trc/*.
11. In addition, the US Department of Energy has stocks of approximately 1 200 tonnes of mercury.
12. See also OECD (2007b).
13. See *www.newmoa.org/prevention/mercury/imerc.cfm*.
14. This section is based on ECOS (2005) and MA DEP (2006).
15. This section is based on ECOS (2005) and MDEQ (2006).
16. Section 112 of the Act requires EPA to first establish Maximum Achievable Control Standards (MACT) for hazardous air pollutants. MACT standards are applicable to both new and existing sources. For existing sources, MACT standards must at a minimum reflect the performance of the "best" 12% of existing sources in the category. For new sources, MACT standards must at a minimum reflect the performance of the "best controlled similar source". Under Section 112, the Administrator must also, within 8 years of establishing MACT standards, set additional standards to protect human health and the environment with "an ample margin of safety" from any residual risks remaining after the application of MACT.
17. "… although there are some clear potential benefits to using auctions for allocating allowances…, EPA believes that the decision regarding utilizing auctions rightly belongs to the States and Tribes. EPA is not requiring, restricting, or barring State use of auctions for allocating allowances", US EPA (2005a).
18. 5% of the allowances will go to a new source set aside, see US EPA (2005a).
19. According to US EPA's definition, "hot spots" are: "Localized areas with unacceptably high levels of mercury." See *www.epa.gov/oig/reports/2006/20060515-2006-P-00025.pdf*.
20. With a cap-and-trade system in place, such differences in the emission limits will, however, not have any impact on *total* mercury emissions – which will be governed by the cap.

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Chapter 7

Analysis of Current Use of Instrument Mixes

This chapter demonstrates that many of the instrument mixes analysed find support in theoretical arguments for using instruments in combination, e.g. *by reflecting the multi-aspect character of many of the environmental issues addressed. Many instrument mixes are also applied to reduce information problems and other "failures" in the markets in which environmental policy operate, and in several cases one instrument enhances the functioning of other instruments applied. However, in a number of situations, the use of overlapping instruments reduces the economic efficiency of the mix, while in other cases, the lack of some instruments can jeopardise environmental effectiveness and/or economic efficiency.*

7.1. Introduction

This chapter analyses the current use of instrument mixes, with a particular focus on the impacts this usage has on environmental effectiveness and economic efficiency, drawing on the country case studies described in the preceding chapters. The first sections below refer back to the theoretical arguments in favour of the application of a mix of instruments discussed in Chapter 1, and provide examples of how countries "have actually put this theory into practice". Subsequent sections focus on various problems that have been identified with some of the instrument mixes that have been examined, perhaps because some instruments are lacking from the mix, or because some of the instruments conflict with each other. The last sections provide examples of the extent which the combination of instruments applied in the different countries have proven (reasonably) environmental effective, and to what extent environmental improvements have been achieved in an economically efficient way.

7.2. Addressing the multi-aspect character of the environmental issue at hand

In Chapter 1 it was emphasised that most environmental problems are of a multi-aspect character. In line with the "Tinbergen rule", this makes the application of several policy instruments necessary in order to reach an "optimal" outcome. A single instrument would simply not be able to simultaneously address adequately all the relevant aspects of a multi-aspect problem.

This *is* an important part of the motivation behind many of the instrument mixes applied in OECD member countries, and many examples can be found in the case studies done for this project.

For instance, both the EU *Nitrates Directive* and the instrument mixes applied in the four countries studied regarding non-point sources of water pollution (Denmark, the Netherlands, the United Kingdom and the United States) take into account that the environmental damage caused by nutrients run-off depends both on the *total amount* of nutrients applied and on *how*, *where* and *when* the nutrients are applied. For example, within "vulnerable zones" according to the EU *Nitrates Directive*, the maximum application of *animal manure* is 170 kg N per ha and year.[1] However, restrictions must be more stringent if this limitation is not enough to secure achievement of the set water quality objectives. The Directive further requires "codes of good agricultural practice" to be established that indicate:

- periods when the land application of fertiliser is inappropriate;
- rules for the application of fertiliser to steeply sloping ground;
- rules for the application of fertiliser to water-saturated, flooded, frozen or snow-covered ground;
- conditions for land application of fertiliser near water courses;
- procedures for the land application, including rate and uniformity of spreading, of chemical fertiliser and livestock manure, etc.

All these – and a number of other – issues can indeed impact importantly on the environmental damages caused by nutrients (mis-)management. Similar issues are addressed in Nutrients Management Plans in the States of Virginia and Maryland in the United States.

In a similar way, there are multiple aspects to the problems pesticides can cause to the environment. *Total amounts* of pesticides use can be of importance, but it can again be just as important *which* products are applied, *how* they are applied, *when* they are applied, *where* they are applied, *how* the spraying equipment is maintained and cleaned, etc. The two countries where instrument mixes addressing pesticides use have been studied in this project (Denmark and the United Kingdom) both have instruments in place that deal with all or most of these aspects. For example, authorisation systems for pesticides keep the most dangerous products completely off the market, while codes of practice inform (and partly instruct) farmers on how they should apply the products that are authorised.

Emissions to air of mercury also stem from a number of different point-sources, dominated by a few large emitters, and large number of products and diffuse sources which have more complex release- and exposure patterns, with waste incineration being a major component. In addition, mercury can evaporate from deposited mercury-containing waste and from land where such waste was previously deposited, as well as from natural mercury sources. Further, while some of the mercury emitted to air will mix in the atmosphere and contribute to a global problem, some of the mercury compounds will deposit closer to the pollution source, thus representing a local and/or a regional environmental problem. Hence, the "science" of the mercury problem calls for instruments to address emissions from point sources, instruments to help isolate mercury in the waste streams, and instrument to address the temporary or permanent storage of mercury-containing wastes, etc. The three countries analysed (Norway, Sweden and the United States) all have instruments in place that seek to address all – or most – of these different aspects of the problem.

For example, Sweden has a broad spectrum of emission standards for point-sources of mercury emissions. It also seeks to prevent mercury from coming into the economy, by strictly regulating the sale of mercury-containing products, by not reimbursing medical expenses if dental amalgam is used, etc. Further, significant subsidies have been applied to promote the collection of mercury from a broad range of historical sources. In addition, all the mercury that has been collected is to be stored in special facilities deep underground, in order to prevent future evaporation. It would not be possible to address all the relevant aspects of the mercury problem properly (only) with a single instrument, like for example a single tax or trading system.

The instrument mixes that were at the centre of the case studies on regional air pollution arising from stationary sources (Canada and Sweden) also take into account the *spatial differentiation* of impacts for acidification. For instance, to reduce NO_x and SO_2 emissions in Canada, emissions trading programmes are applied in combination with individual emission permits at the facility level in some provinces (*e.g.* Ontario and Alberta). This site-specific approach is used to address local pollution effects ("hot spots"). In Sweden, a NO_x charge and a SO_2 tax are applied as complements to direct regulation. Maintaining the NO_x and SO_2 emissions standards is meant to ensure a minimum environmental quality level. In addition, EU Directives with which the country is obliged to comply, make explicit reference to limit values to tackle to issue of local air quality. For instance, the *National Emission Ceiling Directive* uses the notion of "critical loads" to define interim environmental

objectives set for the year 2010.[2] Geographically differentiated taxes rates could have been introduced to address the spatial component. Geographically differentiated standards – like the Norwegian geographically differentiated regulation on the sulphur content of fuel oil – could have served a similar purpose. Designing instruments to address local problems can be complex, and it may be that the associated economics costs are larger that the environmental benefits obtained. In addition, as total depositions in Sweden are larger than total emissions, attaining a reduced level of deposition at the local level also strongly rests on international co-operation to tackle acidification.

Household waste causes environmental damage also when it is managed properly, through incineration, composting, landfilling, etc. Hence, there is a need for instruments to limit waste generation and to stimulate recycling and reuse. There is also a need for instruments that limit the environmental damages related to final disposal. At the same time, there is a risk that household waste could be disposed of in *illegal* ways, which could cause even greater harm to the environment than properly managed waste. Hence, instruments need to be in place to discourage illegal behaviour too.

Jurisdictions where the instrument mixes addressing household waste have been studied in this project (the Netherlands and the United Kingdom – in particular England and Wales) have indeed put in place instruments to limit waste generation, to promote reuse and recycling, to limit environmental harm caused by legal forms of final disposal, and to discourage illegal disposal.[3] For example, both countries apply taxes on landfilling (which could provide incentives for limiting waste generation); both countries have various subsidy and information programmes to promote reuse and recycling; both countries have set standards that operators of final waste treatment plants have to comply with, etc.[4] However, the landfill taxes (currently) in use are more aimed at promoting other waste management options than internalising the environmental externalities stemming from landfilling.[5] Further, only in some cases in the Netherlands are the landfill taxes passed on to households in the form of incentives at the margin to limit the amounts of waste set out for final disposal.

A "multi-aspect" character is not *dominant* for *all* environmental problems. For example, if the interest in residential energy efficiency stems mainly from the related impacts on greenhouse gas emissions,[6] one is closer to a "single-aspect" problem. Even if one could be interested in the energy efficiency of very different "objects" such as the buildings/dwellings or products and appliances, it would seem possible to address most – if not all – of the relevant *environmental* dimensions of this problem through one instrument, such as a tax based on the greenhouse gas emissions caused by different energy products, or an emission permit trading system with a similar basis.

However, there can be various market failures in the residential sector that make the use of several instruments necessary, even if the underlying *environmental* problem only should have one "dimension" (see Section 7.3).

A policy lesson that can be drawn from this section is that an assessment should be made of whether the environmental problem at hand is of a "single-aspect" or a "multi-aspect" character. In the latter (more common) case, a *combination* of instruments is likely to be required to address the problem adequately.[7]

7.3. Addressing non-environmental market failures

Correction of various non-environmental market failures – that co-exist with the environmental externalities that are the prime focus of the environmental polices examined in this project – is also an important case for the application of several instruments in combination, rather than just one, both from a theoretical point-of-view, and for practical policy formulation. As is well-known from the literature on "the second best", (at least) one instrument per market failure present is needed [*cf.* Lipsey and Lancaster (1956)]. Several types of non-environmental market failure have been of importance in the cases examined in this project.

7.3.1. Information failures

One category of market failure often deemed to be of importance in relation to environmental problems is that various relevant decision-makers are lacking sufficient information to make "rational" or "correct" decisions.

In order to address such problems, many countries undertake various information campaigns and employ a number of labels or logos to provide better information to decision-makers. For example, the United Kingdom combines the EU energy efficiency label – that rates products from the most efficient, down to the least efficient – with the "energy saving recommended" logo for the most energy efficient products, in order to better inform households on the energy use of various appliances.

When considering the impacts of using such instruments, one should keep in mind that in the area of residential energy efficiency, various labels, logos, etc., will primarily provide information on *private benefits* households can obtain by making choices that reduce their energy consumption. There is empirical evidence that improved information can significantly impact on consumer behaviour in such cases.[8]

Information instruments will, however, only play a limited role in improving environmental performance if this in practice was fully determined by other instruments, such as legally binding energy efficiency standards set for the various product categories. Better information will only be of much relevance if people have the option to make a *choice*.

Nor would providing information to households by itself suffice to guarantee an "optimal" environmental outcome. There is no reason to think that the "*public good*" elements associated with reduced energy consumption will be fully reflected in households' decision-making, even if they were "well informed".[9] Hence, measures to address information failures should be combined with instruments that more directly target the environmental externality. For example, information-based instruments can complement a tax designed to internalise relevant environmental externalities (*e.g.* in the UK case, a carbon tax that also covers domestic energy use). A label would help increase consumers' awareness as regards energy use, and hence increase the price-elasticity of energy demand, making a tax on energy more environmentally effective and economically efficient. Likewise, the tax could make consumers pay more attention to a logo or an energy labelling scheme – hence increasing the effectiveness of these instruments.[10] A virtuous circle would thereby be established.

Information to farmers also plays an important role in most of the instrument mixes aimed at non-point sources of water pollution in agriculture that have been examined in this project. For example, great emphasis is placed on providing better information and training to farmers in the Voluntary Initiative on pesticides use in United Kingdom.[11] The cases

studied illustrated that guidance and information instruments, including Nutrient Management Plans *can* well be used to address *how* to apply a given amount of nutrients or pesticides in such a way that the environmental impacts are as benign as possible. However, it is not certain that better informed and trained farmers *will actually behave differently* than before, as long as they do not receive an economic (or other) incentive to do so.[12] A tax on pesticides (as in Norway, where the tax is based on the recommended area-doses of the different products)[13] could provide such an incentive – at least as concerns the *total amounts* of pesticide use.

Christensen and Huusom (2003) made similar observations regarding information instruments used to promote better pesticide practices in Denmark. They emphasised that it is not enough to make agronomical knowledge available to the farmers in order to secure further reductions in pesticide use. For this knowledge to be converted into action, farmers need to be motivated to limit the pesticides use. A direct incitation has to be in place for each farmer, in order for the pesticides usage to decrease.

Another issue concerning information instruments has come up in several of the case studies: quite often the information documents, user guides, "best-practice" documents, etc., that are issued seem rather long. This is certainly in part because the environmental issue at hand is complex, thereby requiring many considerations to be taken into account. And often a detailed "step-by-step" explanation can be of more help to the user than a summary description.[14] However, the lengthy documents might reflect that the instrument being explained is too complex. There is also the danger that the persons targeted by the information documents simply will not read them – at least not carefully.[15]

Hence, policy makers should consider carefully whether any information instrument is well adapted to the audience one is seeking to influence. While there is some evidence that instruments that convey information regarding private benefits have been effective in influencing behaviour, the extent to which information instruments in general in fact do help overcome existing "information failures" is an empirical question that goes beyond the scope of this project.

7.3.2. Asymmetric information

A special example of information failures – which perhaps represents more of an equity problem than an environmental problem *per se* – occurs when buyers and sellers have *asymmetric information* about the (environmental) characteristics of a given product.[16] In the housing market, for example, the seller normally knows (at least to a certain extent) more about what the real quality of the building is – for instance, as regards thermal isolation. In order to address this problem, anyone selling a home in the United Kingdom after 2007 will be required to put together a *Home Information Pack* for potential buyers, which will bring together key information about the quality of the house, including some of its environmental characteristics.[17] This measure is meant to make the home buying and selling process more transparent and thereby reduce transaction costs – *inter alia* providing home-buyers with a better picture of the future heating expenditures they can expect.[18]

Obviously, preparing such information will trigger significant costs – which in practice will be shared between the sellers and buyers. If the *environmental* benefits of improved information on the energy efficiency of the house do not outweigh the related costs, it is *possible* that other benefits of improved transparency in the housing market could be sufficiently large to outweigh these costs.

7.3.3. Incomplete property rights

Another argument for using more than one instrument to address a particular environmental problem is the existence of "incomplete property rights". An example of this is the fact that inventors do not reap all the economic benefits of the inventions they (potentially would) make. Whereas the possibility to patent inventions goes some way to limit this problem, subsidies to stimulate innovations beyond what would take place in an unregulated market could hence be appropriate. The cases studied include several examples of various types of subsidies for technology development.

For instance, several of the cases regarding non-point sources of water pollution point to efforts aimed at bringing about better feeding regimes for different animal categories – and major improvements have been made, leading to significant reductions in nutrient losses.[19] Much research has also been undertaken to provide better knowledge about the links between fertiliser applications and nutrients run-off, etc.

Similarly, support for the development of clean technologies has played an important role in addressing regional air pollution in Canada.[20] The Federal *Programme for Energy Research and Development* (PERD),[21] has encouraged the development and diffusion of improved technology for coal-burning power plants.[22] In addition to using a combination of direct regulation and economic instruments address acidification, Sweden provided approximately EUR 44 million in financial support to stimulate innovation to reduce NO_x and SO_2 emissions over the period 2003-07.

While subsidies for technology development *can* lead to welfare improvements, policy-makers should keep in mind that there is a risk of significant dead-weight losses – meaning that the support may be going to projects that would have been undertaken anyhow. One should also take into account that there are efficiency losses related to the financing of any subsidy scheme, because subsidies increase the need for taxation that normally will cause structural distortions in the economy.

Another situation where one can talk about incomplete property-rights concerns situations involving "split incentives" between landlords and tenants – for example, regarding investments in improving the energy efficiency of buildings. Landlords will hesitate to make such investments, because a major part of the benefits – in the form of lower expenditures on heating (or cooling) the building – will inevitably accrue to tenants. Tenants will also hesitate to undertake such investments, because the some of the benefits – the increased quality of the building – will accrue to the landlord. However, if the tenant intends to live for a long time in the building in question, this problem will diminish in importance.

Landlords will especially lack incentives to improve the energy efficiency of their buildings if *e.g.* public regulations or long-term contracts prevent them from increasing the monthly rents, in order to better reflect the services provided.

A policy lesson that can be drawn here is therefore that one should consider carefully the incentive impacts of various regulations in the housing market. Many such regulations are introduced with an aim to protect consumers in general and low-income households in particular – but some of them might have unintended side-effects that are not desirable from an environmental perspective.

Landlords who pay income tax in United Kingdom can, through the *Landlords Energy Saving Allowance*, receive upfront relief of up to GBP 1 500 on capital expenditure for the installation of loft insulation, cavity wall insulation and solid wall insulation in residential

property which they rent. In addition, the UK Government intends to make a *Wear and Tear Allowance* for property rents conditional on the energy efficiency level of the property. While the underlying differences in incentives between landlords and tenants will remain, this package of instruments will increase the amount of investments in energy efficiency that landlords will find profitable to undertake, and will provide stronger incentives for them to improve the quality of their property. Whether the *additional* benefits to society as a whole of these energy efficiency improvements – beyond what will be achieved through the information provision described in Section 7.3.2 – exceed the costs involved, is an empirical question beyond the scope of this project.

7.4. Limiting compliance-cost uncertainty

Economic theory suggests that a tax (or fee) could be added to a tradable or non-tradable quota system, in order to limit uncertainty about the costs of complying with the quantitative restriction embedded in the quota system. For example, instead of having to pay large amounts per allowance in a tradable permit system, a firm might be allowed to emit as much as it wants, by paying a (high – but predetermined) fee per unit emitted, for which it does not hold an emission permit.[23] The case studies undertaken in this project have, however, not found any examples of such "safety valves" in trading programmes.[24] They have instead found several situations where quota systems and taxes are being used in parallel, but where the taxes do *not* play a role as a "safety valve".

In the original proposal for the *Clean Air Mercury Rule*, where a trading system that would "cap" total emissions of mercury from coal-fired power plants in the United States at 34 tons per year was proposed, a safety valve provision was included. Federal Register (2004) states:

> "Under the safety valve mechanism, the price of allowances is capped, meaning that if the allowance price exceeds the 'safety-valve', sources may borrow allowances from following years to have access to more allowances available at that price. The EPA proposes a price of USD 2 187.50 for a Hg allowance (covering one ounce). This price will be annually adjusted for inflation. The Administrator will deduct corresponding allowances from future facility allowance accounts.
>
> The purpose of this provision is to minimize unanticipated market volatility and provide more market information that industry can rely upon for compliance decisions. The safety valve mechanism ensures the cost of control does not exceed a certain level, but also ensures that emissions reductions are achieved. The future year cap is reduced by the borrowed amount, and the emissions reductions are achieved."

If the provision had been introduced as described here, the environmental integrity of the trading scheme would *not* have been jeopardized by the safety valve. It was, however, also noted that:

> "this proposed approach may create implementation problems associated with the need to 'reconcile' at some point in time the allowances borrowed from future compliance periods."

The final *Clean Air Mercury Rule* did *not* – probably partly for this reason – contain a safety valve provision. The total cap of the trading system was, however, for other reasons, increased from 34 to 38 tons per year, *cf.* US EPA (2005b).

Increased compliance-cost certainty in a permit trading programme can also be achieved by keeping permit reserves. For instance, under the SO_2 Allowance Trading program in the United States, the Government initially held reserves of permits which it could have

released onto the market if the price per permit had reached USD 1 500.[25] In practice, permit prices never approached this threshold, but some initial estimates were sufficiently high to elicit some concern about compliance-costs. However, the "permits reserve" option has the disadvantage that the price can only be capped for as long as the reserve exists – excessive demand could eventually drive the price higher. Thus, the price effects are less certain, which could undermine the benefits in terms of reduced compliance-cost uncertainty. On the other hand, of course, the environmental effects are more certain with a permit reserve than with a fee-based "safety valve" without additional provisions [see OECD (2003b)].

The cases discussed so far in this chapter are examples where "the logic" behind a choice (or a proposal) of instrument combination is similar to arguments in favour of using instrument mixes found in economic literature. It can, however, also be useful to look closer at some cases where the motivation for a particular instrument mix seems to be different than what economic theory suggests. In relation to a discussion of compliance-cost uncertainty, it is for example also of interest to consider cases where quotas and taxes are jointly applied, but where the tax does *not* serve as a safety valve.

One example of such instrument mixes concerns household waste in United Kingdom, where the tax on landfilling is combined with statutory landfill diversion targets and – in the case of England and Scotland – with a trading scheme for landfill allowances. If a local waste disposal authority does not comply with its *Statutory Landfill Diversion Targets*, it could *not* just pay the *Landfill Tax* for the amount exceeding the target – as could have been the case if the tax were used as a "safety valve". Instead, there are separate sanctions – considerably higher than the rate of the Landfill Tax and considerably higher than the current trading price of a landfill allowance – for any non-compliance with the landfill diversion targets.[26]

The levies of the nutrient accounts system (MINAS) in the Netherlands were combined with a *Manure Transfer Agreement System*, where farms that produced more manure than the amount that could be legally applied on their own land were obliged to have a contract with other farms, where the surplus manure could legally be applied. However, the quantitative instrument was put in place long after the introduction of the price instrument, and paying the MINAS levies was not a way to avoid finding other farmers that would be willing to receive the surplus manure.

In Denmark, a (non-tradable) nitrogen quota system is combined with a system of legal fines. However, the fines are set at such a high level that they can hardly be seen as a "safety valve" for the quota system. For almost all farms, the marginal costs of reducing the nitrogen surplus would likely be lower than the associated fines for non-compliance.

In all these three cases, assuring compliance with the rules of the EU Directives on landfilling or nitrates seem to have been of great importance for the introduction of the quantitative restrictions of the quota systems – although the (additional) environmental impacts of the manure transfer system in the Netherlands have been questioned [see Wossink (2004)]. Based on this, it would perhaps have been strange to include "safety valve" provisions that could limit the environmental effectiveness of the respective quota systems.[27]

It is of relevance here to point out that both landfilling of waste and over-application of nutrients to a large extent are "*stock problems*", rather than "*flow problems*": the environmental damage is more related to the total amount of waste put into a landfill – or the total amount of nutrients applied in a given area over a certain time period – than to the respective amounts *per year*. However, the EU Directives on landfilling and on nitrates set annual ("flow") targets, placing limits on how much waste may be landfilled – and on how much

animal manure may be applied on a given field – *per year*. Such annual targets prevent the use of "safety valve" mechanisms to limit compliance-cost uncertainly – even if such mechanisms could have been introduced in ways that probably would have had few negative repercussions on the environment, as long as one made sure that average waste amounts/run-off were kept within specified limits on average *over a certain multi-year time-period*.

Because the use of "safety valves" potentially can enhance economic efficiency – by reducing compliance-cost uncertainty – the use of any annual target that is not required due to the nature of the underlying environmental problem will lower the economic efficiency of the environmental policy in question.

A policy lesson that can be drawn from this discussion is that one should consider closely whether annual targets *are* indeed required to address a given environmental problem in an adequate way – or if averages over limited time-periods would be more appropriate. The latter approach could allow more wide-spread use of "safety valve" mechanisms to limit compliance-cost uncertainty – which might be important in order to enhance economic efficiency and/or achieve "stakeholder buy-in" to the policy in question.

7.5. Limiting administrative costs

The potential for reducing total administrative costs through the use of instrument mixes has also been suggested. Obviously, it is not *generally* the case that applying more instruments will reduce total administrative costs, but there can be *some* situations where this occurs.

7.5.1. Cases where easily-monitored instruments are added or kept in place

One possibility is that the costs of putting in place, monitoring and/or enforcing one instrument (which presumably can have a major environmental impact) can be lowered if another instrument (which presumably could not "do the job alone") was jointly applied – especially if the latter is more easily controllable than the former. Several examples of such motivations for instrument combinations have been found in the case studies.

For instance, in the Netherlands, the original intention was to abolish the *Manure Quota Trading Scheme* – which set upper limits on the number of animals per area unit – once MINAS was in place.[28] An instrument that placed a ("correct") price on nutrient losses should in principle also provide ("correct") incentives to control the generation of manure.

However, Dutch authorities later found it necessary to continue to apply the *Manure Quota Trading Scheme*, as it was a relatively *easily controllable instrument* that placed a limit on total manure generation. Without such a limitation, the size of the "goodwill fees" livestock farmers would have to pay to farmers of arable land could have increased significantly, which could have triggered even more fraud in the MINAS system than what actually took place.[29] It is easier to count the number of animals on a given farm than to verify all aspects of a nutrient balance. A similar argument is now used by Dutch authorities for keeping in place specific quotas for pig and poultry production – in addition to the EU milk quotas, which affect cattle numbers – alongside the new application standards that replaced MINAS from 2006.

Another example is that each pesticide product sold in Denmark has to carry a label indicating the maximum legal price of the product – which is used as the basis for calculating the pesticides tax. This labelling system is relatively easy to monitor, and greatly facilitates the administrative task for the tax authorities – but is quite cumbersome for the producers and importers.

7.5.2. Cases where one instrument allows the use of a "simple" version of another instrument

A somewhat different way instrument mixes can lower total administrative costs is also found in Denmark. The relatively strict authorisation system for pesticides helps keep the products that potentially could be most harmful to human health and the environment completely out of the market.[30] Given this fact, a relatively "crude" tax on pesticides could be justified (and implemented), with the tax rates related to the price of each product (within different product categories). A more "sophisticated" tax, *e.g.* based on the recommended area doses for each product, would at the outset have had a lower probability of being implemented – especially as the Danish Constitution requires all tax rates to be specified explicitly in the tax law (*i.e.* not "only" in administrative rules and regulations).

A lesson that can be drawn for this section is that policy-makers should consider any potential for reducing total administrative costs by adding an additional instrument to the mix.

7.6. Mutually supporting instruments

Still another theoretical argument for using several instruments, rather than only one, is if it can be documented that the application of an additional instrument would enhance the environmental effectiveness or economic efficiency of the first instrument. This possibility is already alluded to in several of the sections above.

For example, information-based instruments can complement an environmentally related tax. A label would help increase consumers' awareness, and hence increase the price-elasticity of the relevant demand, making a tax more environmentally effective and economically efficient. Likewise, the tax could make consumers pay more attention to a logo or labelling scheme – thereby increasing the effectiveness of these instruments.

A variable waste collection charge and municipal collection programmes for recyclables could – in principle – also be mutually reinforcing: recycling programmes should reduce incentives for households to dispose of waste illegally in response to waste collection charges (hence improving the environmental performance of charges), and the charges should stimulate higher participation in recycling programmes.[31] However, while separate collection of recyclables now is common in both countries examined in this project, only some 30% of the municipalities in the Netherlands apply variable waste collection charges – and *none* in the United Kingdom do so at present.

Clearly, policy-makers ought to seize any opportunities to apply mutually enforcing instruments – as long as the benefits of doing so exceed the related costs.

7.7. Instruments added to address non-environmental policy concerns

Environmental concerns are obviously not the only concerns that are – or should be – given political attention in OECD member countries, and a number of other priorities are taken into account when environmental policy instrument mixes are designed. Sometimes non-environmental concerns lead to modifications in a "strictly environmental" instrument – like exemptions or rate reductions for certain industrial sectors within an environmentally related tax. On other occasions, a given instrument is given a dual purpose – like a subsidy scheme to increase the energy efficiency of houses of people in "fuel poverty". In still other cases, a "non-environmental" instrument is implemented at the same time as an "environmental" instrument is introduced – as when an increase in social security payments is announced in parallel with an increase in taxes on energy.

7.7.1. Instruments added to address income distribution concerns

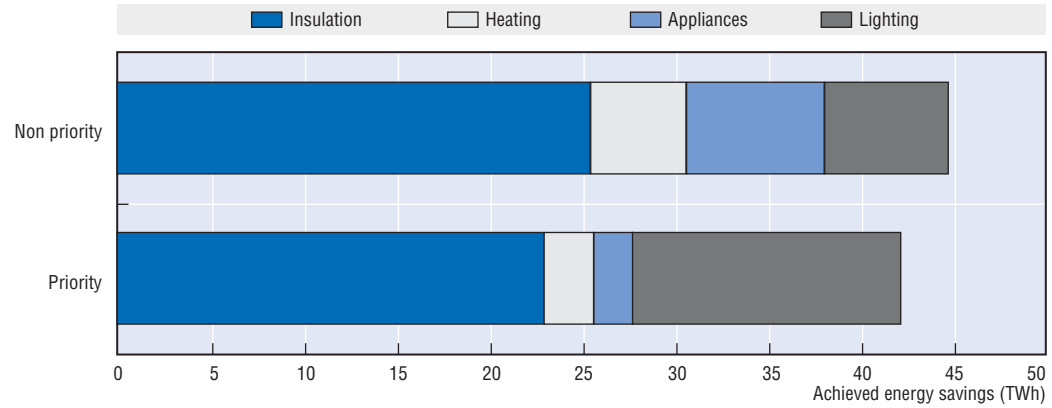
Among the environmental issues studied in this project, income distribution concerns are probably of most relevance for instrument mixes addressing residential energy efficiency.[32] The discussion below focuses in particular on the instrument mix applied in this area in the United Kingdom.

"Fuel poverty" is an issue that for many years has been at the forefront in debates on residential energy efficiency in the United Kingdom.[33] A first comment here is that the definition of "fuel poverty" focuses on variations in energy consumption across taxable *income* groups. It is, however, probable that *some* of the people that have high energy consumption relative to their incomes are relatively rich persons with a low taxable income.[34] It is also probable that *a part of* the high energy expenditures is explained by such things as wasteful "heating" practices (leaving a heater on below an open window in winter-time, etc.).

To improve residential energy efficiency, and to simultaneously address the fuel poverty issue, two important subsidy schemes have been introduced in the United Kingdom – the *Energy Efficiency Commitments* (EEC) – where the subsidies are somewhat indirect, in that it is energy suppliers who are required to invest in energy efficiency improvements – and the *Warm Front*. The *Energy Efficiency Commitments* programme is the main mechanism applied to increase energy efficiency of existing homes, and at least 50% of the energy savings obtained must be focussed on a priority group of low-income consumers.

OFGEM (2005) gives a review of the performance of the *Energy Efficiency Commitments* for the 2002-05, when the overall target was 65 TWh in energy efficiency improvements. First, the overall target was in general met (it was even *significantly over-fulfilled*). This should, however, be seen in the context of an early announcement by the Government that the scheme would be extended – with stricter targets – beyond 2005, but with a possibility for energy suppliers to carry-forward any over-fulfilment to the next period. Second, the requirement that 50% of the savings should focus on the "priority group" was by-and-large met. One can, however, note a difference in *how* the obligations *vis-à-vis* the priority and non-priority groups were met, *cf.* Figure 7.1. The free distribution of low-energy light bulbs played a much larger role in the fulfilment concerning the "priority group" than for other households.[35] There is little information available on the extent to which these bulbs are actually being used.

Figure 7.1. **Energy savings triggered by the EEC, by type of measure and priority category**



Source: OFGEM (2005).

As a consequence of the carry-forward option, suppliers had *already fulfilled* 25% of their (stricter) obligations for 2005-08 at the outset of that period.[36] Adaptations made by suppliers to the changes in energy-saving assumptions made between the two time periods will, however, tend to lower the environmental effectiveness of the scheme.

NAO (2004) and CPA (2005) have also investigated the *Energy Efficiency Commitments*.[37] They *inter alia* point to the possibility that some of the assumed energy-savings credited to the programme might not be realised in practice. OFGEM was criticised for focusing more on *outputs* (number of insulations made, number of light-bulbs distributed, etc.) than on *outcomes* (realised improvements in energy efficiency). One possibility alluded to was that the households might take out more of the benefits of *e.g.* an improvement in insulation in the form of higher room temperature, rather than through a reduction in energy use.

The significant costs of this subsidy scheme have also been pointed out – in spite of lacking cost data from the energy suppliers concerned. It is estimated that commitments in the period 2002-05 added about GBP 8 (almost EUR 12) per year to the energy bill of the average household. With the stricter targets for the period 2005-08, the annual increase in energy costs for the average household could be about GBP 18. Taking into account that the measures undertaken under the commitments only to a limited extent target the fuel poor, National Energy Action refers to the "regressive nature" of the *Energy Efficiency Commitments* in CPA (2004).

Under *Warm Front*, grants to improve energy efficiency are provided to vulnerable people in owner-occupied homes or in private rented accommodation. Every year since 2000, about GBP 150 million has been spent under this scheme. According to NAO (2003), only 77% of the amount was spent on grants, while 23% was used for administration of the scheme in 2001-02. Adding to this problem, around 60-70% of those receiving *Warm Front* grants may not be "fuel poor" – whereas about a third of those that *are* in fuel poverty do not qualify according to the eligibility criteria set for the scheme.[38]

Still another problem was the fact that the formal target set for the scheme managers focused on the *number of households being helped* – with little regard to the magnitude of help each household actually received. This issue was described as follows in CPA (2004):

> "Warm Front's contribution towards the aim of reducing fuel poverty has been monitored through a Public Service Agreement (PSA) target 'to reduce fuel poverty among vulnerable households by improving the energy efficiency of 600 000 homes between 2001 and 2004'. Measurement towards the PSA target has been calculated by counting the number of households in a year receiving grants whether assistance comprised, say, provision of two energy efficient light bulbs or a new central heating system or insulation measures. The current target does not therefore reflect the impact which Warm Front has made on fuel poverty, fuel bills or the energy efficiency of homes assisted. The Department agreed that the current PSA target was inappropriate and planned to discuss a revised target with the Treasury. The target should address fuel poverty more directly, and also incentivise Scheme Managers to achieve success."

In spite of the problems that have been pointed out as regards the *Warm Front* scheme, in the pre-budget 2006[39] it was stated that:

> "The Warm Front programme aims to help low income households, including the poorest pensioners, heat their homes to an adequate standard of warmth by providing heating, energy efficiency measures, and benefit entitlement checks. Through the Energy Efficiency Commitment, energy suppliers also play an active role in helping

> keep the homes of the fuel poor warm, by offering a range of energy efficiency measures, like insulation, free of charge. They also offer discounted energy efficiency measures to all other households. Last year, the Chancellor announced a further GBP 300 million to expand the Warm Front programme to enable pensioners on Pension Credit to have central heating systems installed free of charge and to provide GBP 300 discounts for central heating systems for all pensioners without it. Efficient and effective co-ordination and delivery of these policies will be important if we are to achieve our fuel poverty goals. Evaluation of Warm Zone pilots earlier this year highlighted their potential role in facilitating this. To build on this work, the Government today *announces new investment of GBP 7.5 million to improve the effectiveness of Warm Front and Energy Efficiency Commitment*. This will fund projects aimed at using an area-based approach to identify households and provide the right co-ordinated set of advice and measures to them." (*Emphasis in the original*.)

In conclusion, the case study on residential energy efficiency in the United Kingdom has shown that it is very difficult to design an instrument mix that *effectively* addresses the *environmental* externality of concern – that being domestic fuel use in general and/or carbon emissions from the sector.[40] To begin with, the "deadweight costs" of several of the schemes applied seem considerable. And the costs to households of the (*sometimes* rather inefficient)[41] instruments applied will materialise in the end – even if one tries to "hide them under the carpet" (by placing obligations on the energy-suppliers, that not necessarily are the best suited to address neither the environmental nor the income-distribution problem).[42]

The case study also illustrates the real-life problems policy-makers face when trying to address several different policy targets with any given instrument, such as the *Energy Efficiency Commitments* and *Warm Front*, which in each case tries to address *both* fuel poverty *and* household energy efficiency at the same time. From an energy efficiency/climate change point of view, it could make sense to focus only on those improvement measures that gave *large reductions in (expected) energy use* per GBP spent. Isolating cavity walls can be relatively cost-effective, whereas isolating houses with solid walls would be much more costly per unit of energy saved. However, a person living in a house with solid walls would be much more likely to be in "fuel poverty" than a person living in a house with cavity walls. Hence, focusing only on cavity wall insulation would jeopardise the desired impact on income distribution.

One can also ask why the political focus has been placed on *fuel* poverty as such. Is not the main problem *poverty* in itself – in certain parts of the population? Would it not be better to focus some policy instruments – especially the social security system and the income tax system – on reducing poverty *per se*, and direct the environmental policy instruments more exclusively towards the environmental externalities, in an as cost-effective manner as possible?

Another example of instruments added to an instrument mix to address social concerns is the restrictions on trades between animal categories in the Dutch manure trading scheme. The motivation for these restrictions seems to have been to protect the income position of the Dutch pig farmers – as a kg of phosphorous released has the same environmental impact, regardless of whether it stems from the breading of cows or pigs. The social problem being addressed was apparently of relatively modest scope. There were 6 403 (full time) pig farmers in the Netherlands in 1997. A quota buy-out in the early 1990s

for pig production *could* – according to Wossink (2004) – have prevented a lot of environmental damage and human grief – if there had been sufficient political will to limit the very intensive pig and poultry production.

7.7.2. Sectoral competitiveness concerns – stakeholder buy-in

In part because two of the five environmental issues examined in this project address household behaviour (household waste and residential energy efficiency), the case studies have not highlighted many examples where additional instruments have been *included* in the mix in order to address sectoral competitiveness concerns.[43] There are apparently more examples of the opposite case, where sectors have been *exempted* from regulation, or from inclusion in a particular scheme – among other reasons, in order to protect sectoral competitiveness. For example, agriculture is exempted from the *Clean Water Act* in the United States, the mercury trading scheme in the United States only covers power plants (and *e.g.not* manufacturing industries) and the tax on measured emissions to air in Norway only applies to waste incinerators (*not* manufacturing). Power plants (in the United States) and waste incinerators face less international competition than many manufacturing sectors.

Nevertheless, some examples from the case studies of instruments that have been added in order to limit impacts on sectoral competitiveness, and thereby to secure "stakeholder buy-in", can be mentioned. Among the many instruments applied in the Netherlands to address nutrients run-off from agriculture was the "Social Economic Plan for Animal Husbandry" and a "Quota Buy-out for the Termination of Livestock Farms".[44] Whether the Danish nitrogen quota buy-out scheme was meant to improve the competitiveness of the farmers involved (which it probably did), or to improve the environment (which it hardly did), is, however, somewhat unclear.

In Sweden, the charge on NO_x emissions from energy-generating plants has been combined with a refund mechanism that returns almost all the revenues back to the affected firms as a group. The motivation for including such a refund mechanism was indeed to address competitiveness concerns – both *internationally* and *vis-à-vis* other (smaller) *domestic* plants that (due to the high costs of the equipment needed to measure the emissions) do not have to pay this charge. The refund mechanism also made it possible to set rather high rates in the charge: SEK 40 per kg NO_x, which *e.g.* is significantly higher than the value of externalities related to NO_x emissions estimated by ECON (2001).

In the Netherlands, several instruments are being used to protect the competitiveness position of municipal waste incinerators:

- *Domestically*, the very high rates in the landfill tax and the (incomplete) ban on landfilling protect incinerators *vis-à-vis* (the otherwise less costly) landfill operators.
- *Internationally*, the ban on exports and imports of waste destined for disposal protect the (relatively high-cost) Dutch incinerators from competition from incinerators in other countries – where some of the emission requirements are less strict.

Measures to protect the competitiveness position of some sectors can always reduce the economic efficiency at which a given environmental goal is achieved. A statement in the National Waste Management Plan (2002-06) could, however, be particularly detrimental to economic efficiency:

> "should it prove however during the period of the National Waste Management Plan (2002-06) that the landfill rates are nevertheless too low for managing the waste in the desired way, the need to raise the tax further will be reviewed".

Such a "protection promise" could significantly weaken incinerator operators' incentives to operate the plants as effectively as possible – perhaps prolonging the "need" for international protection.

A policy lesson that can be drawn here is the importance of making sure that all polluters have a (reasonably "correct" – in line with the magnitude of the externality involved) *incentive at the margin to abate emissions*, and an incentive to conduct their operations as efficiently as possible. When certain polluters are exempted from regulation altogether, (obviously) no marginal abatement incentive will be provided.

7.8. Mixes where *lacking instruments* hamper effectiveness or efficiency

The case studies revealed a number of areas where the environmental effectiveness of an instrument mix is jeopardised because some potential instruments are not applied – or only very partially applied. In other cases, significantly higher environmental effectiveness would seem within reach, if economic incentives to reduce emissions had been given to important categories of polluters. To the extent that certain environmental targets are seen as "given", reduced environmental effectiveness of the instruments that are applied would also tend to lower the economic efficiency of the instrument mix as a whole.

7.8.1. Completely lacking or very partial regulation

One example where the lack – or low-level – of regulation seems to be placing significant limits on the environmental effectiveness of other instruments that *are* being applied is the relatively few restrictions which exists concerning conversion of agricultural land into housing projects in the Chesapeake Bay area of the United States. This situation is creating a significant risk that such land-conversions could take place, if environmental restrictions (and other factors) were to significantly reduce farmers' incomes. In this connection, it is important that model simulations of the Chesapeake Bay Program indicate that converting a field of land from agricultural production to new housing areas can lead to an *increase* in total nutrients run-off to the Bay, when the impacts of new septic systems (which are not necessarily required to have nutrients treatment), more impervious surfaces, increased road traffic, etc., are taken into account. In the absence of stricter land-use management tools, and with limited regulations placed on housing developers – it can be important to avoid stimulating further conversion projects, by not placing a significant net burden on farmers.[45]

Another example, also from the Chesapeake Bay area, concerns the regulation of "Confined Animal Feeding Operations" (CAFOs) and the subsidies provided for transport of poultry litter in Virginia. Virginia's Department of Environmental Quality (DEQ) is responsible for tracking and accounting for poultry litter that is being transferred off poultry operations. While DEQ collects such information, there is no state-wide assessment of the places to which the litter is being transferred – and the receiving farms are *not* obliged to have a Nutrients Management Plan in place. This seems to seriously jeopardise the environmental effectiveness of both the regulation of the CAFOs, and the subsidies being provided to have the litter transported.[46]

NAO (2006) points to a detail in the Waste and Emissions Trading Act 2003 that could make the *Landfill Allowance Trading Scheme* (LATS) in England "lose credibility due to the late or inaccurate notification of validated results to the authorities": Responsibilities related to waste management in England are shared between "waste collection authorities" and "waste disposal authorities" – where the latter often cover a larger geographical area than the former. The waste collection authorities are "not tied into the scheme by regulation",

even though they are encouraged to work with waste disposal authorities to support landfill diversion and provide waste statistics.

As NAO (2006) makes clear, the environmental effectiveness of the LATS scheme will depend on the authorities' rigour in imposing penalties and other sanctions in cases of non-compliance. In this connection, NAO found that the data on amounts of waste sent for treatment and disposal in the past was neither complete nor timely – in part, because waste collection authorities were not obliged to submit the relevant information. This problem would seem likely to create additional uncertainty, and hence higher transaction costs, in the market for landfill allowances. Imposing an additional reporting obligation on the waste collection authorities could probably reduce this problem.

7.8.2. Lacking economic abatement incentives for important polluters

The cases studied in this project have provided numerous examples where the environmental effectiveness of the instrument mix in use could have been enhanced if (additional) economic incentives for abatement had been provided to important polluters.

Whereas about 30% of municipalities in the Netherlands apply some form of variable waste collection charges for household waste, local authorities in the United Kingdom are legally prevented from applying such charges. In consequence, households in United Kingdom are *not* getting any direct incentive from the Landfill Tax to reduce the amounts of waste they generate, and to increase their recycling activity. Local authorities that collect and dispose of household waste do, however, get an incentive from the tax to reduce the waste amounts – *e.g.* by putting in place better separate collection systems for recyclable wastes, and by encouraging home composting.

The importance of direct waste limitation incentives from waste collection charges for households is somewhat disputed. Fundamentally, it boils down to a question about the (short and long-term) price-elasticities of waste generation in households – and the price elasticities of illegal waste dumping. Fullerton (2005) provides estimates of impacts of variable charges on the amount of waste generated that generally are quite modest, while Dijkgraaf and Gradus (2004) provide significantly higher estimates. They analysed various waste collection payment systems in the Netherlands and compared the price elasticities of different forms of variable collection charges.[47, 48]

In any case, it is difficult to understand why central authorities should *deny* local authorities an *option* to introduce waste collection charges that vary with the collected amounts. Part of the reason for the existing ban on such charges in the United Kingdom is probably a fear that giving local authorities the option to introduce such charges could be attacked as introduction of "new taxes in disguise". There is also an issue about which modifications – if any – should be made to the current Council Tax, if variable waste collection charges were introduced.[49]

Still another issue that has been raised is the possible negative impact of variable charges on the disposable income of low-income households. Ekins and Dresner (2004) studied this issue in detail and found that it would be possible to compensate low-income households if charging per unit was introduced, even if there should be no reductions in the amounts of waste generated (a somewhat unrealistic assumption).

Other important examples of missing economic incentives were found in the case studies on residential energy efficiency (Canada and United Kingdom). Whereas *e.g.* a large number of regulatory instruments, subsidy schemes and information tools are in use in

both countries, neither internalises the environmental externalities related to energy use in the energy prices faced by households. In the United Kingdom, where industry faces a Climate Change Levy that at least partly addresses the negative externalities of energy use, households' energy use is even indirectly subsidised, because a lower-than-normal VAT rate applies to *domestic energy use*.[50]

One reason given in the United Kingdom for exempting the residential sector from taxes on energy is the fear of aggravating the situation for households in "fuel poverty". Ekins and Dresner (2004) provide some support for such a fear, in an *ex ante* analysis of possibilities to compensate low-income households for new environmentally related taxes or charges *inter alia* in the areas of domestic energy use:

> "In general, it is possible to solve the regressivity problem sometimes associated with environmental taxes and charges through either tariff/charging design or a targeted compensation scheme.
>
> However, the consumption of key environmental resources tends to be widely distributed about the mean within a given income group. This means that, under any practicable compensation system (and assuming no change in household behaviour), some low-income households will end up as net losers from any charging-plus-compensation scheme, even when most low-income households end up as significant gainers."

The problem is, according to their study, in particular related to domestic energy use:

> "…, the enormously skewed distribution of energy consumption within the income deciles means that the average result conceals great differences in net gains and losses within each decile. In fact, none of the [thirteen] investigated compensation packages manages to reduce the proportion of losing Decile 1 households much below 20 per cent. The conclusion is that, although redistributing the revenues from a carbon tax through means-tested benefits would certainly be progressive overall, and would bring some households out of fuel poverty, no way of effecting such a redistribution was found that would not also worsen fuel poverty for those who are already most badly affected by it. This makes introducing a carbon tax on household energy use politically problematic at best, and probably politically infeasible."

However, as discussed earlier, the definition of "fuel poverty" focuses on the variations in energy consumption across taxable *income* groups. It is probable that *some* of the people that have high energy consumption relative to their income are relatively rich, with low taxable incomes. It is also probable that *part of* the high energy expenditures can be explained by *e.g.* wasteful "heating" practices. It would *a priory* seem possible to design a compensation scheme directly targeted at ("needy") persons with very high energy expenses compared to their income, if a tax on domestic energy use were to be introduced. Hence, the conclusion of Ekins and Dresner (2004) on this issue may be too pessimistic.

In the case of non-point sources of water pollution, Danish and Dutch farmers *do* get some economic incentives to limit nutrients run-off via penalties that are part of their respective quota systems. The farmers in the State of Virginia in the United States will also get an incentive to reduce their emissions through the new system of general permits for nutrient loads that is being introduced. Any *new point sources*, and any *expansions* in existing point sources will have to buy off-sets for the nutrient loads they will create, and farmers will be able to sell such off-sets – if they implement control measures that go beyond a defined baseline.[51]

However, unlike Virginia, the State of Maryland has not introduced a point to non-point source nutrient trading system – hence; Maryland farmers do not have a clear economic incentive to reduce their emissions.[52] Similarly, farmers in United Kingdom have few economic incentives to perform better than the existing legislation requires. Special subsidy schemes for organic farming do provide *some* such incentives – but require a major change in the whole operation of most of the farms in order to become relevant. The private costs related to purchases of nutrients and pesticides also give farmers an incentive to avoid using "too much" of these products – but the calculation of what represents "too much" could be tilted more in favour of the environment by introducing economic instruments that increase the price to reflect the negative externalities these products cause.

A policy lesson that can be drawn from this section is that it is necessary to have a *comprehensive view* of the instruments required in order for an instrument mix to be environmentally effective. If certain required elements are missing, the environmental effectiveness of the whole policy can be in danger. Lacking economic incentives for important actors can also hamper both the environmental effectiveness and the economic efficiency of the mix.

7.8.3. Lacking enforcement mechanisms

The environmental effectiveness of several instruments and instrument mixes sometimes also suffer from the lack of enforcement mechanisms.

For example, the State of Virginia applies various subsidy programmes to promote the use of Nutrient Management Plans. Among them, the *Agricultural Best Management Practices Cost-share Program* reimburses farmers for funds expended to implement certain best management practices (BMPs), including the development and writing of Nutrient Management Plans.

The "Joint Legislative Audit and Review Commission of the Virginia General Assembly" has prepared an in-depth *Review of Nutrient Management Planning in Virginia*, *cf.* JLARC (2005). This review makes a number of observations:

- "NMPs that are written in Virginia appear to be generally good in quality. … However, adjustments permitted under the current regulatory language raise some concern that nutrient applications can deviate too far from plan recommendations.
- Survey data from farmers and nutrient management planners present a somewhat mixed picture regarding the implementation of NMPs. Sixty per cent of farmers who are required to implement their NMPs indicate that they always implement their plan. On the other hand, only between 8 and 14 per cent of the certified nutrient management planners indicate their view that farmers always implement their plan. Both groups, however, indicate that farm operators are following their required plans at least most of the time.
- Virginia's approach to nutrient management planning and enforcement of the plans relies heavily on the good will and honest reporting of farmers. The State's inspection and enforcement activities for required NMPs have been weak. Inspections are announced, and compliance checks mostly involve a review of the records kept by farmers. [Virginia's] Department of Environmental Quality (DEQ) inspectors are rarely present when farmers are in the process of applying nutrients to their crops. Furthermore, four of the five DEQ regional offices visited by JLARC during this review were unable to meet a *Code of Virginia* requirement to perform

> annual inspections of the confined animal and poultry feeding operations. DEQ inspections of these operations have found some compliance problems, such as the over-application of nitrogen and phosphorus. In these cases to date, DEQ staff have focused on compliance assistance rather than enforcement actions. For farmers using required NMPs for the first time, this focus may be appropriate. However, most inspections that DEQ now conducts are repeat inspections, and to this point, DEQ has had weak enforcement. The degree of rigor that is appropriate for the State's inspection and enforcement program for NMPs is a policy choice. However, the State may wish to consider whether DEQ staff should proceed more vigorously in instances where there are serious or repeated violations of NMP conditions."

In other words, JLARC (2005) seriously questions the environmental effectiveness of Virginia's approach to nutrients management – pointing to the reliance on the good will and honest reporting of the farmers. It is interesting to note that whereas compliance with the plans in Denmark *inter alia* can be (partially) verified by independent sales records, etc., authorities in Virginia do not have access to such information.[53] All in all, the enforcement of the plans that farmers in Virginia are *being paid to apply* seems quite lax.

A rather obvious policy lesson that can be drawn here is that the environmental effectiveness of an instrument mix will depend crucially on appropriate monitoring and enforcement mechanisms being in place. If the proper workings of an instrument depend more or less solely on voluntary compliance by the "regulated" parties, there is a serious risk that compliance will be partial – unless those being "regulated" have a clear economic incentive to behave in the desired manner.

7.8.4. Lacking supporting instruments

The environmental effectiveness of one instrument could also sometimes be improved if additional "supporting" instruments were introduced.

JLARC (2005) points to the possibility that the lack of a (subsidised) system for farmers to insure against economic losses stemming from voluntary application of a Nutrients Management Plan hampered the up-take of this instrument:

> "In addition to the techniques described above, there may be other opportunities for the State and agricultural community to address the issue of excess nutrients. NMP crop insurance may offer one solution for farmers who fear that following NMPs may result in costly yield losses. Currently there is a Nutrient BMP Endorsement pilot insurance program available to corn farmers in Pennsylvania, Iowa, and Wisconsin. As part of the program, farmers apply the recommended rates of nitrogen and phosphorus to the insured acreage while applying their usual amounts of fertiliser on test strips. If the farmers believe the yield is lower on the fields with nutrient management, they can request an adjustment. If the difference between the fields and test strips is greater than the deductible, the farmers are paid an indemnity. This type of insurance could reduce the risk that some farmers associate with following their NMP recommendations, resulting in potentially higher implementation and compliance rates.
>
> A similar type of program, called Yield Reserve, has been proposed in Maryland. This program offers financial incentives to farmers for reducing fertiliser applications below recommended rates for field crops such as corn. According to a policy group chaired by the Chesapeake Bay Commission, the program is based on the understanding that crops are relatively inefficient nutrient users at high yield levels,

so substantial reductions in nutrient applications result in limited to no reductions in yield. In addition, according to researchers from the Chesapeake Bay Program, current recommended application rates include an insurance level of 35 per cent more than what crops actually need. In the yield reserve program suggested in Maryland, the farmer would reduce nitrogen application rates by 15 per cent. An incentive would then be used to cover the risk of yield loss."

The proposed Yield Reserve in Maryland is still under consideration. While such insurance schemes can lead to greater take-up of Nutrients Management Plans, by reducing the economic risk to the farmers, they come at a budgetary cost – they can also cause "moral hazard" problems. Once a farmer knows that his income is more or less assured, regardless of how well he cultivates his fields, his incentive to run his farm as economically efficient as possible (within the existing environmental regulation) is weakened.

Hence, policy-makers ought to consider carefully the full range of impacts that such insurance schemes might have.

7.9. Mixes where instrument design hampers effectiveness or efficiency

Small design details can compromise the environmental effectiveness and/or economic efficiency of both individual instruments and the instrument mix overall. This project has found several examples of instruments that functioned less well in practice than intended, due to mistakes in the instrument design.

One example is the Danish scheme where farmers could sell-back a part of their nutrient quotas. It is emphasised again that what the farmers offered to sell was a reduction in their *nitrogen quotas* – not necessarily a reduction in their *actual nitrogen use*. If a farmer did not use a considerable share of his legal quota at the outset, he could sell (some of) the unused part, without any impact either on the actual nitrogen application or on his yields.

For a farmer that (without any restrictions) applies nitrogen in an optimal manner, the "shadow value" per kg nitrogen should equal the price of N in fertiliser. Jacobsen (2004) indicates that this price is around EUR 0.5-0.7 per kg, and he estimates that the shadow value of a kg N *actually applied* in the Danish context, with the 10% reduction in the nitrogen norms described in Section 3.6.1.2, is about EUR 0.8-1.0. This contrasts with an average price of the accepted bids in 2002 of about EUR 3.8 per kg nitrogen *quota* reduction[54] – making the instrument very lucrative for farmers that participated, and very costly for society as a whole.

The instrument also included forestry farms, many of which are not required to produce nitrogen accounts. Hence, there is considerable uncertainty about how large a reduction in actual nitrogen losses the instrument induced. Estimates vary between 4 and 67 tonnes [see Blicher-Mathiesen *et al.* (2003)]. With annual costs related to the scheme of some EUR 1.5 million, this leads to *a cost of between EUR 23 and EUR 380 per kg* of actual reduction in nitrogen leaching!

There were three problems with this instrument. First, the fact that it focussed on reductions in the *quotas* rather than on *actual nitrogen use* opened up opportunities for widespread free-riding – inviting offers from farmers who (for some reason or another) would not have used their full quota anyhow. Second, it did not focus on areas where reductions in nitrogen application would provide the most important environmental benefits. Third, and most importantly, far too costly bids were accepted. There ought to

have been a ceiling for accepting bids that reasonably well reflected the economic value of the environmental damage being caused by an additional kg of nitrogen loss.

In connection with the elaboration of *Action Plan III for the Aquatic Environment* in Denmark, the possibilities for, and impacts of, *replacing* some of the existing regulations on nitrogen use in the agriculture sector *e.g.* with a tax on the nitrogen surplus (calculated for the sector as a whole) were considered. Inputs of nitrogen to the sector (feedstuff, fertilisers, etc.) would have been taxed at the wholesale level, while refunds would be given to those who took nitrogen out of the sector (slaughterhouses, mills, etc.). The taxes and refunds would be passed on backwards and forwards in the supply chain, leaving a tax on the nitrogen surplus as the net result. Several studies found that such a change in approach could lead to increased economic efficiency, while preserving the environmental integrity of the instrument mix [see Larsen (2004), Ministry of Taxation (2003) and Jacobsen *et al.* (2004)].

Such a tax on the nitrogen surplus of the sector could also have allowed a similar tax on the phosphorous surplus of the sector to be implemented – instead of the rather narrow tax-base that was in fact implemented, as described in Section 3.6.2.2. A key link between the two tax-base choices is that the administrative costs related to the necessary reporting and enforcement mechanisms would have been spread over two rather similar taxes – the expected revenues from a nitrogen surplus tax would also be much larger than the expected revenues from a phosphorous surplus tax.

In spite of the economic efficiency arguments in favour of letting a tax on the nitrogen surplus of the sector replace current regulations, such a change in policy approach was not made. Part of the reason is that the current regulatory system *has* provided significant environmental improvements – although probably at a higher economic cost than necessary. There seems to be a certain "*status quo* bias" among policy-makers – who hesitate making changes to a "team" that *looks as if* it is "winning".

The *Energy Efficiency Commitments* and the *Warm Front* subsidy scheme in United Kingdom also seem to have been designed in a less-than-optimal manner, *cf.* the criticisms from the National Audit Office referred to in Section 7.7.1 above. For example, the regulator OFGEM was criticised for focusing more on *outputs* than on *outcomes* as regards the *Energy Efficiency Commitments*. Further, as much as 23% of the money spent on *Warm Front* was used for administration in 2001-02. Adding to the problem, around 60-70% of those receiving *Warm Front* grants may not be "fuel poor" [see NAO (2003)].

To avoid Waste Disposal Authorities turning too much towards incineration when stricter limits are placed on the amounts that may be landfilled, *statutory weight-based recycling targets* have been set for local authorities in the United Kingdom. An interesting question is whether the environmental effectiveness has been negatively affected because the targets are *weight-based*. For instance, lightweight packaging constitutes a low proportion of municipal waste material by weight, and has a low bulk density. As the infrastructure required for collecting it is largely the same as for other materials, the cost of collection is higher per tonne of material than for other post-consumer waste. Forum for the Future (2004) states *inter alia*:

> "The recovery of lightweight packaging to meet legislative targets is hampered in the UK by weight based waste collection targets for Local Authorities. Packaging made from materials such as plastic and aluminium make little contribution to meeting these targets. This encourages Local Authorities to go for heavier collectables such as glass, paper, and green waste.

> This bias towards heavy materials has led to questionable outcomes. For example, the UK is currently exporting over a million tonnes of recovered paper as far as China because there is too much of it for UK markets, and importing recovered aluminium to keep the UK aluminium recycling industry going. In 2003, Alcan, the UK market leader, bought 10 000 tonnes of aluminium cans recovered from the UK and had to import 45 000 tonnes from all over the world. Yet three out of four aluminium cans consumed in the UK (about 3.25 billion cans) were sent to landfill or incineration…"

On the other hand, if the environmental impacts of the current waste management operations are believed to be modest (*cf.* the discussion in Section 2.4.1) and if the prime motivation for the recycling targets is to help secure compliance with the EU Landfill Directive, focussing on recycling the waste streams with the lowest (economic) costs can make sense.

The Dutch MINAS system also had a number of practical design problems – which *inter alia* explains the many revisions that were made to the system during the years it was in operation. Mallia and Wright (2004) describe some of these issues; for example concerning the methods used to measure the nutrients content in manure. There is a clear risk that similar sampling problems as those which became an issue for pig farms in MINAS, could also become important in the new system based on application standards – especially as the new system will also have to keep track of any changes in manure stocks from year to year. Sedimentation in the manure contributes to making it very difficult to estimate the correct average nutrient content, especially as regards phosphate. One should also note that the new system will *not* provide any incentives whatsoever for pig and poultry farmers to lower the nutrient contents in the feeds.

In addition, there is a danger that it can prove (even) more difficult to enforce compliance with the new application standards than it was to ensure compliance with the MINAS system. For example, the burden of proof for levying an administrative fine in the Netherlands is stricter than for imposing a levy under MINAS.

Obviously, the "strictness" with which an instrument is applied is a key to its environmental effectiveness. It is clear that a high tax rate – or a strict "cap" in a trading system – would trigger more behavioural change than a low tax rate or an "accommodating" cap would do. It is of interest to note that whereas the *compulsory* nitrogen quotas calculated for each Danish farm is deliberately set 10% *below* what is believed to be agronomically optimal, the Nutrients Management Plans prepared in some States in the Chesapeake Bay watershed allow application of nutrients up to 30% *above* what is agronomically optimal – to hedge against risks of "surprises" concerning weather conditions, local soil conditions, etc.

Many instruments could also be more environmentally effective if they focused on the areas where their environmental impacts would be the greatest. This seems to have been the case with the Danish nutrients quota sell-back scheme. US EPA and USDA (2006) discuss the same issue in relation to the conservation subsides provided by the Natural Resources Conservation Service (NRCS) of the USDA. The report emphasises that NRCS does not "permit the establishment of geographic priorities in its allocation methodologies; instead, it tends to emphasize conservation from the perspective of the individual producer on a discrete piece of land". US EPA and USDA (2006) also state:

> "NRCS officials have explained that shifting funds from one area to another entails major economic and political ramifications. Moreover, they stated that if they allowed geographic considerations to enter into their allocation process, they would be

> inundated with requests for special consideration from many regions. While we found NRCS' program allocation methodologies reasonable, we question how NRCS can remain an effective Chesapeake Bay Program partner if it cannot fund States to support the program's tributary strategies and it will not allocate funds to support the unique geographical needs of the Chesapeake Bay watershed.
>
> We conclude that how NRCS allocates its funds is a difficult issue, and one that should be resolved by high-level co-operation between USDA officials. …
>
> Whatever the difficulties involved, given the Federal Government's decision to identify Chesapeake Bay as a priority for environmental cleanup, some corresponding priority for funding must also be arrived at."

GAO (2006b) comes to a similar conclusion – from a more general perspective – concerning the *Environmental Quality Incentives Program* (EQIP) of NRCS, through which subsidies of about USD 1 billion are distributed each year:

> "NRCS's funding process is not clearly linked to EQIP's purpose of optimizing environmental benefits; as such, NRCS may not be directing EQIP funds to states with the most significant environmental concerns arising from agricultural production."

A lesson for policy-makers that can be drawn here is that it can be of vital importance for environmental effectiveness and economic effectiveness how each instrument in a given instrument mix is designed. As the saying goes, "The Devil is in the detail". One should therefore do careful *ex ante* assessments of all new instruments, and regularly do *ex post* evaluations to verify how a given instrument mix is actually working out in practice.

7.10. Mixes where instruments applied for *other purposes* hamper effectiveness or efficiency

The effectiveness of a given instrument mix will obviously be hampered if subsidies are given to "neighbouring" sectors of the economy, when these sectors are major contributors to the environmental problem in question.

For example, the very significant subsidies provided to agriculture in almost all OECD member countries are prime examples of such possible conflicts, when it comes to various forms of water pollution. Agricultural subsidies tend to increase the size of the areas used for agricultural production and to make production per area unit more intensive – even if some shifts in the targeting of these subsidies have been seen in some countries in recent years.

For example, according to Stolwijk (2005), it was especially the high minimum prices for grain in the EU that made it possible for Dutch livestock farmers to out-compete farmers in other EU member States, by importing alternative feeds (*e.g.* cassava). Proximity to the Rotterdam harbour made this option easier to use for Dutch farmers than for farmers in other EU member States. This in turn was one of the explanations for the severe nutrients-loss problems that gradually developed in the Netherlands.

Reducing or reforming the subsidies in the *Common Agriculture Policy* (CAP) could thus be a low-cost option for society to improve environmental quality. However, some caveats should be kept in mind. For example, to the extent that *e.g.* the Nitrates Directive is a *binding constraint* on farmers' production decisions, it is *not given* that lower subsidies would cause less nutrients to be applied on the fields. Similarly, where dairy farming today is effectively constrained by the system of milk quotas, it is not *sure* that production levels would decrease if agricultural subsidies in general were to be reduced.

Further, new provisions that have been included in the CAP can have some importance for developments in pollution stemming from the sector:

- *de-coupling* (farmers now receive one single farm payment, independent of production levels);
- *cross-compliance* (the single farm payment is subject to compliance with a core set of EU legislation on environment, food quality and animal protection); and
- *modulation* (shifting considerable budgets from traditional market support mechanisms to measures for rural development and agri-environmental schemes).

Farmers now also have to maintain[55] their agricultural land in *good agricultural and environmental condition*. Failure to comply with these preconditions can result in a reduction or exclusion of subsidy payments.

CEU (2003) states explicitly that compliance with the *Nitrates Directive* and the *Directive on the Protection of Groundwater against Pollution Caused by Certain Dangerous Substances* is a requirement for a farmer receiving direct payments as of 2005. From an environmental point of view, one can welcome the fact that subsidies are only given to farmers complying with the existing environmental regulations. The fact that the CAP reform decouples a large part of the (still *very* substantial) subsidies from current production levels should also have beneficial impacts for the environment. However, the agricultural sector is still clearly being treated rather generously compared to other parts of the economy: in (many) other sectors, it is not debated that existing legislation *is* to be complied with. No particular carrots are offered to "promote" this – in line with the non-subsidisation idea behind the OECD's "Polluter-Pays Principle".

In the case of regional air pollution, the favourable energy prices available to many heavy industries in Canada can also, in isolation, be expected to increase the energy demand from the manufacturing sector. This would tend to counteract efforts to limit emissions per energy unit produced at electrical power plants, etc.

A policy lesson stemming from this section is that an in-depth evaluation of the environmental impacts of subsidies given to other sectors ought to be made.[56]

7.11. Mixes where instruments applied to address different environmental issues conflict

In some cases, the instruments that are applied to address one environmental problem can make another environmental problem even more difficult to manage. One example from the case studies concerns the Netherlands, where a focus on the promotion of composting household waste tended to aggravate the country's already serious problems concerning nutrients management in the agriculture sector. The soil resulting from the composting process contained significant amounts of nutrients, which were adding to the problems of manure surplus from the intensive agriculture sectors.

Promoting incineration of waste, rather than landfilling, could also lead to larger emissions to air of mercury.[57]

There is also a conflict among environmental targets that has come to light with the introduction of a new "cap-and-trade" emission permit system for mercury emissions from coal-fired power plants in the United States. The use of mercury-containing lamps has long been promoted as a means to increase energy efficiency. Even if there are some emissions to air of mercury related to the disposal of such lamps,[58] these emissions have up to now been

more than counterbalanced by reduced mercury-emissions from coal-fired power plants, in response to lower electricity-demand for lighting purposes. However, with the introduction of a "cap" on the emissions from the power-plants, this argument is no longer valid. A reduction in electricity demand for lighting purposes would no longer lead to a reduction in the *mercury* emissions from the power-plants. Increased use of mercury-containing lamps would hence increase *total* mercury emissions to air – when taking emissions relating both to electricity generation and to the disposal of the lamps into account.

This does *not* mean, however, that one should avoid using a "cap-and-trade" system to control mercury emissions – or that one (necessarily) should avoid promoting the use of mercury-containing lamps. The use of such lamps would still contribute to a reduction in total electricity demand – which entails a number of other (non-mercury) environmental benefits.

A policy lesson that can be drawn here is that it is necessary to have a method to compare the magnitude of qualitatively different environmental impacts of environmental policies – like CO_2 emissions and mercury emissions. More-or-less formalised cost-benefit analyses allow such comparisons, through an economic valuation of the various environmental impacts – and based on the preferences of the population at large. This approach also allows a direct comparison of the net environmental benefits of a policy with the related costs.[59] OECD (2006e) provides a discussion of recent developments in cost-benefit analyses.

7.12. Mixes where the economic efficiency is hampered by overlapping instruments

Whereas overlap between some types of instruments can be beneficial and/or mutually reinforcing (*e.g.* taxes and labelling schemes and taxes used as "safety valves" in trading schemes), overlap between other types of instruments (*e.g.* taxes and product standards) can both hamper the proper working of the instruments involved or cause redundancies and unnecessary administrative costs.

7.12.1. Cases where an economic instrument is hampered by overlapping regulations/bans

Several fully or partly overlapping instruments are applied to address household waste in the Netherlands. The focus here is specifically on the combination of the tax on landfilling (with a very high tax rate) and the ban on landfilling – which is applied with some exemptions. If a complete ban on landfilling – without any exemptions – had been practised, "overlap" between the two instruments would be of little importance – as the Landfill Tax would then not been levied on any waste. Such a ban would, however, imply that an extremely high value was being placed on the damages caused by landfilling. Such a valuation is not confirmed by careful economic estimates, see Section 7.14. Even the way in which the ban currently is practiced – with a number of exemptions – tends to unnecessarily prevent flexible disposal decisions that the landfill tax could have allowed, especially if the tax rate had been *lowered*, to come closer to the available estimates of the environmental damages that landfilling of waste actually cause.

Another example where the flexibility of an economic instrument was hampered by an overlapping regulation can also be found in the Netherlands. The EU *Nitrates Directive* did not allow the flexibility that MINAS offered. MINAS was an integrated instrument, in that it impacted on (almost) all nutrient loss pathways. In theory, a nutrient balance at farm level – or at the sectoral level – is the best instrument for decreasing total losses of nutrients from agriculture, and hence nutrients pollution.[60]

7.12.2. Cases where two or more economic instruments overlap

Household waste management in the United Kingdom is also clearly a case where several instruments address the same environmental issue. One can ask, for example, whether *e.g.* the administrative costs of putting in place the *Landfill Diversion Targets* and the *Landfill Allowance Trading Scheme* (LATS) could not have been avoided by instead relying on increases in the tax rate for "active" waste in the *Landfill Tax* – and by increasing the expected impacts of the Landfill Tax on household behaviour by allowing separate and variable charging for municipal waste collection.[61]

Part of the justification for using several overlapping instruments is likely that the UK authorities are seeking a *higher degree of certainty* of reaching the EU targets regarding landfill diversion than the *Landfill Tax* alone would bring.[62] The country could face considerable fines if it failed to comply with these targets. Decreasing the amount of biodegradable waste being landfilled by more than 25% in four years will, indeed, be a demanding task – and one can only hope that the environmental (and other) benefits of the policy will justify the costs involved.

Once the *Landfill Diversion Targets* and the *Landfill Allowance Trading Scheme* have been put in place, one can ask what the remaining role for the *Landfill Tax* is. Clearly, for *biodegradable municipal waste*, the binding constraint related to the environmental impacts is now the landfill diversion targets – and to a lesser extent, the statutory recycling targets that have also been introduced for all local authorities. This seems likely to be the case in future years as well, even if the tax rate for "active" waste is to be significantly further increased[63] – since the landfill diversion targets will also become stricter. As far as *biodegradable municipal* waste is concerned, the role of the Landfill Tax at the moment (and in the foreseeable future), seems primarily to be to raise revenues.[64]

However, the *Landfill Tax* is also the main mechanism for encouraging producers of commercial and industrial waste to move away from disposing waste at landfills – and it can, along with the statutory recycling targets, contribute to increased recycling of non-biodegradable recyclables, such as glass and metals. Hence, it would not be an easy option to replace the Landfill Tax *completely* by the Landfill Diversion Targets and the associated trading scheme. One option, however, could be to drop the tax for *municipal* waste, while keeping it in place as regards other types of waste that are sent to landfills. If so, it could be useful to align the tax rates more closely to the value of the estimated negative externalities caused by landfilling, especially those related to "active waste".

7.12.3. Cases where two or more regulatory instruments overlap

In the United Kingdom, there is a clear overlap between the producer responsibility obligations for packaging waste and the statutory recycling targets set for local authorities. By putting in place kerb-side collection systems for, *inter alia*, packaging and similar materials, municipalities do part of the job the producers have been told to do – without being paid for it. From an efficiency point of view, however, it could be a more important question whether the overlap between these two schemes leads to contradictions, confusion among some actors, unnecessary administrative costs, etc.

A policy lesson that emerges here is that one should consider carefully the impacts of applying overlapping instruments. Unless a case can be made that two or more instruments are likely to *mutually reinforce* each other, it could be better to increase the "dosage" of *one* instrument (if that is required to reach a well-funded environmental target), rather than to apply additional instruments that would overlap each other.

7.13. Mixes with large differences in marginal abatement incentives for different sectors

7.13.1. Cases where one sector faces much stricter regulations than other sectors

The case study on non-point sources of water pollution in the Chesapeake Bay area provides one example of differences in the strictness in regulations facing different economic sectors. Based on provisions in the Federal *Clean Water Act*, quite strict pollution abatement requirements are placed *e.g.* on wastewater treatment plants – while the agricultural sector mostly is completely exempted from the provisions of this law.[65] Similarly, at the State level, the regulations facing the agriculture sector are more lenient that those facing other sectors.

New provisions for point- to non-point emission permits trading in some States – where farmers are allowed to sell "offsets" to more strictly regulated point sources if they (the farmers) undertake measures that go beyond a predefined "baseline"*can* reduce the significance of this problem in the future.

Regarding household waste, the *Protocol on Persistent Organic Pollutants* within the framework of the United Nations Economic Commission for Europe (UNECE) *Convention on Long-range Transboundary Air Pollution* sets a legally binding limit value for the emission of dioxins and furans of 0.1 ng TE per m^3 (Toxicity Equivalents), for installations burning more than 3 tonnes of municipal solid waste per hour. A similar *common* upper limit is not set for emissions of dioxins and furans from *e.g.* electricity power generation plants. Limits on emissions of dioxins and furans can, however, be set on a case-by-case basis in environmental permits. If so, the limits set tend to be less strict for power plants than for waste incinerators.

Similarly, the mercury case studies have illustrated that limits on mercury emissions for waste incinerators tend to be stricter than similar emission limits for other point sources.

If differences in emission limits correctly reflect differences in technical abatement possibilities, (meaning that it would be much more costly for some sources to reduce the emissions down to a certain concentration level than for other sources), such an approach would *to a certain extent* mimic the outcome of an economic instrument – in that the *marginal* abatement costs could be more-or-less equalised.

There are, however, important differences:

- An approach based on setting different emission limits for different pollution sources *assumes* that public authorities have correct information about the cost functions of all relevant sources – which in practice is improbable.
- A tax or a trading system would give all the polluters a continuous incentive to reduce their emissions further – where as no such incentive is provided once compliance with an emission limit is achieved.

7.13.2. Cases where taxes/permit systems that apply to one sector do not apply to others

In the United States, a new trading system for mercury emissions is being prepared, but it will *only* cover emissions from coal-fired power plants. Clearly, the economic efficiency of the overall instrument mix addressing air emissions of mercury could have been enhanced if this instrument had been broadened to cover other major point sources as well.[66]

Also in the mercury area, the recently introduced tax on measured or estimated emissions from waste incinerators in Norway – which levies a tax per gram mercury emitted – does not cover other potential point sources of mercury emissions to air. According to sources interviewed during the case study, a broadening of this tax was considered, but the idea was dropped *inter alia* due to fears that it would impact negatively on the international competitiveness of certain industrial sectors.[67] It was also stated out that existing pollution permits for mercury already cover relevant point sources.[68]

Concerning energy efficiency, it has already been commented above that industry in United Kingdom pay a Climate Change Levy. The economic efficiency of climate policies in the country could have been enhanced if this levy was broadened to also include the domestic sector.

As regards regional air pollution, the Swedish charge on NO_x emissions only covers combustion plants larger than a certain limit, *cf.* Section 5.4.2. These plants cause only about 3% of all Swedish NO_x emissions. While it in also here, in principle, could have been an advantage if the charge had covered all relevant sectors, the costs of measuring the NO_x emissions at individual sources makes it difficult to increase the scope of the charge much further.

The SO_2 and NO_x trading systems in the Province of Ontario at the outset only covered the electricity sector. A broadening of the systems to cover several large industrial sectors would enhance economic efficiency, by bringing in plants with a larger variation of marginal abatement costs.

7.14. Mixes where further analyses could be appropriate

7.14.1. Cases where further analyses of the environmental targets could seem warranted

The case studies also illustrated that further analyses of the targets set in some environmental areas could seem warranted, *inter alia* to see whether the scientific and economic underpinnings are valid, and whether the benefits of reaching the targets can be expected to exceed the related costs.

For example, waste policies at both the EU level and within individual member countries focus strongly on the "waste hierarchy", where: *i)* reuse and recycling are preferred to all final disposal options; and *ii)* where incineration is preferable to landfilling. However, several recent studies cast (renewed) doubts on the foundation for preferring incineration over landfilling, when the *total costs* to society are taken into account.

For example, HM Customs and Excise (2004) presents both a low and a high estimate for the environmental externalities relating to landfilling and incineration in the United Kingdom, building both on a scientific study of the health and environment impacts of different waste management options and a survey of recent valuation studies. It was found that "in the central case, the external costs of landfill may be around GBP 10 per tonne of municipal solid waste and the external costs of incineration (with energy recovery) may be around GBP 13 to GBP 14 per tonne of municipal waste".

It was further concluded that "sensitivity analysis around the central case shows that the range of possible externality estimates is large. For example, varying the emissions levels from landfill sites and incinerators, and holding all other factors constant, gives a range of external costs of GBP 5 to GBP 20 (EUR 7.4 to EUR 29.5) per tonne for landfill and GBP 7 to GBP 21 (EUR 10.3 to EUR 30.9) per tonne for incineration".

Dijkgraaf and Vollebergh (2004) and Bartelings *et al.* (2005) compared the private and social costs of landfilling and incineration in the Netherlands, see Table 7.1 below. These estimates concern a *new site*, using *state-of-the-art technologies*, in accordance with current Dutch emission standards, etc.

Table 7.1. **Social costs of incineration and landfilling in the Netherlands**

Euro per tonne waste

		Landfilling	Incineration
1.	**Gross private costs**	40	125
2.	**Private cost savings**		
3.	Energy	–4	–21
4.	Materials	–0	–3
5.	**Net private costs** [1 – (3 + 4)]	36	101
6.	**Environmental impacts**		
7.	Climate change	4.21 *(1.46 – 54.50)*	0.11 *(0.06 – 0.88)*
8.	Other emissions to air	1.22 *(0.58 – 1.85)*	7.22 *(1.50 – 7.22)*
9.	Transport-related impacts	1.25	1.67
10.	Disamenity	3.50 *(3.50 – 3.80)*	9.09 *(9.09 – 9.87)*
11.	Solid waste		0.11 *(0.09 – 5.62)*
12.	Land use	0.00 *(0.00 – 17.88)*	
13.	**Gross environmental costs** (7 + 8 + 9 + 10 + 11 + 12)	10.18 *(6.79 – 79.28)*	18.20 *(12.41 – 25.26)*
14.	**Environmental cost savings related to energy generation and materials recovery**	–1.14 *(–0.85 – –4.46)*	–7.63 *(–6.96 – –12.56)*
15.	**Net environmental costs** (13 – 14)	9.04 *(5.94 – 74.82)*	10.57 *(5.45 – 12.70)*
16.	**Net social costs** (5 + 15)	45.04 *(41.94 – 110.82)*	111.57 *(106.45 – 113.7)*

Source: Based on Bartelings *et al.* (2005).

The gross private costs of building a new incinerator in both studies were found to be much higher per tonne of waste than for building a new landfill. The difference in private costs is reduced when account is taken of the economic value of the energy produced by the different facilities, and of the materials recovered from the ash of the incinerator – but only to a modest extent. Hence, the net private cost for a landfill was found to be EUR 36 per tonne of waste, *versus* EUR 101 per tonne for an incinerator.

The numbers given in Table 7.1 as regards environmental costs are based on the estimates of Bartelings *et al.* (2005). Their "best" estimates are presented in the upper lines, while the ranges between their "low" and "high" estimates are given in the parentheses underneath.

The "best" estimates indicate that the *gross* environmental externalities of incineration are significantly higher than for landfilling – largely due to higher "other emissions to air", and because the disamenities of the incinerators are found to affect a larger number of people.

However, incineration also causes important environmental savings in the energy and in the materials that are recovered in the process. Hence, the *net* environmental costs of

landfilling are only slightly lower than what they are for incineration – EUR 9 *vs.* EUR 10.5 per tonne waste. However, the estimated environmental costs of landfilling are only a small fraction of the current tax rate of the Dutch Landfill Tax.

Adding together the net private costs and the net environmental costs provides the *net social cost* – estimated here to be approximately EUR 45 per tonne for landfilling, and EUR 112 per tonne for incineration.

Estimates of the value of the environmental externalities of both landfilling and incineration presented in the Dutch studies are therefore well within the range presented in HM Customs and Excise (2004).[69] The differences in *environmental costs* of landfilling and incineration are in all cases so small that the ranking of the two options in terms of their *net social costs* is likely to depend mostly on the *private costs* of landfilling *vs.* incineration.

Against this background, further consideration of the foundation of the present strong preference for incineration over landfilling – which has strong bearings on the current policy-choices in both the Netherlands and the United Kingdom – could seem desirable.

OMB (2006) is a draft report to the US Congress on the costs and benefits of Federal regulations. Concerning the *Clean Air Mercury Rule*, the report presents an estimate of annual benefits in the order of USD 2 million, while annual costs are found to be about USD 500 million. It is, however, emphasised that the only benefit that have been monetised in this study is the value of avoided decrements in the IQs of children who had prenatal exposure via maternal fish consumption. Co-benefits in the form of reductions in emissions of particulate matter are not included – nor are any other health impacts or ecosystem benefits.[70] It is, however, not given that the inclusion of any such impacts in the analysis would make the benefits to society as a whole stemming from the rule larger than the costs.[71]

Another point to make about targets for environmental policy in the instrument mixes studied in this project is that they are often of a dichotomous nature – meaning that the focus is on reaching a certain limit or "value" for a specific parameter. For example, the instruments addressing pesticides use in Denmark serve the aim of bringing the concentration of pesticides below 0.1 µg per litre drinking water.[72] Similarly, the EU Nitrates Directive has a target of limiting concentrations of nitrate in groundwater to 50 mg nitrate per litre.[73]

While such targets can be useful for setting long-term directions for policy development, they do not – by themselves – allow much scope for consideration of whether the benefits of further environmental improvements would exceed the costs. If, for example, the concentration everywhere is 49 mg nitrate per litre, a country would be in compliance with the EU Nitrates Directive. If the concentration in *one* reservoir of groundwater is 51 mg nitrate per litre, the country is (*formally*) in *non*-compliance. No comparison is to be made between the *additional* costs and the *additional* benefits of bringing the concentration in "the last" reservoir down from 51 to 49 mg.

The targets set in the *Chesapeake 2000 Agreement* are of a similarly dichotomous nature: The aim is to have the Chesapeake Bay taken off EPA's list of impaired waters in 2010.[74] At some stage, it might become necessary to consider whether the costs to society as a whole of approaching the targets further could exceed the benefits to society of doing so.[75]

One policy lesson that emerges here is therefore that more, and broader, analyses of the benefits and the costs related to environmental policy targets and priorities ought to be made. In this regard, it is important to monetise as many relevant environmental impacts

as possible – in order to include them *quantitatively* in the analyses. It is also important to get a better picture of how the benefits and the costs would vary with different ambition levels and with different instrument choices – to allow the selection of targets where the marginal benefits reasonably well approximate the marginal costs when the most cost-effective instrument or instrument mix is applied.

7.14.2. Cases where further analyses of the impacts of the instruments could seem warranted

Many elements of the impacts of specific instrument mixes could usefully be analysed further – both *ex ante* (before the instruments are applied) and *ex post* (after introduction of the instrument in question). Even if many countries have rules in place requiring *ex ante* assessments of the economic impacts of environmental (and other) policy instruments (as well as of the environmental impacts of non-environmental instruments), and even if some *ex post* evaluations are being made in some countries, much more could usefully be done in this area. The discussion here will only highlight a few relevant points.

A problem found both in the *design* of a number environmental policy instruments, and in *assessments* made of their environmental impacts, is that they focus on "outputs" rather than "outcomes". As discussed earlier, this is the case for the *Warm Front* subsidy scheme in the United Kingdom. As highlighted by CPA (2004), the focus of that Programme is on the *number of households* that has been helped (an "output"), rather than on how much improvement in energy efficiency – or reduction in greenhouse gas emissions – (an "outcome") the scheme has contributed to.

US EPA and USDA (2006) makes similar observations regarding the subsidy programmes run by various USDA agencies, *inter alia* in support of improvements to the water quality in the Chesapeake Bay area. For example, concerning programmes run by the National Resources Conservation Service (NRCS), they conclude:

> "To gauge how well NRCS is achieving its goals, the agency identified performance measures that associate program activities with appropriate units of input; these units measure how many acres or miles have been treated with a given conservation practice rather than how effectively that practice has improved water quality. In other words, these units do not measure outcome, but output. Thus, NRCS can reasonably show the number of conservation practices completed or the number of acres in the Wetland Reserve Program, but it cannot show the effect of the first acre or mile on water quality. Without outcome-based performance measures, NRCS cannot confidently state the degree to which its programs have succeeded.
>
> To correct this shortcoming, NRCS is developing the Conservation Effects Assessment Project (CEAP). The agency asserts that CEAP will provide scientifically credible estimates of the environmental benefits obtained from NRCS conservation programs. Begun in 2003, CEAP is collecting data to determine the best methodology and remains a work in progress…
>
> In May 2006, NRCS published a new strategic plan for 2005-10 entitled, *Productive Lands – Healthy Environment NRCS Strategic Plan*. The new strategic plan has six new goals (…) with outcome-based performance measures. For example, for the Clean and Abundant Water goal, NRCS has a clear outcome-based performance measure that addresses agricultural sediment and nutrient influence on water quality. The performance measure calls for agricultural producers to reduce potential delivery of a

specific number of tons of sediment and nutrients from their operations. Specifically, agricultural producers are to reduce sediment delivery from agricultural operations by 70 million tons (…); reduce delivery of nitrogen from agricultural operations by 375 000 tons (…); and reduce delivery of phosphorus from agricultural operations by 70 000 tons (…). Once these performance measures are fully implemented, they will provide USDA with useful outcome-based data concerning the success of NRCS' programs in the Chesapeake Bay watershed area."

Obtaining outcome-based estimates of the environmental impacts of relevant policy instruments would seem to be a prerequisite for meaningful economic valuation of these impacts – which in turn is necessary for comparing the magnitude of environmental impacts pulling in opposite directions (*e.g.* a decrease in CO_2 emissions *versus* an increase in mercury emissions), and for comparing costs and benefits of a given policy.

7.15. Are the mixes used at present environmentally effective?

In this Section, an attempt is made to compare the environmental effectiveness of the instrument mixes used in the countries that have been subject to the case studies that were done for this report. It is emphasised that *these comparisons are incomplete* – largely due to data availability problems. Many relevant "aspects" of the various environmental issues have thus not been compared. The results presented below should therefore be interpreted as "indicators", not as "statements of facts".

7.15.1. Household waste

Figure 7.2 illustrates generation intensities for household and municipal waste in OECD member countries in 2005. It is interesting to note that – even though the Netherlands had a much more "active" waste policy over many years compared to those in the United Kingdom, total amounts of both household and municipal waste per capita are slightly *higher* in the former country. Hence, in this respect, the Dutch instruments mix does not seem to have been more effective in this respect than its UK counterpart.

However, the instrument mix applied in the Netherlands *has* succeeded in securing a much higher share of recycling and composting of municipal waste than in the United Kingdom (see Figure 2.2). There is broad agreement that this is environmentally beneficial. The Dutch mix has also directed a larger share of the remaining waste towards incineration (*i.e.* not to landfilling), compared to what has been the case in the United Kingdom. *To the extent* incineration in fact is preferable to landfilling from an environmental point of view (*cf.* Section 7.14), the Dutch instrument mix has also been more environmentally effective than the UK mix in this respect.

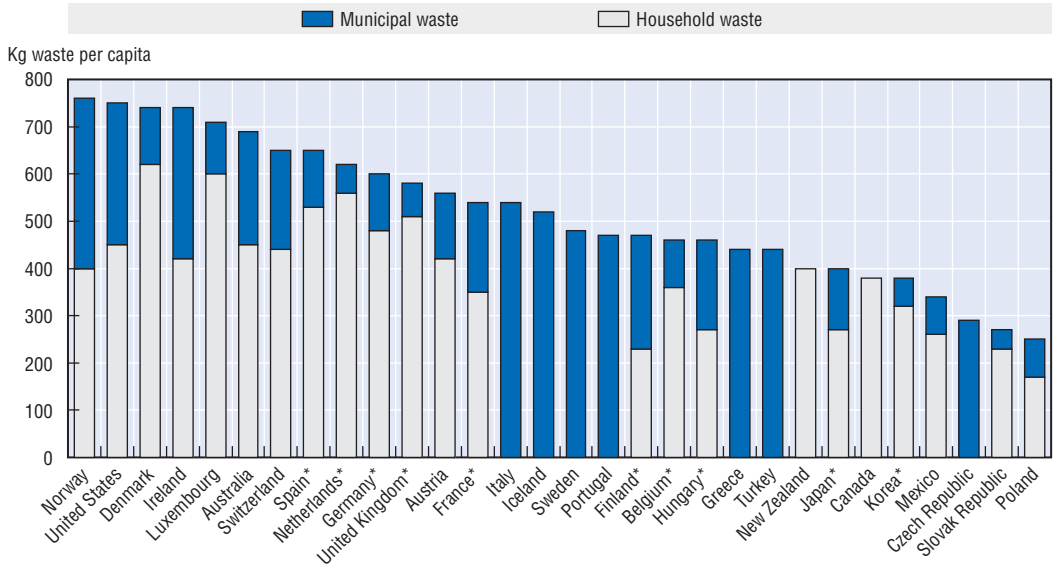
However, Figure 7.3 indicates that the more stringent policies concerning municipal waste introduced in the United Kingdom in recent years have started to have an impact, both on total amounts of municipal waste generated and on how that waste is treated. While there have only been minor changes in the use of incineration, the share of municipal waste in England being recycled or composted increased from 12% in 2000/01 to 27% in 2005/06, leading to a significant decrease in the amount and the share of municipal waste going to landfills.

7.15.2. Non-point sources of water pollution

Figure 7.4 presents estimates of nitrogen balances per area unit for OECD member countries. Measured in *tonnes*, the total OECD nitrogen surplus declined by 3% over the

Figure 7.2. **Generation intensities for household and municipal waste**

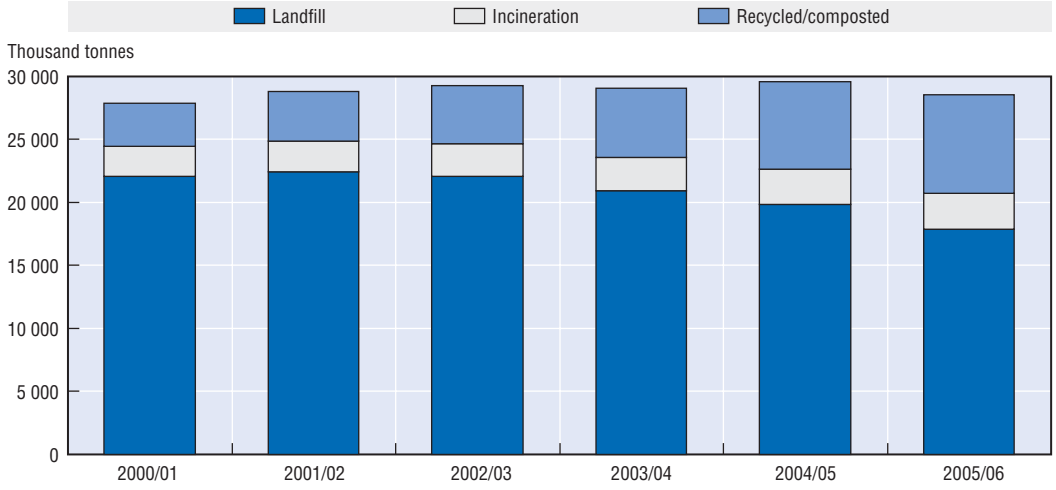
Kg waste per capita, 2005



Source: OECD data. An asterisk behind a country name indicates that the underlying number is provisional, stems from an earlier year or is estimated by the OECD Secretariat.

Figure 7.3. **Municipal waste treatment in England**

Thousand tonnes

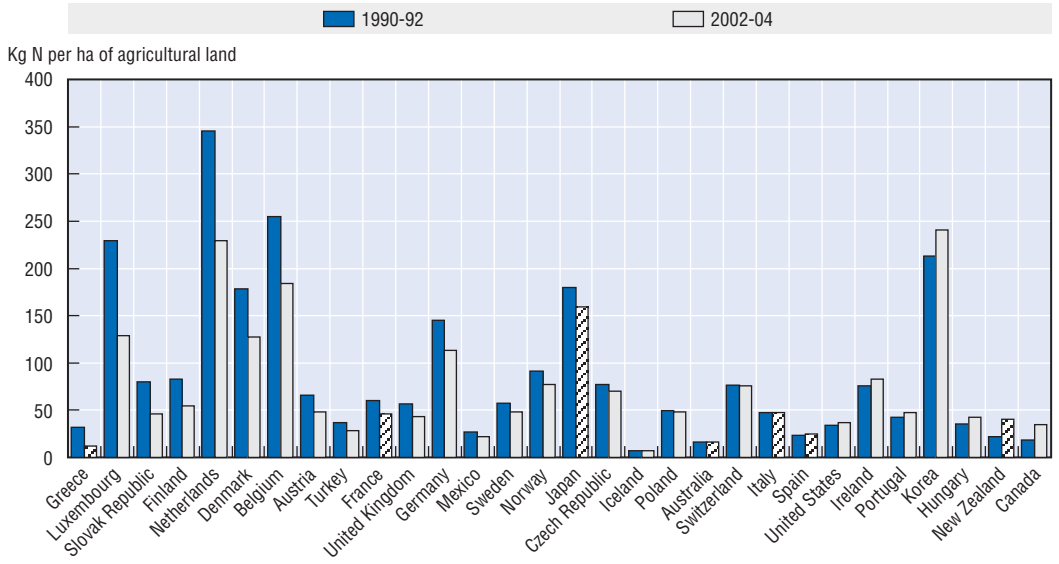


Source: *www.defra.gov.uk/environment/statistics/wastats/archive/mwb200611.xls.*

period 1990-2004. Nitrogen surpluses per ha showed the largest increases in Iceland, Canada, New Zealand, but also rose in the Hungary, Korea, Portugal, Ireland, United States and Spain. Despite the increases in these countries, they had (with the exception of Ireland and – especially – Korea) an intensity of kg nitrogen *per hectare* of agricultural land well below the OECD average in 2002-04. At the same time, a number of countries where nitrogen surpluses have shown large (or, at least, significant) reductions over the past 15 years continue to have high surpluses of nitrogen per hectare of agricultural land (notably the Netherlands, Belgium, Japan, Luxembourg, Denmark and Germany).

Figure 7.4. **Nitrogen balance estimates for OECD member countries**

kg N per ha agricultural land. 1990-92 and 2002-04



The gross nitrogen balance calculates the difference between the nitrogen inputs entering a farming system (*i.e.* mainly livestock manure and fertilisers) and the nitrogen outputs leaving the system (*i.e.* the uptake of nutrients for crop and pasture production).

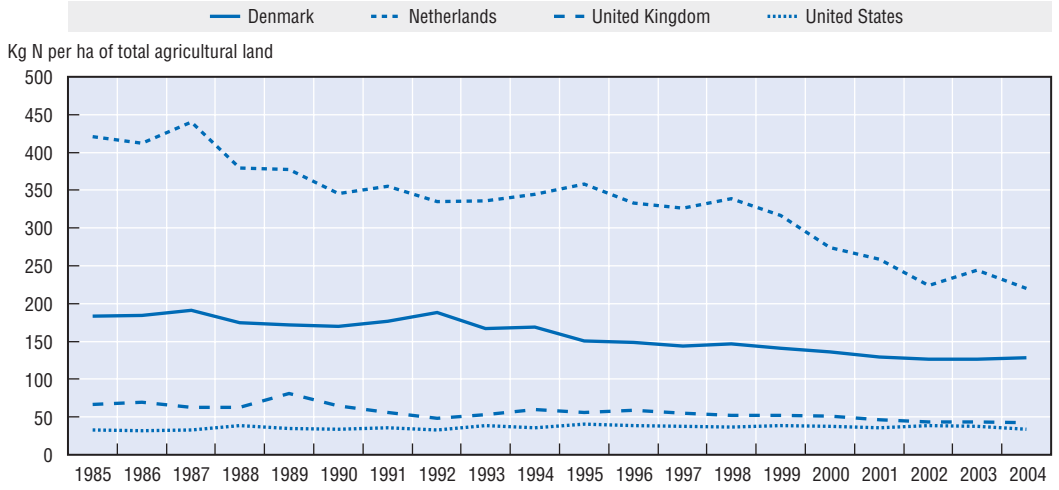
The countries have been sorted according to the percentage change in the nitrogen balance per ha agricultural land. For shaded bars, estimates for 2002 only are used. For Greece, the average for 2002 and 2003 is shown.

Source: Based on OECD (forthcoming).

Focusing on the countries that have been analysed especially in this project, the instrument mixes applied have brought about *significant environmental improvements* in nitrogen surpluses expressed per area unit in Denmark and in the Netherlands, as can be seen from Figure 7.5. After a rapid reduction from the peak in the balance around 1990, environmental improvements in this regard are harder to document in the United Kingdom

Figure 7.5. **Nitrogen balances in Denmark, Netherlands, United Kingdom and the United States**

kg N per ha of total agricultural land



Source: OECD (forthcoming).

– but the starting point was much better than in the former two countries. Hence, the need to apply instruments that significantly influence farmers' behaviour was less acute in the United Kingdom than in the Netherlands and Denmark.

According to the data behind this figure, the situation within the United States *on average* is very similar to the situation in the United Kingdom. However, national averages do not allow any conclusion to be drawn with respect to the environmental effectiveness of the instruments addressing farmers in the Chesapeake Bay area (see Figure 7.9 for more information).

Looking now at final environmental *outcomes*, Figure 7.6 illustrates that the impacts on nitrate concentrations in the upper level of groundwater of the policies implemented in the Netherlands vary significantly. In some regions, the average nitrate concentration *increased*

Figure 7.6. **Nitrate in groundwater at a depth of 5-15 meter in the Netherlands**

Change in average nitrate concentration between 1996 and 2002.



Source: RIVM (2004b).

between 1996 and 2002, according to RIVM (2004b), while in other regions the concentrations *decreased* – in some cases; by more than 5 mg per litre water. Interestingly, places where the concentration has decreased are often located relatively close to places where the concentration has increased.

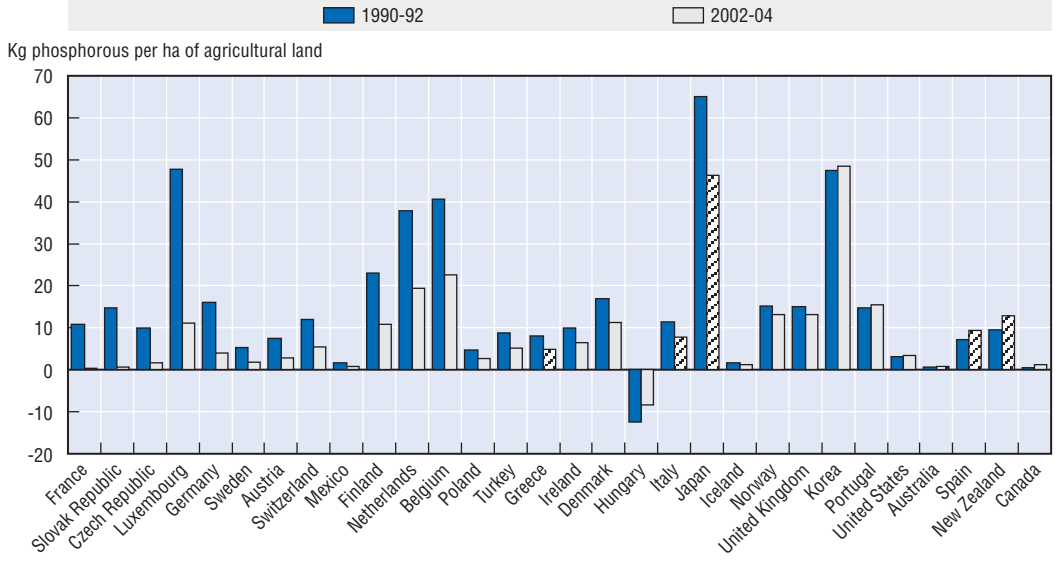
According to GEUS (2004), there are now some signs of a reduction in the concentration of nitrate in the "youngest" parts of the groundwater in Denmark. They report that this reduction perhaps is a result of changes in agricultural practices since the first *Action Plan for the Aquatic Environment* was introduced. However, the concentration of nitrate is also higher in younger groundwater than the target set of 50 mg per litre. GEUS (2004) thus concludes that "the nitrate concentration is heading in the right direction, but… the measures implemented so far [are] not necessarily… enough to reduce the nitrate concentration in the groundwater to a sufficient extent".[76]

Figure 7.7 illustrates estimates of phosphorous balances per area unit in OECD countries in 1990-92 and 2002-04. There was an 18% reduction in the total agricultural phosphorus surplus (measured in *tonnes*) between 1990 and 2004 in the OECD countries. This was a much larger percentage reduction than for the total OECD nitrogen surplus, mainly because of a 9% decrease in phosphate fertiliser use. In about half of all OECD countries, phosphorous surpluses per area unit declined by more than 40% over the period. All of these were European countries, with the exception of Mexico. However, the intensity of phosphorous use per hectare of agricultural land for some of these countries (including the Netherlands) still remains among the highest in the OECD.

Figure 7.8 illustrates that phosphorous surpluses, expressed as *kg per Ha agricultural land*, declined considerably in the Netherlands and Denmark between 1985 and 2004 – in the Dutch case, from a very high starting point. In spite of hardly any instrument being

Figure 7.7. **Phosphorous balance estimates for OECD member countries**

Kg phosphorous per ha agricultural land – 1990-92 and 2002-04

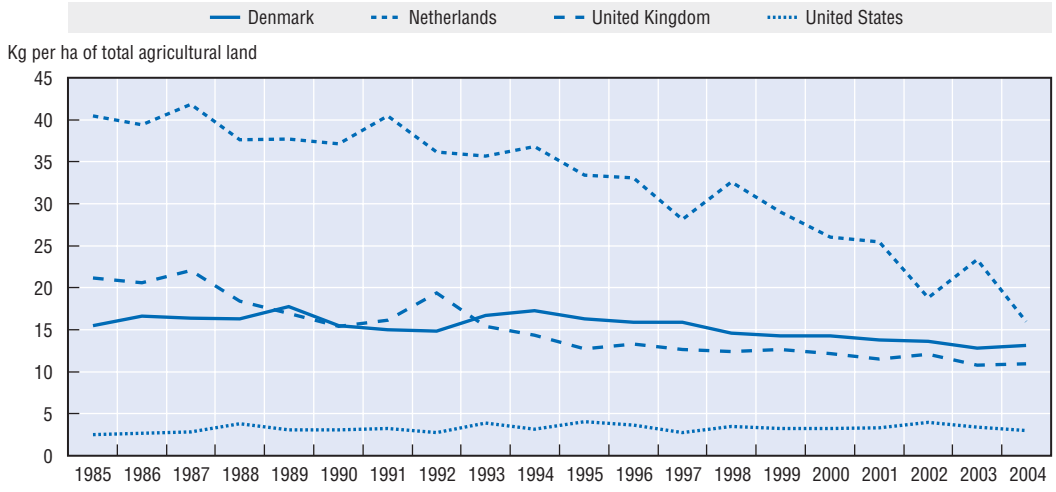


The gross nitrogen balance calculates the difference between the phosphorous inputs entering a farming system (*i.e.* mainly livestock manure and fertilisers) and the phosphorous outputs leaving the system (*i.e.* pasture production). The countries have been sorted according to the percentage change in the phosphorous balance per ha agricultural land. For shaded bars, estimates for 2002 only are used. For Greece, the average for 2002 and 2003 is shown.

Source: Based on OECD (forthcoming).

Figure 7.8. **Phosphorous balances in Denmark, Netherlands, United Kingdom and the United States**

Kg per ha agricultural land



Source: OECD (forthcoming).

applied to address phosphorous specifically in the United Kingdom, there has been some improvement since 1995, albeit not quite as rapid as in the former countries. At a national level, the surplus is relatively low, but slowly *increasing*, in the United States. However, the data presented here again does not allow any conclusions concerning the environmental effectiveness of the instruments specifically applied in the Chesapeake Bay area.

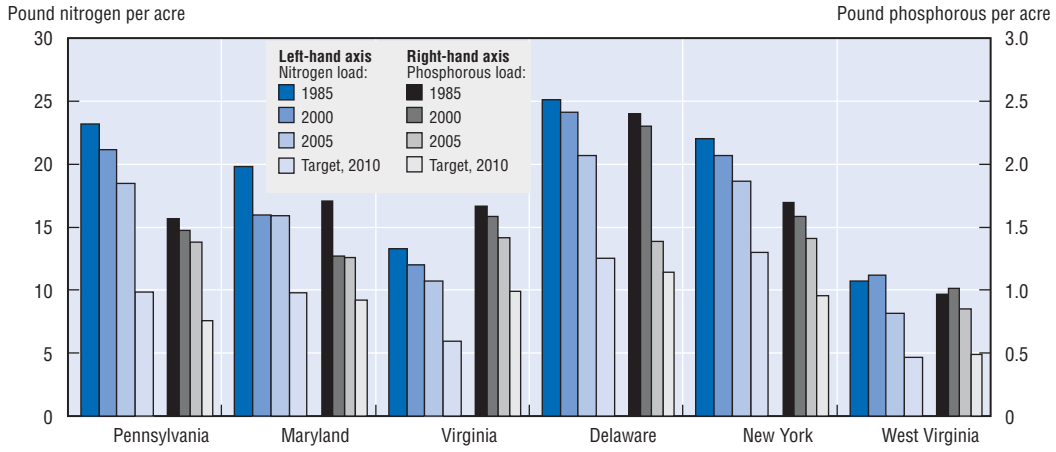
To shed some light on the environmental effectiveness of the instruments applied to limit nutrients run-off in the Chesapeake Bay area, Figure 7.9 presents estimates of the nutrient loads *per acre* to the Bay *from agriculture* at the "edge-of-stream"[77] in the different States, as assessed by the Chesapeake Bay Program Phase 4.3 Watershed Model. Next to the estimates of the nutrient loads in 1985, 2000 and 2005 are presented the targets each State has set for the loads from agriculture in their respective tributary strategies. In general, these targets were set *inter alia* based on expectations of where it would be most cost-effective to achieve the reductions.

It is clear from Figure 7.9 that more needs to be done in *all* of the States, if nutrient loads per acre from agriculture are to be reduced to the targets set in the tributary strategies. Focusing on the two States that have been studied in this project, one can notice that steady progress was made in reducing loads of both nitrogen and phosphorous in Virginia between 1985 and 2005. In Maryland, larger progress than in Virginia was made between 1985 and 2000 for both nutrients – but between 2000 and 2005, improvements were quite modest. Unless more environmentally effective instruments are developed, it is unlikely that any of the States will reach the targets they have set by 2010 – and the EPA has indicated in their Strategic Plan that the Bay Program goals will not be met by that date.

Figure 7.10 illustrates changes in the number of *tonnes of active ingredients* of pesticides used in OECD member countries between 1990-92 and 2001-03. There are very large differences in the developments in different countries over this period. While pesticide use in some countries increased 50% or more, the number of tonnes active ingredients applied was reduced by at least 35% in Denmark, Netherlands and Hungary. On average for all OECD countries, there was a 5% reduction in the amount of active ingredients of pesticides being used – while in the UK, the reduction was 6%.

Figure 7.9. **Estimated nutrient loads from agriculture to the Chesapeake Bay**

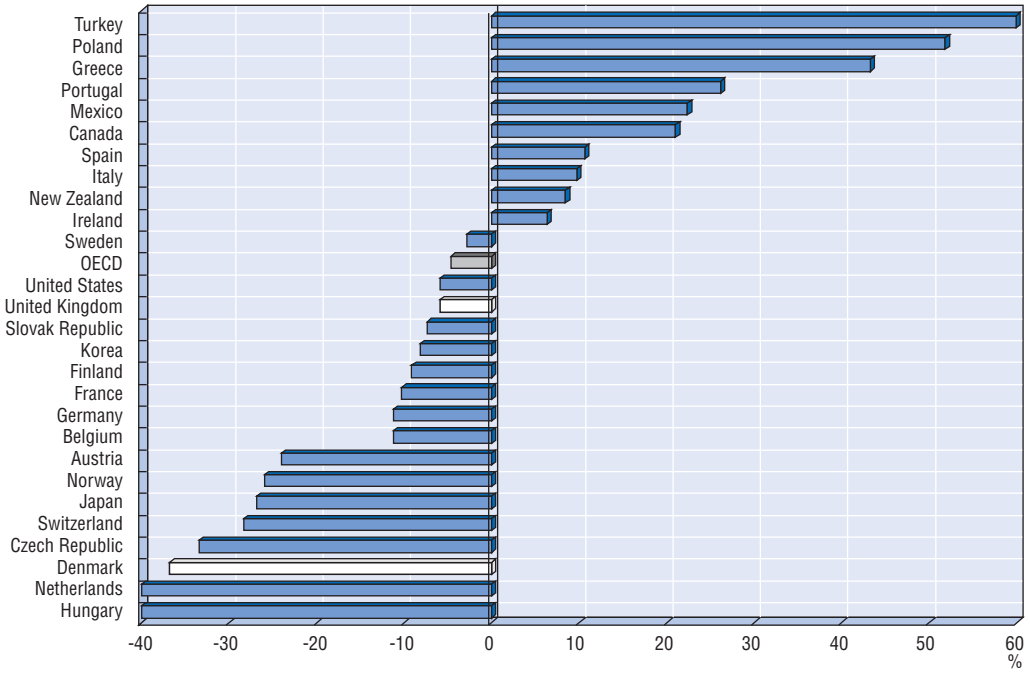
Edge of stream, pounds per acre, 1985, 2000, 2005 and targets for 2010



Source: Data provided by the Chesapeake Bay Program Office.

Figure 7.10. **Change in usage of pesticides in OECD member countries**

Percentage change in the number of tonnes active ingredients between 1990-92 and 2001-03



Caution is required in comparing trends across countries due to differences in definitions, coverage, etc.

Source: Based on OECD (forthcoming).

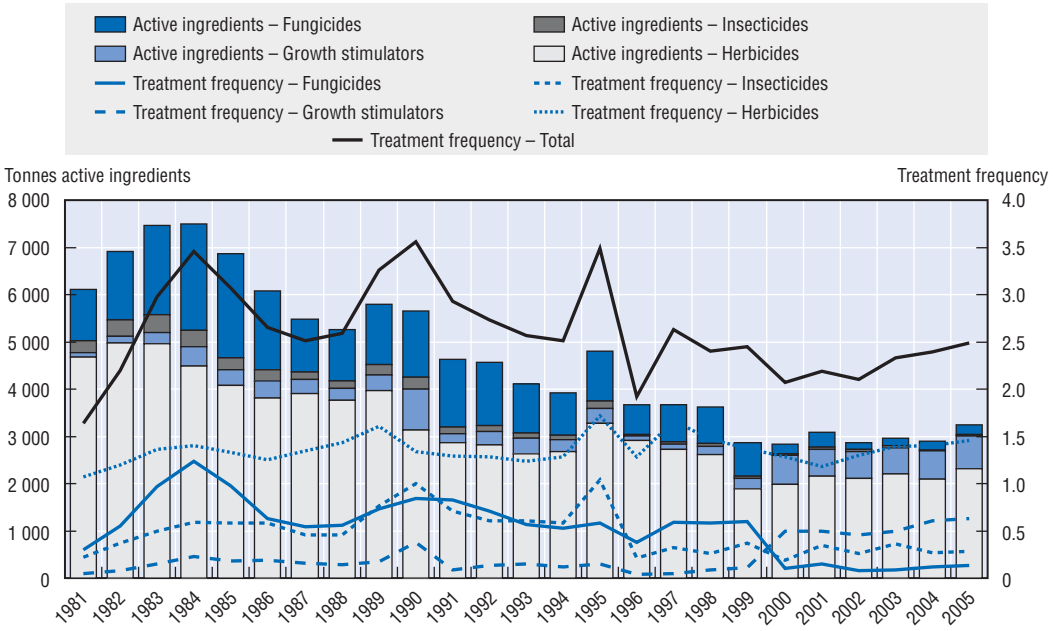
When judging these developments, one should take into consideration that *e.g.* the toxicity of different active ingredients varies greatly and that many low-dose pesticides have come on the market over the last decade. Greater use of low-dose pesticides tends to reduce the number of tonnes active ingredients, without necessarily reducing the related environmental risks.

Keeping these caveats in mind, and looking in particular at the total amounts of pesticides applied in the two countries that have been the focus of this project, it seems that the (partly) tax-based approach of Denmark has been more environmentally effective than the (mostly) voluntary-based approach in the United Kingdom.

Looking specifically at Denmark, Figure 7.11 suggests that although the number of *tonnes of active ingredients* of pesticides used decreased significantly following the issuing of the Action Plan on pesticides in 1986, the *treatment frequency* showed major variations from year to year, but did not decline much overall. In fact, the treatment frequency was higher in the period 2003-05 than in the early 1980s. Hence, one can hardly say that the instrument mix applied has been effective in fulfilling the stated environmental targets.

Figure 7.11. **Pesticides use and treatment frequencies**

Denmark, 1981-2005



Source: Statistics Denmark, *www.statistikbanken.dk/statbank5a/default.asp?w=1280.*

GEUS (2005) indicates that the number of samples *taken in groundwater* where the limit values for pesticides in drinking water have been exceeded has increased somewhat in recent years in Denmark, see Figure 7.12 below. However, in samples *taken in wells actually used for water abstraction*, the number of cases with a higher content of pesticides than the drinking water limit of 0.1 μg per litre has decreased.[78]

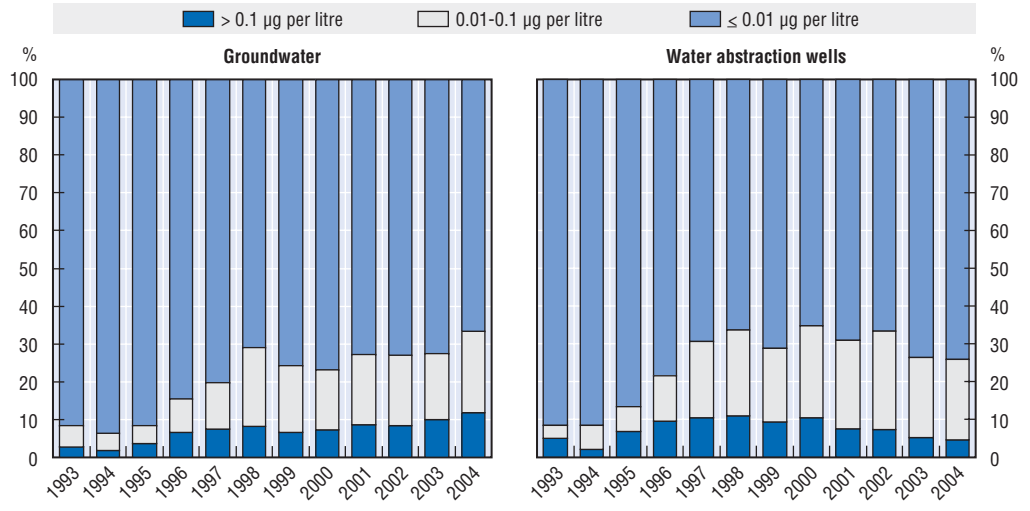
7.15.3. Residential energy efficiency

Figure 7.13 illustrates trends for total residential energy use per capita in OECD countries for the period 1970-2004. Trends are differentiated, with a steady increase in some countries and significant decreases in others.

Focusing only on electricity, the consumption has been rising for most countries. An important driver of this growth in electricity use is the increase in electrically powered equipments and appliances used in buildings. For example, the number of electric appliances per European household has increased tenfold over the last three decades [see IEA (2006)].

Figure 7.12. **Pesticides in groundwater and water abstraction wells**

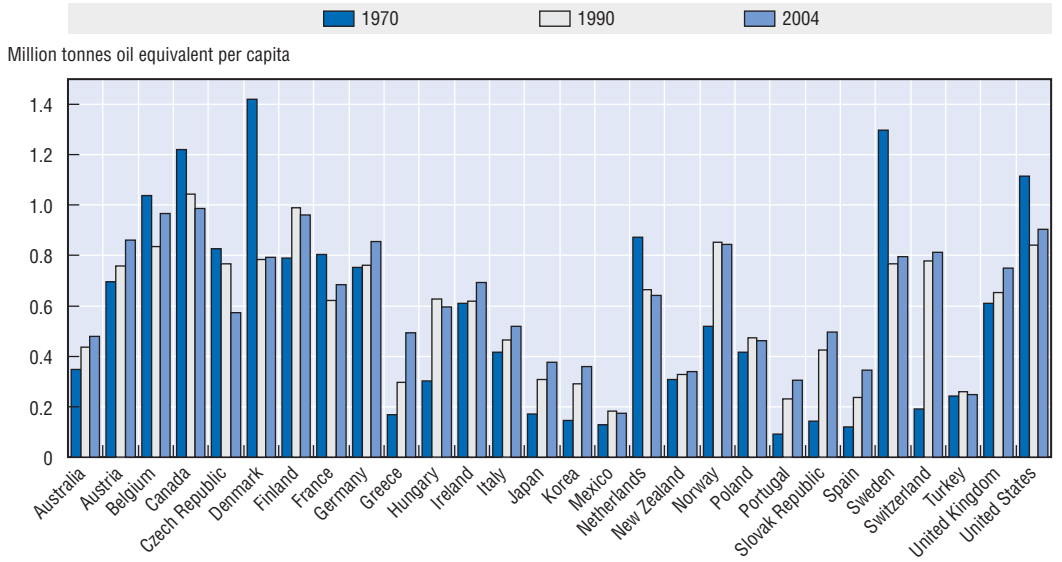
Denmark, 1993-2004



Source: GEUS (2005).

Figure 7.13. **Residential energy use per capita in OECD countries**

Million tonne oil equivalents per capita, 1970, 1990 and 2004



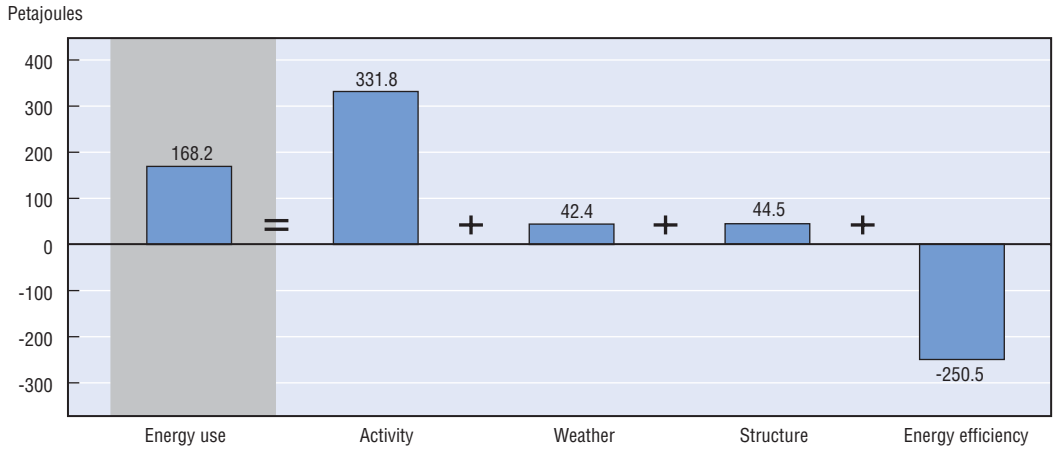
Source: IEA.

The residential sector accounts for 17% of secondary energy use in Canada. Between 1990 and 2003, residential energy use increased by 13% according to OEE (2005).[79] Without energy efficiency improvements, energy use would have risen by 32% between 1990 and 2003, instead of the observed 13% increase. Figure 7.14 decomposes the change in residential energy use that took place (168.2 PJ), with the following explanation:

- "activity, defined as a mix of households and floor space, increased by 26%, resulting in a 331.8 PJ increase in energy and a corresponding 18.2 Mt increase in GHG emissions. Growth in activity was driven by a 29 % increase in floor area and by a rise of 23% in the number of households;

Figure 7.14. **Breakdown of changes in residential energy use in Canada**

Petajoules, 1990 to 2003



Source: OEE (2005).

- the winter in 2003 was colder and the summer was warmer than in 1990. As a result, energy demand for space conditioning purposes increased by 42.4 PJ and GHG emissions rose by 2.3 Mt;
- changes in the structure (*e.g.* mix of end-uses); specifically, increases in the relative energy shares of water heating, lighting and space cooling, resulted in the sector using an additional 44.5 PJ of energy and emitting 2.4 Mt more GHGs; and
- improvements to the thermal envelope of houses and to the efficiency of residential appliances and space and water heating equipment led to an overall energy efficiency gain in the residential sector, saving 250.5 PJ of energy and 13.7 Mt of GHG emissions."

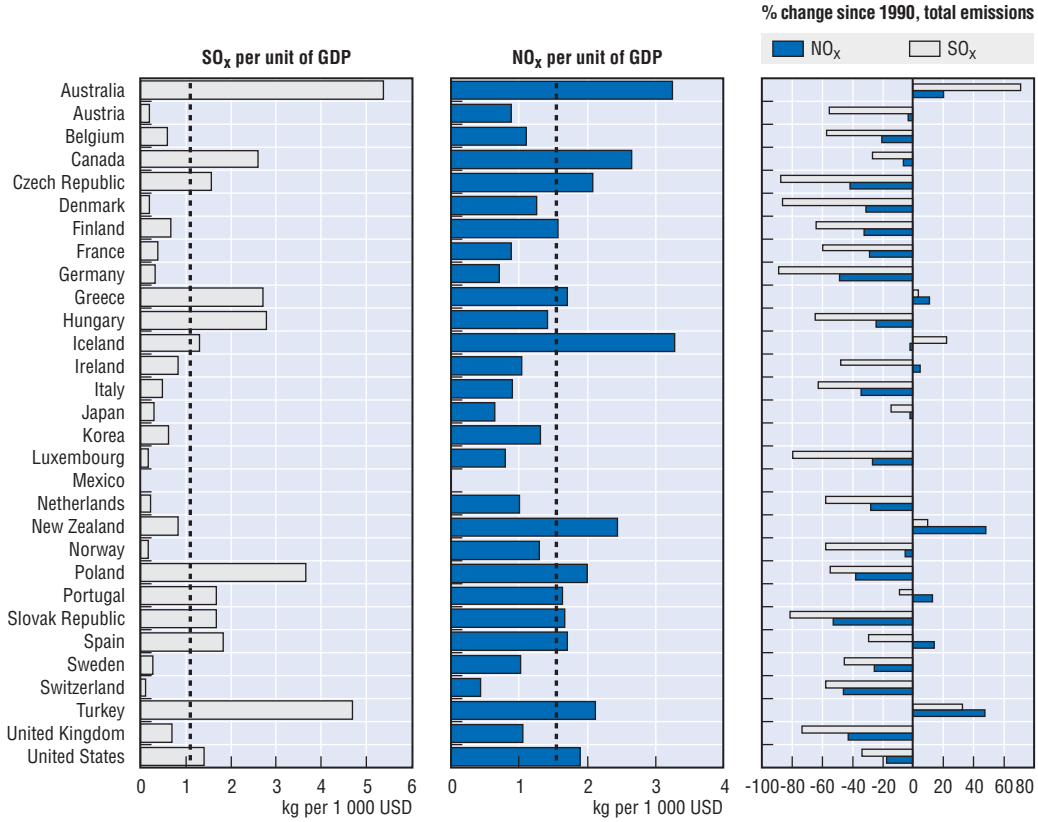
It is not clear to what extent the observed increase in energy efficiency comes in response to the instruments that have been put in place, or whether they would have taken place in any case, *e.g.* as a result of higher energy efficiency in foreign products households choose to buy.[80]

7.15.4. Regional air pollution

Figure 7.15 illustrates changes in emission intensities for SO_x and NO_x in all OECD countries since 1980. Focussing on the two countries studied for this report, it seems that the instrument mix applied in Sweden has resulted in larger environmental improvements than the mix applied in Canada has achieved. One can also notice that, although SO_x and NO_x intensities in Sweden are among the lowest among the OECD countries, the opposite is the case for Canada. Hence, Sweden's use of several economic instruments – providing clear incentives for polluters to reduce their emissions – can seem to have been more environmentally effective than the Canadian approach.

Figure 7.16 illustrates developments in total NO_x emissions in Sweden between 1990 and 2004. In total, the NO_x missions fell from 306 000 tonnes in 1990 to 197 000 tonnes in 2004, *i.e.* by 35%. By far, most of the emission reductions between 1990 and 2004 took place in the transport sector. Further action must be taken if the target set for 2010 is to be met, and Swedish Environmental Objectives Council (2006) indicates that additional measures *will* be taken, concentrated on the transport sector.[81]

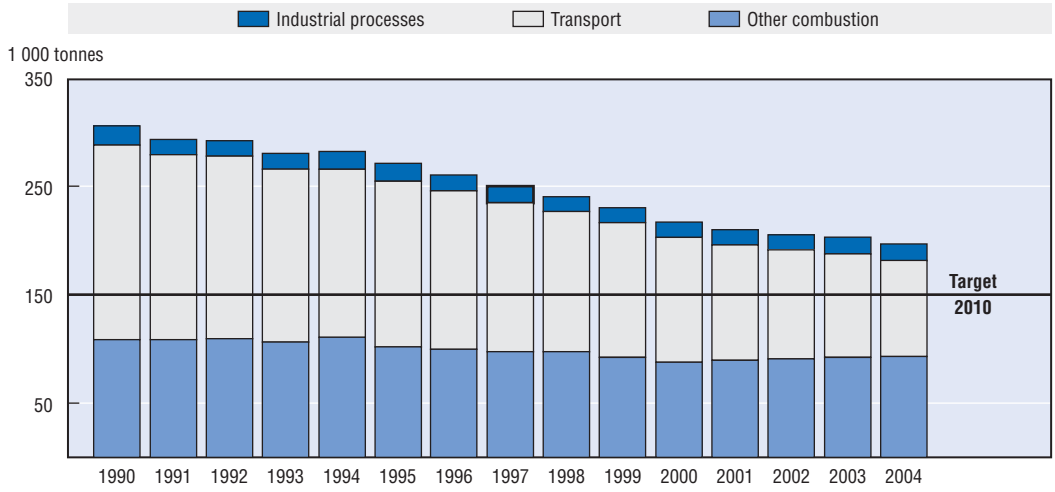
Figure 7.15. **Emission intensities for SO_x and NO_x**



Source: OECD (2005d).

Figure 7.16. **Total emissions of nitrogen oxides in Sweden**

1990–2004

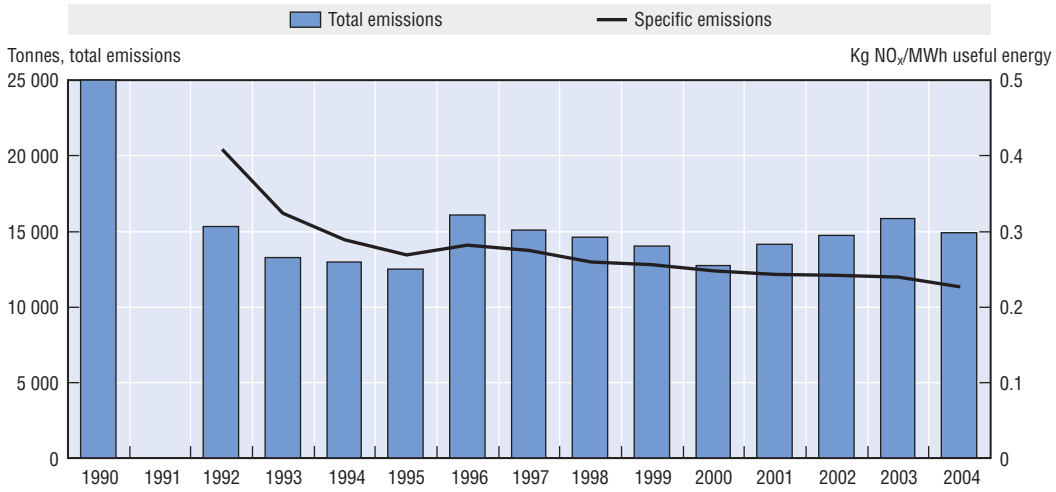


International bunker fuel emissions are excluded.

Source: Swedish Environmental Objectives Council (2006).

Although it affects only 8% of total Swedish NO_x emissions (*cf.* Figure 5.1), it is of interest to look closer at the environmental impacts of the charge on NO_x from combustion plants over a certain size in Sweden.[82] Figure 7.17 illustrates that this charge has been quite effective in reducing emissions from the plants that it covers – in part, because the rate is quite high, at EUR 4.4 per kg NO_x emitted.[83] Figure 7.17 illustrates both *total emissions* from the (increasing number of) plants covered, and the *emissions per MWh* energy produced. An increase in the number of plants covered by the charge explains why total emissions have not decreased even more than what Figure 7.17 shows. Emissions per unit energy produced are, however, now less than a half of what they were in 1990 (before the charge was introduced).[84]

Figure 7.17. **Absolute and specific NO_x emissions from combustion plants in Sweden**



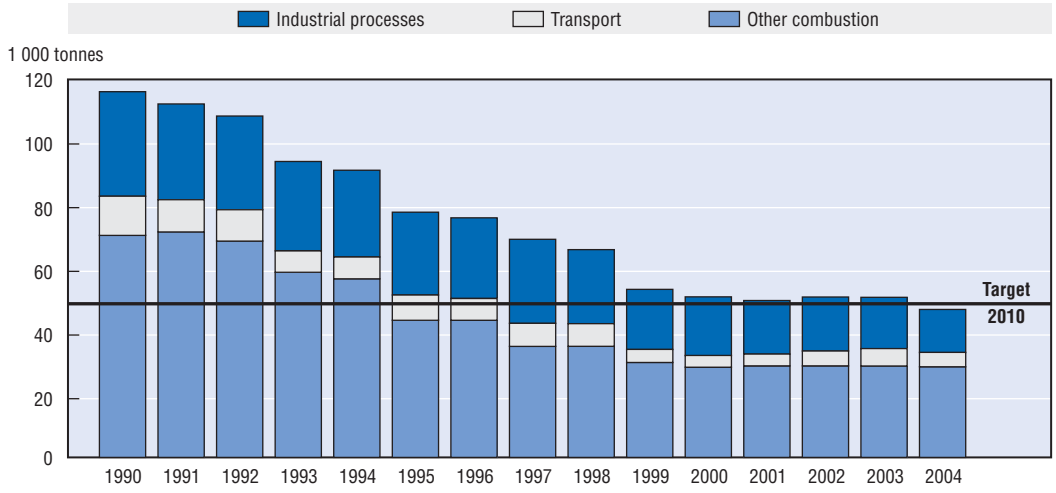
Source: Swedish Environment Protection Agency.

Figure 7.18 illustrates that total SO_2 emissions in Sweden have decreased strongly, from almost 120 000 tonnes in 1990 to 47 000 tonnes in 2004. The 2010 target – which in 2005 was revised downwards by the Swedish Parliament, from 60 000 to 50 000 tonnes – have thus already been met. By far the largest emission reductions have taken place in the category "other combustion".[85] This is probably related to the Swedish sulphur tax (introduced in 1991), which has led to a fall in the sulphur content of oil-based fuels of more than 50% beyond the legal standards. The sulphur content of light oils has now fallen below 0.076% (*i.e.* less than half the legal limit of 0.2%). The tax is estimated to have reduced emissions of sulphur dioxide by 80%, compared to 1980 [see Nordic Council of Ministers (1999)].

7.15.5. Mercury emissions to air

Figure 7.19 compares emissions to air of mercury in Norway and Sweden between 1990 and 2005. In both countries, the emissions were *more than halved* over this period. The emissions are of a similar magnitude in total in the two countries – but, as Sweden is larger than Norway, emissions per capita or per GDP unit are significantly higher in Norway than in Sweden. In both countries there has been some increase in emissions in recent years, which might be due to increased metals production in a favourable business climate.

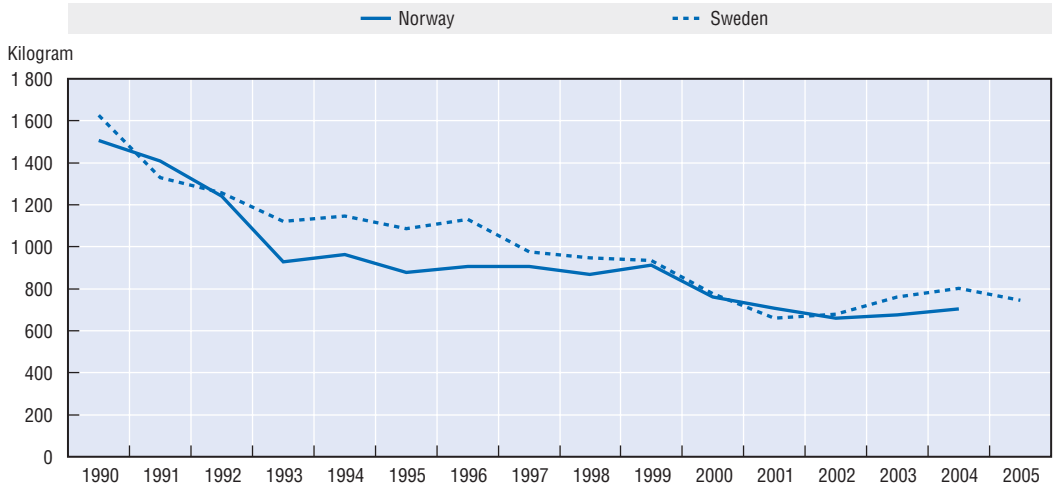
Figure 7.18. **Emissions of sulphur dioxide in Sweden**
1990-2004



International bunker fuel emissions are excluded.

Source: Swedish Environmental Objectives Council (2006).

Figure 7.19. **Emissions to air of mercury in Norway and Sweden**
Kilogram, 1990–2005



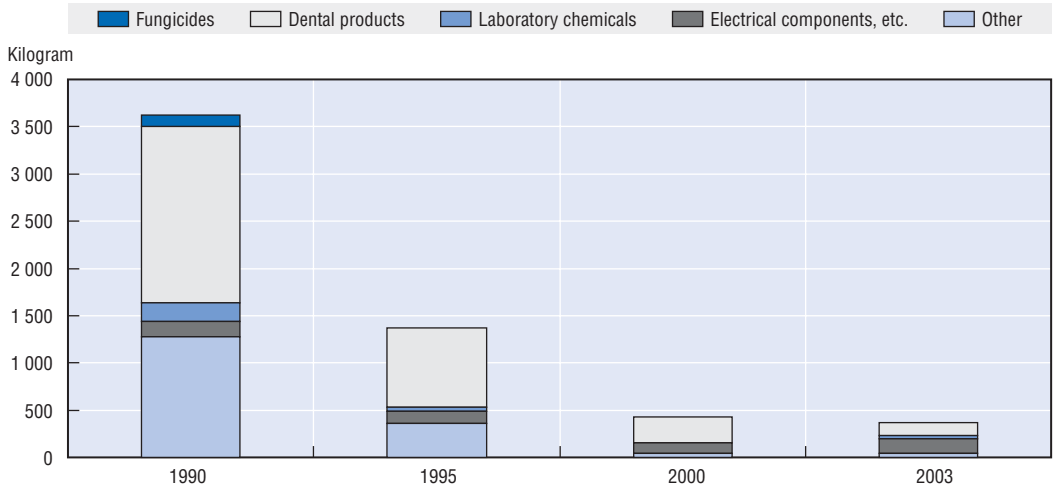
Source: Swedish Environment Protection Agency, Statistics Norway and *www.environment.no*.

Figure 7.20 shows that the use of mercury in products in Norway decreased from more than 3 600 kg in 1990 to 370 kg in 2003. Most of the reduction stem from dental products and "other" products. The use of mercury in electrical components, etc., was at the same level in 2003 as in 1990.

Changes in mercury depositions over Norway are illustrated in Figure 7.21. The figure shows concentrations of mercury per gram moss in different parts of the country – and indicates that very significant reductions in these concentrations have occurred in many parts of the country.

Figure 7.20. **Use of mercury in products in Norway**

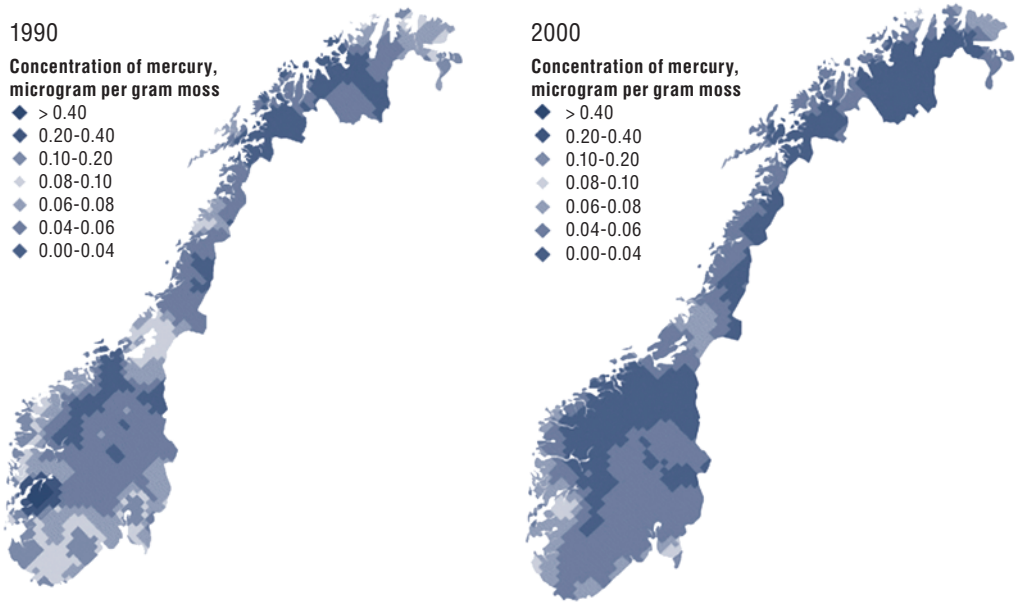
Kilogram, 1990–2003



Source: Statistics Norway and *www.environment.no*.

Figure 7.21. **Concentrations of mercury per gram of moss in different parts of Norway**

1990 and 2000

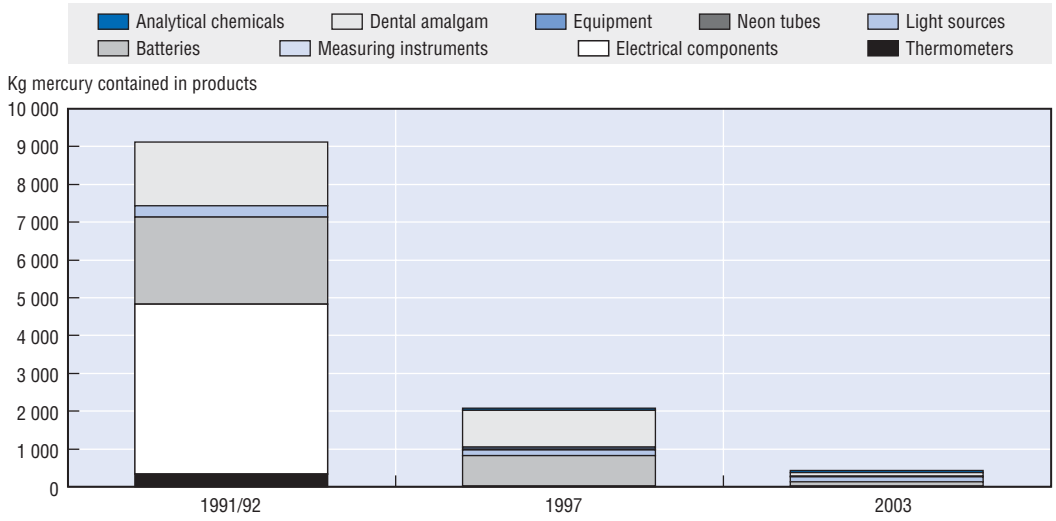


Source: *www.environment.no/templates/pagewide____4130.aspx*.

Figure 7.22 illustrates that the amount of mercury contained in products sold in Sweden in 2003 represented only a small fraction of the amount in 1991/92. The reductions in consumption are closely linked to increased use of substitutes to mercury. UNEP (2002) points out that besides legislation, improvements in technological performance have also been driving forces behind the substitution of mercury in many uses.

Total reported atmospheric mercury releases in the US have been reduced from 191 tonnes per year in 1990 to 107 tonnes per year in 1999, a 44% reduction [see ACAP

Figure 7.22. **Amounts of mercury contained in products sold in Sweden**



Amounts of mercury sold in Sweden contained in products, including batteries, light sources and dental amalgam. Sales of mercury metal (for chlor-alkali production, etc.) are not included.

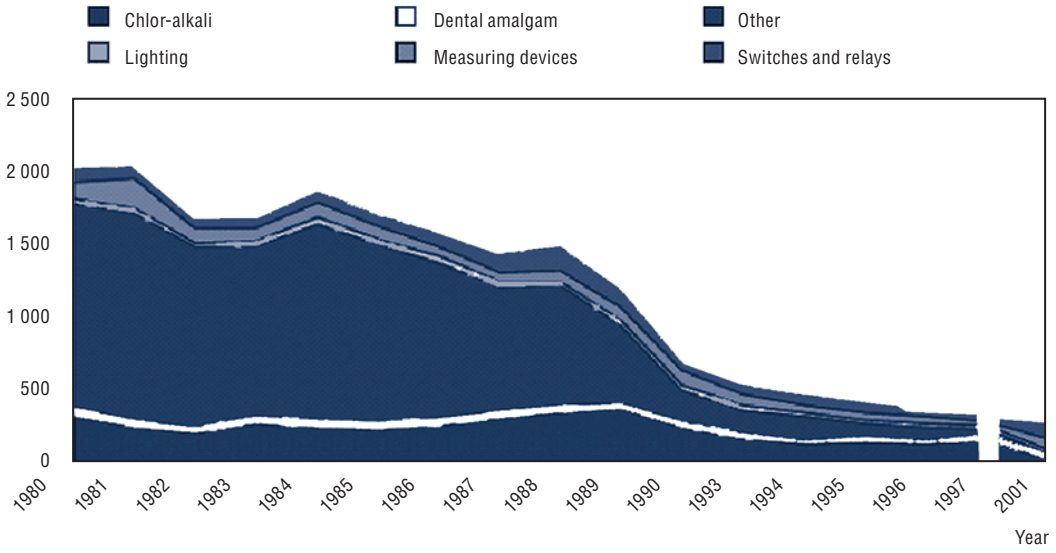
Source: Swedish Environmental Protection Agency, based on KEMI (2004).

(2004)]. Between 1980 and 2001, annual intentional mercury use – another indicator of effects of instruments – in the US shrank from 2 225 to 271 metric tonnes per year (US EPA, 2006c) – an 88% reduction.

Figure 7.23 shows developments in the use of mercury among industrial sectors in the United States. The figure shows a decrease in consumption of more than 75% since 1980. Between 1990 and 2001, total reported consumption fell by more than 50% – from more than 700 to less than 350 metric tonnes.

Figure 7.23. **Use of mercury in the United States**

Metric tonnes, 1980–2001

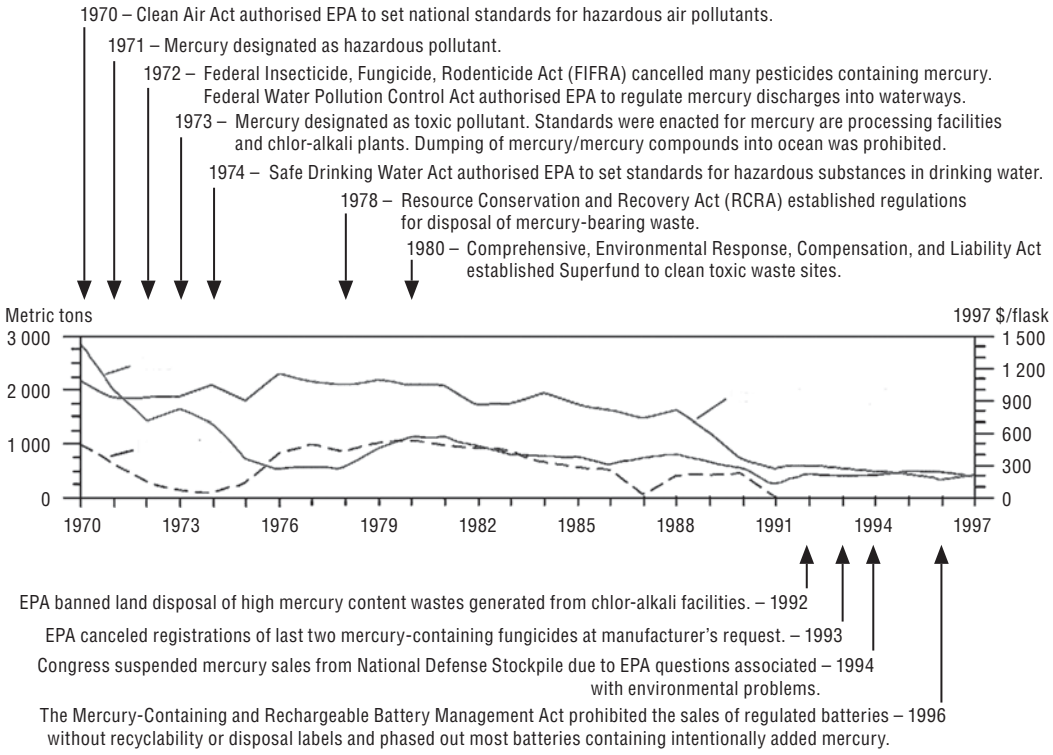


Source: US EPA (2006c).

The two major causes of this reduction were the elimination of mercury in batteries by regulation and technological advancements, and elimination of mercury-based fungicides in paints by regulation. Figure 7.24 provides more details on the phasing-in of various policies in the United States up to 1997, and their impacts on the production, use and price of mercury.

Figure 7.24. **Mercury production, consumption, prices and legislation in the United States**

1970–97



Source: Sznopek and Goonan (2000).

Figure 7.25 illustrates developments in emissions to air of mercury between 1990 and 1999 in the United States, and presents projections for how the emissions are expected to develop until 2020. Clearly, there was a very strong reduction in emissions from municipal and medical waste incineration between 1990 and 1999. As regards other emission sources, the developments are much more modest – and varied – with *increases* in the emissions from some sources. Whereas the reduction in emissions from coal-fired power plants was modest between 1990 and 1999, much more significant reductions in emissions will be made when the new cap-and-trade system comes into operation.

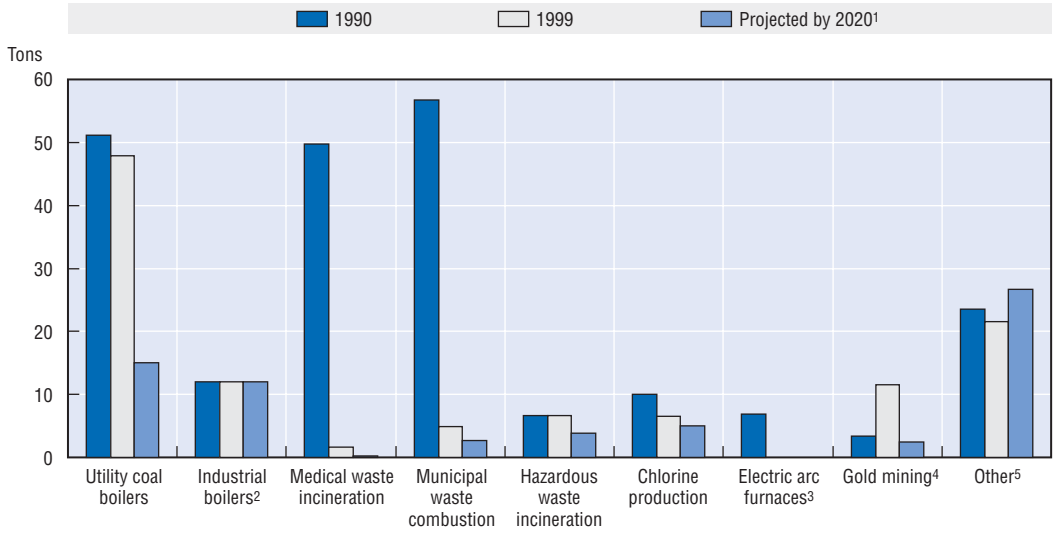
7.16. Are the mixes used at present economically efficient?

The short answer to this question is clearly "No" – but there are significant differences from case to case.

A first criterion to judge economic efficiency is whether the targets set in the different cases reflect a *reasonable balance between the benefits and the costs* to society as a whole of reaching them. Obviously, it can be difficult to make such an assessment, as it is technically

Figure 7.25. **Emissions to air of mercury in the United States**

Tons – 1990, 1999 and projections for 2020.



1 ton equals 0.9070 metric tonnes.

1. Except utility coal boilers. Fifteen tons per year will be achieved when full implementation of the Clean Air Mercury Rule is achieved, which may exceed 2020.
2. Growth in this sector is being offset by regulation.
3. Electric Arc Furnaces data not available for 1999. The 2002 estimate is 10 tons per year.
4. The 1990 emissions estimate is a preliminary estimate and is based on back calculations and assumptions using data from 1999 along with information about types of processes, production rates, and ores used in 1990 compared to 1999.
5. These projected emissions do not account for reductions from non-regulatory actions described elsewhere in the Roadmap.

Source: US EPA (2006c). Explanations are reproduced as they are provided in the source.

difficult – and often politically controversial – to place a "value" on the environmental impacts involved. However, in order to design "optimal" (or even "improved") instrument mixes, additional assessments seem to be called for in several cases.

It is far beyond the scope of this project to make a comprehensive assessment of all the targets – or lack of targets – applied in the environmental areas analysed.[86] The case studies have nevertheless indicated that further assessments of the strong focus on landfill avoidance in the household waste area could be warranted. Another issue that could benefit from further consideration is the use of "*dichotomous*" targets in several areas – for example regarding drinking water quality.

A second criterion to judge the economic efficiency of the instrument mixes used is whether or not *all* polluters contributing to a particular environmental problem face *equal incentives at the margin* to abate their emissions. This does not seem to be the case in *any* of the cases analysed. For example, agriculture is clearly treated more leniently than other sources of water pollution (*e.g.* in the Chesapeake Bay area). Households in most cases do not face waste collection charges that vary with the amount and the composition of the waste they generate, whereas businesses in most cases have to pay "tipping fees" that depend on the amounts of waste they dispose of. Different sources of mercury emissions to air also face different abatement incentives in all of the three cases analysed.

The large *diversity of instruments* applied to address residential energy efficiency also makes it unlikely that all the households in a given country face more-or-less similar marginal abatement incentives. For example, when subsidies are provided for certain types of energy efficiency improvements – but not for others – the incentives for improvements will differ. While not being panaceas that can be used to address *all* environmental problems, taxes and emission trading systems could provide at least similar abatement incentives for all polluters covered by the schemes.[87]

In addition to the issue of *differences* in marginal abatement incentives, it is of relevance for economic efficiency *whether or not major polluters have economic incentives to abate emissions at all*. Section 7.8.2 pointed out several examples of where this was *not* the case. For instance, most households have no economic incentive to reduce the amount of waste they set out for collection; no households in the cases analysed face energy prices that (fully) internalise the environmental externalities of energy use; industries are not charged for the costs to society of the air emissions on NO_x, SO_2 and mercury they create; and farmers in the United Kingdom and in parts of the Chesapeake Bay area have no economic incentives to reduce nutrients run-off or pesticides use.

There are several ways to create economic incentives to abate emissions: One could introduce subsidies for the most environmentally benign alternatives – but that would have important budgetary impacts, and could be in conflict with the "Polluter Pays" principle. Another option is to introduce taxes, fees or charges on polluting activities or products. This would raise revenues that could be used in various ways, but could likely lead to opposition from affected industries, etc. A third possibility is to create an "option-value" on pollution by establishing emission trading systems – whether based on cap-and-trade (with "grandfathering" or auctioning of permits), or on a baseline-and-credit system.

Notes

1. As it can be difficult to observe directly how many kg of manure are applied on a field, not to mention problems in measuring how much nitrogen there is per kg manure, the Directive allows member States to calculate the amounts referred to here on the basis of numbers of animals. Hence, many countries have set fixed rules for how many animals farmers may have per ha or similar. Such an approach, however, takes away the incentive for farmers to lower the nutrient content in the manure, *e.g.* by changing feeding practices. Clearly there can be a trade-off between what is administratively practical and what would provide the "best" incentives from an environmental point of view.
   The use of a "flow-related target" (*i.e.* kg N per year) for an environmental problem that is to a significant extent dependent on the total "stock" of pollutants in the soil is discussed further in Section 7.4.
2. However, the Directive does not contain any concrete requirements for sources. It is up to the member States do identify those sectors where cost-effective measures could contribute to achieving the ceilings.
3. There can be some conflicts between these objectives. For example, variable municipal waste collection charges – which are used in some parts of the Netherlands, but not in the United Kingdom at present – can help limit total waste generation and promote recycling, but could also (to some extent) trigger illegal disposal. Dijkgraaf and Gradus (2004) did not find evidence that the variable charges in the Netherlands have led to illegal dumping. OECD (2006a) discusses further the impacts of variable waste collection charges.
4. AEA Technology (2006) evaluates the effectiveness of recent pilot incentive schemes in England in encouraging households to reduce, reuse, recycle and compost their waste – based on assessments from local authorities. Interestingly, the study finds that only 57% of the trials had a positive, attributable, impact that increased the tonnage of recyclables collected.

5. The tax rates in both cases seem to be higher than what an internalisation of the externalities related to landfilling would imply – especially in the case of the Netherlands.

6. In addition, one could want to increase residential energy efficiency *inter alia* due to the close links with some non-environmental concerns, such as "energy security".

7. One should, however, always consider whether the benefits to society as a whole (in terms of additional environmental improvements, etc.) of applying an additional instrument would outweigh the additional costs to society of implementing this instrument.

8. See, for example, Jänicke *et al.* (1998) for a discussion of instruments addressing the energy efficiency of refrigerators in Denmark. In addition to an increase in the tax on electricity and the use of a labelling system, they *inter alia* found that training given to sales staff in the retail trade had important impacts.

9. Clearly, a number of households would seek to take *public good* impacts of their behaviour into account if they were well informed – but this is not likely to be the case for all (and perhaps not even a majority of) households.

10. The situation would be somewhat different regarding *e.g.* a combination of a label concerning the energy efficiency of electric appliances and a cap-and-trade system for CO_2 emissions covering the power-plants that produce the electricity used to run these appliances. In such a case, the total emissions from the power-plants (and from any other firms covered by the cap-and trade system) would be determined by the cap. Hence, even if the labelling system could contribute to a reduction in electricity use in households, with a cap-and-trade system in place, this would not impact much on total emissions from the sectors included in the trading system.

11. A new Code of Practice on pesticides use was issued in the United Kingdom in 2006; *cf.* DEFRA *et al.* (2006). This Code has an interesting legal status. Users are told that: "If you follow its advice you will be doing enough to keep within the law. But you may be able to work in a different way from the Code as long as that way is just as safe. If you are prosecuted for not following the Code, a court will find you guilty unless you can show that you have obeyed the law in some other way."

12. This is – of course – not a critique of farmers as a group, and would be equally valid for any other part of society. While better information to some extent can help shape behaviour in a more environmentally manner, other "carrots" or "sticks" could be necessary to trigger significant and lasting behavioural changes.

13. The Norwegian pesticides tax can thus probably better reflect the environmental damage of each product than similar taxes in Sweden and France (where they are based on the weight of the active ingredient) and in Denmark (where the tax is based on the price of each product). The pesticides tax in Belgium is also based on the weight of the active ingredient – but exempts all agricultural use.

14. Examples include the (167-page) Code of Practice on Pesticides Use in United Kingdom; [see DEFRA *et al.* (2006)], the (111-page) User's Guide to "*The Producer Responsibility Obligations (Packaging Waste) Regulations 1997 (as amended)*" in United Kingdom, the (66-page) brochure explaining the manure quota trading system to Dutch farmers and the (89-page) *Home Information Pack Regulation 2006* in the United Kingdom.

15. Various web-based technologies can help present relevant information in user-friendly ways – but the issue of the complexity of the instrument that is being explained could still be of relevance. For example, for *some* potential users, accessing any kind of information via the Internet remains an obstacle in itself.

16. Akerlof (1970) discussed this problem in detail, using the market for automobiles as an example.

17. The Information Pack must in particular contain an *Energy Performance Certificate*, which will provide prospective purchasers with an assessment of the property's energy consumption, together with a list of practical measures that can be taken to cut their fuel bills and carbon emissions. This is required by the EU *Energy Performance of Buildings Directive* (2002/91/EC). For further details, see: *http://europa.eu.int/eur-lex/pri/en/oj/dat/2003/l_001/l_00120030104en00650071.pdf*.

18. The length of the *Home Information Pack Regulations 2006* gives an indication that there will still be significant transaction costs involved in selling and buying a home, *cf.* HMSO (2006).

19. While the use of phytase to limit phosphorous losses from animals is wide-spread in Denmark, this seems more to be considered as a "new approach" in the Chesapeake Bay area. According to US EPA and USDA (2006), while 75% of the 2010 target of use of phytase feed additives for poultry had been reached by 2004, 0% of the target regarding the use of such additives for swine production

had been met by that time. Part of the explanation for the difference between the two countries is perhaps that a Danish industrial company is a globally leading phytase producer – a fact that might have enhanced the attention among both authorities and farmers to this option.

20. For details on Canada's *Clean Coal Technology Roadmap*, see: *http://strategis.ic.gc.ca/epic/internet/intrm-crt.nsf/vwapj/cctrm_e.pdf/$file/cctrm_e.pdf*.

21. For further details, see: *www2.nrcan.gc.ca/ES/OERD/english/View.asp?x=1318*.

22. Similar to the US Clean Coal Technology Program to reduce SO_2 emissions.

23. It is emphasised that no fee in this case would be paid for the emissions for which the firm does hold an emission permit. While reducing *compliance-cost* uncertainty, this "safety valve" provision would create an uncertainty as regards fulfilment of the *environmental* objective of the scheme – unless additional provisions are put in place (for example an obligation for the firm to surrender permits in a subsequent year if it has previously used the "safety valve").

24. Beyond the case studies of this project, "safety valves" *are* used in the EU trading system for CO_2 emissions (with an additional provision that maintains the overall environmental effectiveness of the scheme) and in the planned CO_2 trading system in Massachusetts in the United States.

25. See Tietenberg (1998).

26. In England, the penalty for non-compliance is GBP 150 per tonne waste, while Wales and Scotland have set the penalty at GBP 200 per tonne.

27. Strategy Unit (2002) indicated that the United Kingdom risked fines of up to GBP 180 million per year for non-compliance with the targets of the EU Landfill Directive. NAO (2006) estimated – based on available plans for building alternative waste treatment facilities, etc. – that "potential penalties could be as much as GBP 40 million for missing the 2010 targets and GBP 205 million for missing the 2013 targets".

28. It is emphasised that the *Manure Quota Trading Scheme* is *not* the same instrument as the *Manure Transfer Agreement System* referred to in Section 7.4.

29. See OECD (2007a) and Mallia and Wright (2004) for further discussion.

30. While there are very good reasons for keeping highly toxic products off the market, one can note that formal cost-benefit analysis does not seem to play a major role in the authorisation process.

31. See OECD (2006a) for further discussion of these impacts.

32. For a further discussion of income distribution impacts of environmental policies, see Serret and Johnstone (eds.) (2006) and OECD (2006b).

33. According to National Energy Action, fuel poverty is present: "Where a combination of poor housing conditions and low income mean that the household cannot afford sufficient warmth for health and comfort. The widely accepted definition of fuel poverty is where a household needs to spend 10% or more of income to meet fuel costs." See *www.nea.org.uk/Glossary#DtoF*.
According to NAO (2004), 2.25 million households in the United Kingdom were classified as being in fuel poverty in 2002, compared to about 4.5 million households in 1999. Half of the reduction was estimated to stem from increases in income over this period, while the other half from *decreases* in energy prices that took place. Hence, policy instruments applied to improve energy efficiency seem to have made a very modest contribution. On the other hand, increases in energy prices will likely have increased the number of households in "fuel poverty" considerably since 2002.

34. This *could* include (elderly) people with a considerable wealth.

35. According to OFGEM (2006), of the energy savings that have been achieved *in the first year* of the new commitment period, 40% has been in the priority group and the 60% in the non-priority group. In relation to the earlier discussion of "split incentives", it is emphasised that this issue is *not* of much relevance as argument for subsidising the use of low-energy light bulbs, as users will generally reap all the economic benefits of purchasing these bulbs.

36. According to OFGEM (2006), to the end of the first year of this new period, the suppliers combined had achieved 60% of the overall target.

37. Eoin Lees Energy (2006) provides a more positive assessment of the *Energy Efficiency Commitments*.

38. Largely for simplicity reasons, the criteria focus on whether the household qualify for receiving certain types of social security benefits, etc. As is often the case, there is a trade-off between administrative costs and the ability to target the subsidies towards the desired recipients.

39. See *www.hm-treasury.gov.uk/media/571/C1/pbr06_chapter5.pdf*.

40. In an initial consultation on how to formulate the *Energy Efficiency Commitment* scheme for the period 2008-11, DEFRA (2006) raises the possibility of shifting the focus from energy efficiency improvements *per se* to carbon emission reductions. This could allow microgeneration measures and measures (other than energy efficiency) for reducing the consumption of supplied energy, for example behavioural measures, to be included under the scheme. *A priori*, this would seem to make the scheme capable of addressing the underlying environmental externality more directly.

41. As in the situation where – even after a considerable amount of preparatory work – a household only receives two light bulbs.

42. Energy supply companies should, *a priori*, have a stronger incentive to promote the sales of *new* energy-efficient appliances (that could lead to higher overall energy use) than to promote insulation of buildings (that with more certainty would tend to reduce future energy use). However, Figure 7.1 above provides no indications of any such "bias" in the measures implemented, as insulation measures represented a major part of the energy savings in both household categories.

43. If, for example, the energy efficiency case study had been broadened to cover energy efficiency in general – or climate change in general – more instruments would certainly have been added to address such concerns.

44. The latter did not, of course, protect the international competitiveness of the livestock farmers – as they were to close down their production – but it helped make the overall policy more "acceptable" among important "stakeholders". This could also have been classified as an income-distribution measure.

45. It is emphasised that this argument is definitively of a "second-best" nature. In a "first-best" world, stricter land use restrictions *could* be put in place, to reflect society's degree of preference for *e.g.* protecting the water quality of the Bay, even if this would place limits on land-owners' rights to draw economic benefits from their properties. Likewise, in a "first-best" world, adequate regulations would be introduced to limit nutrient loads stemming from septic systems (as is done regarding waste-water treatment plants), and the externalities related to road transport would be internalised in fuel prices, vehicle prices and/or road use charges.

46. Collins and Basden (2006) provide an *ex post* evaluation of subsidies for transportation of poultry litter in the neighbouring state of West Virginia. They indicate *inter alia* that it is difficult for such programmes to stimulate uses of the litter that would not also be dependent on subsidies in the long term. US EPA and USDA (2006) state that "State subsidies in Maryland and Delaware cause displacement of markets (*i.e.* Pennsylvania haulers cannot compete, even in Pennsylvania)".

47. The price-elasticities Dijkgraaf and Gradus (2004) found for systems with payments per bag for both unsorted and compostable waste were almost as high as the elasticities of a system based on weighted waste amounts. This can be important, because the administrative costs of a payment-per-bag system are likely to be significantly lower than the costs of running a weight-based system.

48. For further discussion of impacts of variable waste collection charges, see OECD (2006a) and (2006b). Bartelings *et al.* (2005) provide a brief overview of studies of impacts of landfill taxes and various waste collection schemes.

49. "Recommendation 1" in Strategy Unit (2002) was that: "Local authorities that wish to take forward household incentive schemes to help reduce waste volumes and increase recycling should be allowed to do so. Schemes could include Council Tax discounts for households that compost; Rewards/prizes for homes that recycle; and variable charging schemes to reduce Council Tax – those who reduce waste and recycle pay less."

50. With the introduction of the EU Emission Trading Scheme (ETS) for CO_2 emissions in 2005, *some* of the externalities related to domestic energy use have been internalised in energy prices. The ETS covers electricity generators, which has contributed to increases in electricity prices – also affecting the households in the United Kingdom.

51. As the rest of the economy continues to grow, this trading system is likely to increase in importance. Allowing trades between sources that face different marginal abatement costs can clearly increase the economic efficiency of the overall abatement effort.
    It will, however, be of vital importance for the environmental effectiveness of the whole effort to limit nutrients run-off to the Chesapeake Bay how the baselines for farmers are defined, and how the size of the offsets they can sell are calculated. If not enough rigour is applied when calculating the offsets, the environmental effectiveness of the current regulation of point sources will be weakened.

Due to the strong interactions among many "best management practices", one clearly should *not* assume that the impact of the implementation of one such measure at a given farm is independent of the existence of other such practices at that farm.
This problem is closely linked to a (voluntary) baseline-and-credit option, combined with a cap-and-trade system; there will always be a risk of "adverse selection". Montero (2000) found that the "substitution provision" of the US EPA SO_2 Allowance Program tended to be taken up by firms which were allocated permits *gratis* far in excess of actual emissions [see OECD (2003b)].

52. On the other hand, Maryland can seem to apply somewhat stricter measures towards agriculture than other States in the Chesapeake Bay area (even if Figure 7.9 indicates that progress in Maryland in recent years has been modest). If that is correct, the difference in marginal abatement costs in agriculture compared to other sectors would *a priori* be lower in Maryland than elsewhere – which would make the potential benefits from trading lower.
However, in 2006 Maryland adopted a *Healthy Air Act* that will cut emissions of NO_x, SO_2 and mercury to air from six large power plants by 70-80% by 2009/10, compared to 2002 levels. (For further information, see *www.mde.state.md.us/Air/MD_CPR.asp*.) This will *inter alia* have important consequences for nitrogen loads to the Chesapeake Bay, as a large share of the nitrogen emitted to air is later deposited in the Bay. While this certainly is a welcome development, it would tend to increase the differences in marginal abatement costs of sources contributing to the pollution of the Chesapeake Bay, thereby strengthening the case for allowing point to non-point sources trades – even if managing trades of emissions to different environmental media (air and water) would still be complicated.

53. The information that Danish authorities have at their disposal allow the preparation of pre-filled nutrient accounts for each farm – as well as pre-filled applications for subsidies under the EU's Common Agricultural Policy – which eases the administrative burden placed on farmers considerably. It would not seem possible for authorities in the Chesapeake Bay area to do anything similar, as sales records from wholesalers, etc., there is considered "confidential".

54. In 2003, the average price of the accepted bids was about EUR 3.5 per kg nitrogen *quota* reduction.

55. It is debatable whether "*maintain* agricultural land in Good Agricultural and Environmental Condition" is the appropriate wording. In some areas of the EU, land has become so degraded, *e.g.* through nutrients run-off and pesticide leakage, that it makes little sense to claim that it is in what could be described as "Good Agricultural and Environmental Condition".

56. See OECD (2003a), (2005c) and (2006c) for further discussion.

57. A ban on landfilling of mercury-containing wastes could thus increase the need to establish separate collection schemes for such wastes, and to restrict the use of mercury-containing products.
It should, however, be kept in mind that mercury can evaporate from landfills. Further, the negative health impacts related to emissions to air of mercury stem primarily from methylmercury being formed from elemental mercury in water, leading to higher concentrations of methylmercury in fish eaten by humans. Methylmercury is also formed from elemental mercury by biological processes in landfills; landfills thereby can cause such water pollution "directly". The magnitude of this direct methylmercury pollution is still not well understood, whereas substantial mercury releases from waste incineration is relatively well documented in developed countries.

58. Goonan (2006) estimates that mercury-containing lamps led to emissions of 2.9 tons mercury in 2001, mostly linked to lamps that broke during transport to a landfill or an incinerator.

59. In a number of States in the United States, both emissions of mercury, SO_2, NO_x and CO_2 from power plants will in the future be governed by "cap-and-trade" systems. The more emission categories controlled by fixed "caps", the lower will be the likelihood that the promotion of mercury-containing lamps would pass a cost-benefit test.

60. This is in contradiction to the judgement of the European Court of Justice (2003a) – which did not focus much on environmental economics. Paragraph 74 of the judgement, for example, stated:
"Use standards such as those required by the Directive are applied beforehand and appear to be necessary for the purpose of reducing and preventing pollution, while the loss standards under the MINAS system are applied at a subsequent stage of the nitrogen cycle, and any exceeding of those loss standards will necessarily contribute to pollution."

61. Using just the Landfill Tax *was* indeed one of the options outlined in the "Limiting Landfill" consultation in 1999. However, 70% of respondents to that consultation argued for adding a trading scheme.

62. HM Treasury (2006) states *inter alia*: "Since 1997, the Government has introduced a number of measures to develop more sustainable waste management practices, reduce the UK's reliance on landfill and ensure that waste producers consider the full costs of the disposal of waste when

making decisions, including increasing the standard rate of landfill tax and introducing the Landfill Allowance Trading Scheme for local authorities. These measures aim to ensure that the UK will meet its international obligations, including the EU Landfill Directive. Defra is due to publish a review of its Waste Strategy in early 2007 and, as part of the 2007 Comprehensive Spending Review, the Government will assess the findings and consider whether there is a case for further incentives to encourage investment in more sustainable alternatives to landfill.
The landfill tax increases the price of waste sent to landfill, encouraging more sustainable ways of managing waste. The standard rate of landfill tax applying to active wastes (those that give off emissions), currently GBP 21 per tonne, has been increased by GBP 3 per tonne in each of the last two years as part of the Government's medium to long-term aim of reaching a rate of GBP 35 per tonne. The landfill tax has been very successful: overall quantities of waste recorded at landfill sites registered for the tax fell from around 96 million tonnes in 1997-98 to around 72 million tonnes in 2005-06, a reduction of around 25 per cent. The UK is on track to meet its 2010 targets under the Landfill Directive, although subsequent targets in 2013 and 2020 remain challenging. *The Government today announces that from 1 April 2007 the standard rate of landfill tax will increase by a further £3 per tonne to £24 per tonne. The Government will also consider whether the standard rate of landfill tax needs to increase more steeply from 2008 onwards, or go beyond the £35 per tonne already committed to for the medium to long-term, in order to encourage greater diversion of waste from landfill and more sustainable waste management options*" (Emphasis in the original).

63. The fact that these tax rates – significantly higher than the estimated negative externalities related to landfilling – do not suffice for the United Kingdom to comply with the obligations of the EU Landfill Directive can be related to the lack of economic incentives for households to reduce the amount of waste they set out for collection. It can, however, *also* indicate that the targets for landfilling of biodegradable waste set in the EU *Landfill Directive* are stricter than what the relevant externalities call for.

64. In 2005, the *Landfill Tax* raised GBP 733 million in revenue – equal to approximately 0.15% of total tax revenue in the United Kingdom. However, in connection with the ongoing increases in the tax rate for "active" waste in the tax, local authorities are given an allocation to meet the increased tax payments, with the intention being that the reform should be revenue-neutral across local authorities. Hence, the *net* amount of revenue raised is much less than what these numbers indicate.

65. It is interesting to note that a recent evaluation of the instruments addressing agriculture's contribution to the problems of the Chesapeake Bay [see US EPA and USDA (2006)] seems to take "for granted" that agriculture should be addressed primarily by voluntary programmes providing publicly-funded subsidies. The report does not discuss what improvements in water quality could be obtained if more restrictive instruments (*e.g.* non-tradable nutrient quotas for each farm) or stronger economic incentives (*e.g.* taxes on nutrients surpluses, or a tradable nutrient quota system) were to be applied in the sector – possibly in combination with stricter regulations on land use changes.

66. At present, the US EPA does not have a legal mandate to include other sectors in the trading scheme. Over time, new legal mandates could, however, be obtained.

67. This fear for negative competitiveness impacts was probably more related to the other types of emissions covered by the tax than it was for mercury. *Total* Norwegian emissions of mercury to *all* environmental media (from both point and non-point sources) were estimated at 1.1 tonnes in 2003. With a tax rate of NOK 27 per gram, the *maximum* tax payment related to mercury – if all emissions were covered, and if no behavioural changes were to be triggered – would be less than NOK 30 million (or EUR 3.6 million) per year.

68. The fact that a given activity is already addressed by a certain instrument does, obviously, not "prove" that the activity is addressed in the best possible way – even if the expected environmental improvements are obtained. Sometimes one could achieve the same environmental improvements at a lower cost by using another instrument.

69. Zoughaib, Rabl and Spadaro (2006) estimated the environmental damages of incineration to be in the range of EUR –10 to EUR +22 per tonne of waste (depending on the assumptions made about energy recovery), while they estimated the environmental damages of landfilling to be in the range of EUR 6 to EUR 11 per tonne of waste.

70. Some epidemiological studies suggest that methylmercury is associated with cardiovascular disease in some populations.

71. OMB (2006) also indicated that the benefits of the *Clean Air Interstate Rule* – which sets tighter caps for emissions of SO_2 and NO_x from coal-fired power-plants, and as a side-effect will lead to significant reductions in emissions of particulate matter and mercury – would by far exceed the

related costs. The annual benefits of this rule (mostly stemming from the reductions in emissions of particulate matter) are estimated to between USD 12 and USD 152 billion, while the costs are estimated to between USD 1.7 and USD 1.9 billion.

72. However, the Bichel Commission (1999) found that less than 1% of human intake of pesticides in Denmark stems from drinking water, animal products and fish in total. These sources were therefore judged to be without significance for the overall impact. 84% of the human intake of pesticides was found to stem from berries, fruits and vegetables. Some policies aiming to affect the concentration of pesticides in drinking water could, however, also affect the intake of pesticides from (domestically produced) fruits, etc.

73. This target is based on the target set in the EU Drinking Water Directives – which in turn is based on WHO drinking water guidelines. WHO (1993), states *inter alia*:
"Experiments suggest that neither nitrate nor nitrites act directly as a carcinogen in animals, but there is some concern about increased risk of cancer in humans from the endogenous and exogenous formation of *N*-nitroso compounds, many of which are carcinogenic in animals. Suggestive evidence relating dietary nitrate exposure to cancer, especially gastric cancer, is available from geographical correlation or ecological epidemiological studies, but these results have not been confirmed in more definitive analytical studies. It must be recognized that many factors in addition to environmental nitrate exposure may be involved.
In summary, the epidemiological evidence for an association between dietary nitrate and cancer is insufficient, and the guideline value for nitrate in drinking-water is established solely to prevent methaemoglobinaemia, which depends upon the conversion of nitrate to nitrite. Although bottlefed infants of less than 3 months of age are most susceptible, occasional cases have been reported in some adult populations.
Extensive epidemiological data support the current guideline value for nitrate-nitrogen of 10 mg/litre. However, this value should not be expressed on the basis of nitrate-nitrogen but on the basis of nitrate itself, which is the chemical entity of concern to health, and the guideline value for nitrate is therefore 50 mg/litre."

74. UE EPA and USDA (2006) indicates that it is rather unlikely that this target will be met.

75. Poor, Pessagno and Paul (2007) presents "hedonic value estimates" of ambient water quality in the St. Mary's River sub-watershed of the Chesapeake Bay – looking at how differences in water quality affects house prices. (The main source of increasing amounts of suspended solids and dissolved inorganic nitrogen in that watershed is said to be urban development rather than run-off from agriculture.) The authors *inter alia* found a marginal implicit price of a *one milligram per litre* increase in the concentration of dissolved inorganic nitrogen to be USD 17 642.

76. It could be argued that one should consider the costs and the benefits of reducing the nitrate concentration further – rather than regarding a specific target as "carved in stone".

77. "Edge-of-stream" load estimates illustrate fluxes to local waters – prior to transport downstream where denitrification occurs. The focus on loads per acre corrects for any conversion of agricultural land to other land-uses – which in itself could have a positive or negative impact on net nutrient loads.
The assumptions used in the model regarding the effectiveness of certain agricultural Best Management Practices were modified from 2001 to 2002 – resulting in lower assumed nutrient reduction benefits for many of the practices. The numbers for 1985 and 2000 used in the graph have been corrected to reflect these modifications.

78. According to Det Økonomiske Råd (2004), the limit value for pesticides in drinking water was set at the end of 1970s based on a political desire that there should not be *any* pesticides present in the drinking water at all. 0.1 μg per litre was the lowest concentration measurable at that time. Today it is technically possible to measure even lower concentrations – but the limit value has remained unchanged.

79. As a result, residential energy-related GHGs (including those related to electricity) increased by 15%.

80. A similar comment could, of course, be made concerning most of the other illustrations of environmental impacts that are presented in this section.

81. It is beyond the scope of this project to make an assessment of where the costs per tonne abated of additional measures would be the lowest.

82. When the charge was introduced, only plants producing more than 50 GWh per year were included, but this limit was reduced to 40 GWh in 1995, and further to 25 GWh in 1996.

83. For comparison, the tax rate on measured NO_x emissions from waste incinerators in Norway is EUR 1.8 per kg – based on estimates of the social costs of such emissions. The revenues from the Norwegian tax are, however, not repaid to the incinerators affected. In France, a non-refunded tax of EUR 0.04573 per kg NO_x emitted is levied as part of the "Taxe générale sur les activités polluantes", (General Tax on Polluting Activities).

84. A side-effect of the NO_x charge is somewhat higher emissions of, among other things, carbon monoxide and N_2O, neither of which is regulated by the charge.

85. The emissions stemming from international bunker fuels are not included in Figure 7.18. Swedish Environmental Objectives Council (2006) points out that emissions from international shipping from Swedish ports, which are not covered by the target, were 88 500 tonnes in 2004 – almost twice the total for *domestic* sources.

86. For a recent valuation of human health benefits of controlling mercury emissions from US coal-fired power plants, see Rice and Hammitt (2005).

87. An efficiently working emission trading market would "automatically" create a common abatement incentive for all the units covered – regardless of whether the permits at the outset were allocated for free or auctioned. A tax would (only) do so if a common tax rate applies to all polluters.

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Chapter 8

Conclusions and Policy Recommendations

This chapter offers recommendations on how environmental instrument mixes should be assessed, and on how their design could be improved to enhance environmental effectiveness and economic efficiency. The chapter highlights the importance of cost-benefit analysis, instrument flexibility and the use of mutually enhancing instruments, but warns against instrument overlap. The chapter also advises that environmental policy instruments should not be over-burdened with non-environmental tasks.

8.1. Conclusions

8.1.1. *The specificities of the environmental issue*

There *are* many good reasons for applying a mix of instruments to address a given environmental problem – rather than relying on a single one. Chapter 1 summarised some theoretical arguments for the use of instrument mixes ("multi-aspect" environmental problems, non-environmental market failures, mutual reinforcement of different instruments, reduction of compliance-cost uncertainty, etc.), and Chapter 7 demonstrated that a number of the instrument mixes currently being used *can* find good support in economic theory. Practical considerations and the need to take into account non-environmental policy objectives also explain many of the instrument mixes that are in use at present.

For example, a majority of environmental problems have "multiple aspects". It is not only the *total amount* of surplus nutrients in an area that is of relevance for environmental quality – it matters also *when* nutrients are applied to the fields, *where* they are applied, *how* they are applied, etc. Similarly, it is not only the total amount of household waste that is of importance – the composition of the waste matters at least as much, as does the way in which the waste is recycled or disposed of. To address these different aspects of the problem, the use of several instruments is often required.

In general, it is best to apply environmental policy instruments as close to the underlying externalities as possible. However, in several of the cases discussed in this project, it would be impossible to apply instruments that target the externalities directly. For example, it is not practical to measure the run-off of nutrients from individual farms to surface or groundwater. In such situations, one or more "proxy instruments" will have to be applied. Also in such cases, it is important to address all the different "aspects" of the environmental problem in question.

8.1.2. *The instrument mixes currently being applied*

The fact that there often are valid arguments for applying several instruments to address an environmental problem does not, however, mean that all existing instrument mixes have been designed primarily to take account of these arguments. In several cases, additional instruments could usefully have been included in the mix, and in other cases, some instruments in use at present seem to have negative impacts on either the environmental effectiveness or the economic efficiency of the overall instrument mix – or both.

The case studies have, for example, revealed situations where the environmental effectiveness of an instrument mix is jeopardised because some potential instruments are not applied – or are only very partially applied. For example, limited regulation of land-use changes in the United States makes it difficult to apply instruments that could more effectively limit nutrient run-off from agriculture into the Chesapeake Bay. This highlights the need to have a comprehensive view of the instruments that are required to address a given environmental problem.

8.1.3. Economic incentives are vital

Ideally, everyone that contributes to a given environmental problem should have an economic incentive – at least at the margin – to change their behaviour in a more environmentally benign way. A tax, fee, charge and certain subsidies can provide such incentives directly; and so could obligatory or voluntary participation in a trading system. In several of the cases examined, significantly higher environmental effectiveness would seem within reach, had economic incentives to reduce emission levels been given to important categories of polluters. One example where improvements seem possible is the current prohibition of variable waste collection fees in the United Kingdom.

To the extent that certain environmental targets are seen as "given", reduced environmental effectiveness of the instruments that are applied would also tend to lower the economic efficiency of the instrument mix as a whole; because more economically distorting instruments could have to be applied in the sectors *are* being addressed, for the "given" targets to be reached.

A broadening of economic incentives can also promote economic efficiency. One example is the use of point- to non-point trading of nitrogen emissions to water in several States in the United States. As such a broadening (from only point-sources to also including non-point sources) could bring pollution sources with lower abatement costs into the trading system, overall economic efficiency could be enhanced. It is, however, necessary to watch out for "adverse selection" among participants, and to apply a "careful approach" when defining any offsets non-point polluters could be allowed to sell, in order to protect the overall environmental effectiveness of the scheme.

8.1.4. Mutually reinforcing instruments

In several cases, combining two instruments has been found to enhance the effectiveness and efficiency of both instruments. For example, a labelling scheme can increase the impacts of a tax, and *vice versa*. A well-designed system for separate collection of recyclables can enhance the net environmental benefits of a variable waste collection charge – *inter alia* by limiting the danger of illegal dumping of waste.

The scope for such positive interactions between instruments seems particularly important if the instruments in question provide affected decision-makers (households, farmers, industrial firms, etc.) with a *high degree of choice* as regards how they would comply. If one instrument "instructs" the target groups how they should behave – *e.g.* applying a specific "Best Available Technology" – there is limited scope for other instruments to enhance the environmental effectiveness of that instrument.

There is a difference in the scope for mutual reinforcement of environmental impacts between *price-based* instruments (*e.g.* taxes) and *quantity-based* instruments (*e.g.* a "cap-and-trade" permit system). Whereas an information-instrument can enhance the price elasticity of relevance for a tax (and thus contribute to a greater environmental impact of a given tax rate), the environmental outcome is directly *embedded* in a cap-and-trade system. An information instrument used in combination with such a system would, hence, primarily impact on the price of the permits – *not* on total emissions – and, thus, on the *economic efficiency* of the approach.

8.1.5. *Combinations of quota systems and other instruments*

Combining a cap-and-trade emission trading system, or a non-tradable quota system, with another instrument can more generally give different impacts depending on the particular circumstances:

- If the two instruments address the *same environmental externality*, and if they address the *same target group*, there will be *no additional environmental impact* of adding the other instrument to the cap-and-trade system, and *little, if any, impact on economic efficiency* (except for the administrative costs caused by the other scheme). This is, for example, largely the case with the Landfill Allowance Trading Scheme and the Landfill Tax in the United Kingdom – except that the Landfill Tax also addresses industrial waste generators, as well as municipal waste.
- If the two instruments address the *same environmental externality*, but *different target groups*, with the target group of the "other" instrument indirectly being *covered* by the cap of the cap-and-trade system, there will be *no additional environmental impact*, but there *could be a positive impact on economic efficiency*, if marginal abatement costs within the target group of the "other" instrument are lower than the marginal abatement costs of the firms directly covered by the cap. The combination of the EU Emission Trading System for CO_2 (which includes power generation) and instruments to promote the energy efficiency of electrical household appliances can be a case in point.
- If the two instruments address the *same environmental externality*, but *different target groups*, with the target group of the "other" instrument having the possibility to *opt-in* under the cap of the cap-and-trade system, there could be significant *enhancements in economic efficiency* – due to larger variation in marginal abatement costs between participants – but there is a danger that the *environmental effectiveness could be compromised*, depending on the conditions set for the opt-in. Reference here is especially made to combinations involving a compulsory cap-and-trade system and a voluntary baseline-and-credit system, where there could be "adverse selection" of participants in the latter scheme. The stringency with which the "baseline" is defined is of vital importance for the overall environmental impact. Point- to non-point nutrient trading in some States in United States can serve as an example.
- If the two instruments address *different environmental externalities*, but with the cap-and-trade-type emissions from the target group related to the "other" instrument covered by the cap of the cap-and-trade system, there could in some cases be a *negative environmental impact* if a cap-and-trade system is added to another instrument. The example referred to here is the new cap-and-trade system for mercury emissions from coal-fired power-plants in United States, which will make measures to promote energy-efficiency through the use of mercury-containing lamps lead to net increases in mercury emissions.
- Other quota systems – tradable or non-tradable – can also have somewhat "surprising" impacts on the environmental effectiveness of instruments or measures with which they are applied in a mix. For example, whereas a reduction in agricultural subsidies normally should lead to a reduction in pollution loads from the sector, this is not *given*, if production quotas (*e.g.* on milk production or land-use) provide effective limitations of current production levels.

Several missed opportunities for promoting mutual enhancement between instruments have also been found in the case studies. In this connection, it is also

important to keep in mind that providing *information* to relevant "target groups" (households, farmers, etc.) will not make the persons involved *change their behaviour* unless they have a clear incentive to do so.

It is also important to emphasise that any information instrument needs to be adapted to the audience one seeks to address. Information brochures that are too long might not be read, and complicated labels might not be understood. There is also a risk that if several labels exist side-by-side in a given area, the "target groups" will be confused.

While not necessarily *mutually* reinforcing, it is evident that proper monitoring and enforcement mechanisms are required for instrument mixes to be environmentally effective. (A possible exception is when the other instruments by themselves provide clear economic incentives for behaving in an environmentally friendly manner.) It can sometimes be efficient to add an easily-monitored instrument to the mix, in order to underpin the working of another instrument, which has the potential to address a larger part of the environmental problem – *cf.* the Dutch nutrient case study.

8.1.6. "Safety valves" in quota systems

Economic theory suggests that it could be useful to use a "tax" or a fee as a "safety valve" in a tradable (or non-tradable) quota system, in order to limit compliance-cost uncertainty. However, even though several combinations of taxes/fees and quota systems *have* been found in the case studies examined by this project, in *none* of these cases has the tax or fee actually worked as an upper limit on the permit price.

This can in part be related to the widespread use of binding *annual targets* for emissions, waste disposal, nutrients applications, etc. Annual targets can be advisable when the related environmental problems depend on the *flow* of pollutants being released per year, etc. However, annual targets are sometimes also used in situations where the underlying environmental problem more depends on the total *stock* of pollutants that has been released (over a longer time period). Such latter cases can be adequately addressed, even if emissions are allowed to vary from year to year, as long as they over a defined period of time are constrained to specified limits. It can be difficult – and even meaningless – to introduce a "safety valve" on emission permit prices in a situation where an annual target is regarded as absolutely binding.

While it is not *generally* the case that applying more instruments would reduce total administrative costs, there can be some situations where this occurs – for example, if the costs of monitoring for the principal environmental instrument in the mix could be rather high, were it to be used in isolation.

8.1.7. Non-environmental policy concerns

Environmental concerns are obviously not the only objectives that are – or should be – given political attention in OECD countries, and a number of "other" priorities *are* taken into account when environmental policy instrument mixes are designed. In some cases, the design of instruments applied to address such concerns can have unexpected and unintended harmful impacts on the environment, and it is advisable to regularly "scan" instruments applied in such areas, to see if they are having such undesirable environmental impacts. Sometimes, such negative impacts can be corrected without compromising the non-environmental objective of the instrument in question. In other cases, a trade-off between environmental objectives and non-environmental objectives will have to be made.

Both to deal with such situations and to deal with situations where there can be conflicts between different environmental objectives, a method for comparing different impacts – in terms of "benefits" as well as in terms of "costs" – is needed. Monetising environmental and other benefits allows such comparisons – and comparisons of costs and benefits.

Sometimes, concern about non-environmental objectives has led to modifications of a "strictly" environmental instrument – like exemptions or rate reductions for certain industrial sectors in an environmentally related tax. However, such approaches will generally reduce the environmental effectiveness of the instrument in question, and can lead to significant increases in the administrative costs of the "environmental" instrument.[1]

In other situations, a given instrument is given a dual purpose – *e.g.* a subsidy scheme to increase the energy efficiency of houses of people living in "fuel poverty". The case studies have shown that it can be very difficult to design an instrument with a dual purpose in such a way that it *effectively* addresses the *environmental* externality of relevance.

In still other cases, a "non-environmental" instrument is implemented at the same time as an "environmental" instrument is introduced – as when an increase in social security payments is announced in parallel with an increase in taxes on energy. While a firm conclusion is difficult to draw from the relatively limited number of cases analysed in this report, there are indications that this approach to dealing with non-environmental policy concerns is preferable to the two approaches mentioned above.

The effectiveness of a given instrument mix will obviously be hampered if subsidies are given to "neighbouring" sectors of the economy, especially if these sectors are major contributors to the environmental problem in question. For example, reducing or reforming subsidies to agriculture could help reduce the negative environmental impacts that this sector generates.

8.1.8. Overlapping instruments

Whereas overlap between some types of instruments can be beneficial and/or mutually reinforcing (*e.g.* taxes and labelling schemes; taxes used as "safety valves" in trading schemes, etc.), overlap between other types of instruments (*e.g.* taxes and product standards) can both hamper the proper working of the instruments involved, and cause redundancies and unnecessary administrative costs.

In many cases, small design details can compromise (or enhance) the environmental effectiveness and/or economic efficiency of the instrument *per se* as, well as the instrument mix overall. This project has found several examples of instruments that functioned less well than intended – possibly because not sufficient time and resources had been spent in their preparation and design. It would seem desirable to implement instruments where later modifications can be made relatively easily – without causing major adjustment-problems for the regulated target groups.

8.1.9. Co-ordination is needed

This project has also highlighted the close links that exist between many areas of environmental policy, and between environmental and non-environmental policies. Addressing these interdependencies requires co-ordination within Environmental Ministries, and between Environmental Ministries and many other ministries. Such co-ordination can be promoted by shifting attention further from "end-of-pipe" solutions for a given problem, towards more *integrated control* of several environmental and non-environmental problems.

Close co-ordination is also required between different administrative levels involved in environmental policy – taking due account of the political and constitutional "setting" in each country.

And finally, a generally valid "conclusion" concerning the optimality of applying an instrument mix can not be given – a *case-by-case analysis* will inevitably have to be made. Hopefully, this project has provided some useful input into how such evaluations can be made.

8.2. Policy recommendations

8.2.1. Recommendations regarding assessments of targets and instruments

First and foremost, policy-makers should consider carefully whether the additional benefits of further environmental improvements and the additional costs to society of achieving these improvements balance reasonably well. This implies the need to assess the *targets* set for environmental policy.

If the sum of *all* the benefits of additional improvements can be expected to exceed the related costs, the improvements ought to be pursued. If the costs clearly outweigh the sum of all relevant benefits – even if the most efficient policy instruments were to be applied in order to reach the targets – a reconsideration of present priorities would be appropriate.

Policy-makers should also make in-depth *ex ante* assessment of any new instrument being considered for implementation – to assess what each instrument would contribute as regards environmental effectiveness and economic efficiency. They should also regularly undertake careful *ex post* reviews of the instruments they apply to achieve a given target – and eliminate or modify redundant instruments, *i.a.* to reduce administrative costs.

Whenever modifications are made to some parts of an instrument mix, a reassessment of the impacts of the other parts of the mix is – in principle – necessary.

Concerning both *ex ante* and *ex post* analyses, the *focus of the analysis should be on* "*outcomes*" (such as actual improvements in environmental quality) rather than on "outputs" (such as the number of persons given a particular subsidy, or the number of square km where pesticides are not applied).

8.2.2. Recommendations regarding the design of instrument mixes

Careful review of current or potential instrument use, as well as the design of improved instrument mixes, clearly requires that policy-makers have a good understanding of the environmental issue at hand, covering all the aspects of a "multi-aspect problem", and of the interrelations between this environmental issue and other related environmental and non-environmental policy issues.

Policy-makers should draw on such an in-depth understanding of the environmental issue and its interrelations to give adequate attention to *e.g.* different incentive-impacts of various design options when preparing individual instruments or a mix of instruments. Seemingly small changes in instrument design can sometimes have major impacts on environmental effectiveness and/or economic efficiency.

Both from the perspectives of environmental effectiveness and economic efficiency, policy-makers should apply instruments that address a given environmental problem *as broadly as possible*. In order to "bring in" sectors that have previously largely been exempted from regulation – or those which are so far only being addressed with voluntary approaches – it can be necessary to introduce seemingly "unrelated" instruments – such as *e.g.* limitations on the conversion of agricultural land to other uses, in order to address nutrient run-off from the sector.

Further, one should (as far as possible) provide *similar incentives at the margin* to all producers that contribute to the problem in question. Economic instruments can "automatically" provide equal marginal abatement incentives – especially as regards *total amounts* of a given pollutant. Depending on data availability, various regulatory instruments can (at least partially) do the same job.

For "multi-aspect" environmental problems, policy-makers should supplement instruments that address *total amounts* of pollution with instruments that address *the way* a certain product is used, *when* it is used, *where* it is used, etc. In many cases, regulatory instruments, information instruments, training, etc., can be better suited to address these dimensions than *e.g.* a tax or a trading system.

In order to exploit possibilities for various instruments to mutually underpin each other (so that the application of one instrument enhances the effectiveness and efficiency of another), instruments that *provide as much flexibility as possible* to the targeted groups should be used. Economic instruments will generally provide such flexibility – but *some types* of regulatory instruments can also do so.

Various labelling schemes and other information-instruments can enhance the environmental effectiveness of taxes, fees and charges – especially if they convey information on *private benefits*. Hence – with the general caveat of making sure that benefits exceed costs – such instrument combinations could usefully be applied. It is, however, important to make sure that any information instruments are well adapted to the "target group", and to avoid a confusing multitude of labelling schemes.

Except for situations where mutual reinforcement between instruments is likely, or when the instruments address different "aspects" of a given problem, policy-makers should generally *avoid introducing overlapping instruments* – such overlaps will tend to reduce the desired flexibility and create unnecessary administrative costs.

To maintain flexibility in the design of environmental policy, to promote economic efficiency and to create options for introducing "safety valves" in emission trading systems – which could limit compliance-cost uncertainty and promote stakeholder policy acceptance – policy-makers should avoid setting *annual targets* for environmental policy areas that can adequately be addressed, even if emissions vary from year to year.

It also seems advisable to address some *non-environmental policy objectives* (*e.g.* social concerns) primarily with non-environmental policy instruments (*e.g.* the social security system or the tax system), rather than to modify environmental policy instruments. This will *inter alia* make it easier to provide an incentive at the margin for low-income households to behave in environmentally benign ways.

Policy-makers ought to focus more on the impacts of environmental policy instrument and instrument mixes on the economy as a whole – rather than on the impacts for specific sectors – as instruments that make some firms worse off will generally make some other firms better off. However, to the extent policy-makers want to address any negative impacts on the competitiveness of certain sectors stemming from environmental policies, it is again important to provide an incentive at the margin to abate emissions. Such incentives can, for example, be provided through (both grandfathered and auctioned) emission trading systems and through environmentally related taxes – possibly with recycling of revenues back to the sectors of concern.

It can also often be preferable to primarily address *non-environmental market-failures* (market power, incomplete property rights, split incentives, etc.) with non-environmental instruments, such as competition policy instruments, improvements to patenting systems, deregulation of the housing markets, etc.

In order to address the many relevant needs for co-ordination – and in order to build "political acceptance" for new environmental policies, policy-makers should consider using (or expanding the use) of various consultation processes – both between relevant public authorities and with the public-at-large.[2]

As a basis for such consultations, correct and targeted information should be provided about the environmental problems *per se*, as well as its underlying causes and options available to address it.

Table 8.1 summarises some of the policy recommendations for enhancing the environmental effectiveness and/or economic efficiency of instrument mixes used for environmental policy.

Table 8.1. **Recommendations regarding instrument mixes for environmental policy**

Regarding how to assess instrument mixes, it is recommended to…
Carefully assess the benefits and costs of meeting current environmental targets.
Make in-depth *ex ante* assessments of any new instruments.
Regularly make *ex post* assessments of *all* instruments impacting on an environmental area.
Determine at the outset whether the issue at hand represents a "*single-aspect*" or "*multi-aspect*" problem.
Focus the assessments on "outcomes" rather than "outputs".
Focus on impacts for the economy as a whole – rather than on impacts for individual firms or sectors.
Assess carefully the environmental impacts of subsidies given to "other", non-environmental, sectors.
Assess regularly whether the design of other non-environmental policy instruments have unintended negative environmental impacts.
In order to develop instrument mixes that are environmentally effective and economically efficient, it is recommended to…
Apply instruments that address the environmental problem as broadly as possible.
Provide similar incentives at the margin to all polluters.
Have a *comprehensive view* on which instruments are required to create an environmentally effective and economically efficient instrument mix.
Supplement instruments that address total pollution level with instruments that address other aspects of "*multi-aspect*" problems: Where, when, how, etc.
Enhance possibilities for instruments to mutually reinforce each other by applying instruments that provide flexibility.
Use information instruments to enhance the environmental effectiveness of any taxes, fees or charges.
Pay attention to the incentive impacts of various instrument-design options.
Avoid overlapping instruments, except when they can mutually reinforce each other, or address different aspects of the environmental problem.
Avoid a confusing multitude of labelling schemes within a specific environmental area.
Avoid annual targets for environmental problems that can be adequately addressed even if emissions vary somewhat from year to year.
Put in place appropriate *monitoring and enforcement* mechanisms – to safeguard the environmental effectiveness of the instrument mix.
Consider carefully whether voluntary *opt-in possibilities* in emission trading systems could jeopardise the environmental effectiveness of the trading system – and/or enhance the economic efficiency.
Be *careful when defining the baselines* in any such opt-in options – in order to preserve environmental integrity of the whole scheme.
Address any *social concerns* related to environmental policy instruments primarily through non-environmental instruments.
Make sure to provide a positive incentive to abate at the margin *if* measures to limit sectoral competitiveness impacts are introduced.
Address any *non-environmental market failures* primarily through non-environmental instruments.

Notes

1. See OECD (2006b) for further discussion of these issues.
2. This issue is also discussed further in OECD (2006b).

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Instrument Mixes for Environmental Policy

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Further Reading

The Political Economy of Environmentally Related Taxes (2006)
The Distributional Effects of Environmental Policy (2006)
Cost-Benefit Analysis and the Environment: Recent Developments (2006)
Addressing the Economics of Waste (2004)
Tradeable Permits: Policy Evaluation, Design and Reform (2004)
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