The design features of environmental taxes

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ABSTRACT

This dissertation aimed at assessing what environmental taxes are. It was argued clear policy guidelines for their design follow from understanding them as regulatory instruments aimed at environmental policy goals. Empirical evidence drawn from institutional practices in Denmark (waste tax), Portugal (energy tax) and Sweden (energy tax, CO_2 tax, sulphur tax and the NO_x charge) showed that compliance with such guidelines, which allow the distinction between environmental taxes and environmentally-related taxes, are paramount to the environmental effectiveness of these instruments.

Both environmental taxes and pollution taxes or environmentally-related taxes are raised on polluting tax bases and highlight inefficiencies in abatement and opportunities for technological progress by putting a positive price on pollution, hence raising awareness and sharing responsibility. However, they are substantially different. Their underlying normative tax design is different following the different objectives they pursue. The more environmentally targeted tax design of environmental taxes makes them perform better than pollution taxes as instruments of environmental policy, producing stronger and quicker environmental effects than environmentally-related taxes raised on the same tax bases.

Environmental taxes aim only or primarily at fulfilling precise environmental objectives via behavioural change and technological progress and must be ruled by environmental criteria. Their design induces behavioural change by promoting tax awareness and tax avoidance, as well as by adopting a ‘forward looking’ approach provided by its reference to the opportunity for improvement rather than mere pollution amounts. Therefore, they must be raised on specific polluting emissions or a proxy for them. Their tax rate needs to be referred to pollution abatement costs or relative polluting impacts taking into account a specific pollutant, and be set at the level required to induce the behavioural change necessary to attain the environmental objective pursued. And they must be charged to polluters who control the cause *sine qua non* of pollution and still did not explore all their opportunities for environmental improvement.

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ABBREVIATIONS

AR	Assembleia da República (Portuguese National parliament)
BATNEC	Best available technology not entailing excessive costs
BE	Bloco de Esquerda (Portuguese leftwing political party)
CEFP	Comissão Parlamentar de Economia, Finanças e Plano (Portuguese Parliament Commission of Economics and Finance)
CO_2	Carbon dioxide
DAR	Diário da Assembleia da República (journal of the Portuguese national parliament)
DEPA	Danish Environmental Protection Agency
DGE	Direcção Geral da Energia de Portugal (Portuguese Directorate-General for Energy)
DL	Decreto-lei (Decree-law)
Ds	*Departementsserien* (Ministry publications series)
EAP	Environmental Action Programme
EC	European Communities
EEA	European Environment Agency
EELR	European Environmental Law Review
ETR	Ecological Tax Reform
EU	European Union
HSC	High Sulphur Content
IA	Imposto Automóvel (sale tax on motor vehicles)
IEA	International Energy Agency
IMV	Imposto Municipal sobre Veículos (municipal tax on vehicles)
INE	Instituto Nacional de Estatística (Portuguese National Statistical Institute)
ISP	Imposto sobre Produtos Petrolíferos (tax on oil products)
JE	Jornal Expresso (weekly Portuguese newspaper Expresso)
JLE	The Journal of Law and Economics
LPG	Liquid propane gas
LSC	Low Sulphur Content
Kt CO_2e	(Kilo)Tonnes of CO_2 equivalent
MARN	Ministério do Ambiente e Recursos Naturais (Portuguese Ministry of Environment and Natural Resources)
MENS	Swedish Ministry of the Environment and Natural Resources
MFP	Ministério das Finanças (Portuguese Ministry of Finance)
Mj	Megajoule
MMP	Maximum Market Price System
NCM	Nordic Council of Ministers
NUTEK	Swedish Agency for Economic and Regional Growth
OECD	Organisation for Economic Co-operation and Development
pct	Percentage/percentile
PNAC	Programa Nacional para as Alterações Climáticas (Portuguese Climate Change National Programme)
PPP	Polluter pays principle
PS	Partido Socialista (Portuguese Socialist Party)
SEPA	Swedish National Environmental Protection Agency

SFS	*Svensk Författningssamling* (The Swedish Code of Statutes)
SOU	*Statens Offentliga Utredningar* (The Swedish Government Official Reports series, Committee Reports)
VAT	Value Added Tax
VOC	Volatile organic compounds
tCO_2e	Tonnes of CO_2 equivalent
tep/toe	Tonnes of petroleum/oil equivalent
TFEU	Treaty on the Functioning of the European Union

INTRODUCTION

This dissertation aims at assessing what environmental taxes are. It is argued clear policy guidelines for their design follow from understanding them as regulatory instruments aimed at environmental policy goals. Empirical data show that compliance with such guidelines, which allow the distinction between environmental taxes and environmentally-related taxes, are paramount to the environmental effectiveness of these instruments.

Chapter I presents the main theoretical framework underpinning pollution taxes highlighting the insufficient policy guidance they provide and explaining the conceptual evolution from the economic idea of internalisation of externalities towards environmentally-related taxes that conceptually underpin environmental tax/financial reforms. The argument on which this dissertation builds is next tested against empirical evidence drawn from institutional practices in Denmark, Portugal and Sweden in Chapters II, III and IV, respectively. The Conclusions Chapter builds up into the argument that the empirical evidence supports the conceptual proposal made.

This is a framing chapter aimed at explaining the design features of environmental taxes and at starting to demonstrate how these are scarcely included in institutional practice following a brief analysis of the Energy Taxation Directive (2003/96/EC), which is the basic EU legal framework most relevant to environmental taxation following little EU legislation or harmonization of national tax provisions regarding other polluting bases. This proof shall be strengthened with the empirical data from the national cases presented in Chapters II to IV, an important part of which is influenced by the referred Directive.

This chapter evolves as follows. It starts with the overview of the argument of the dissertation, which is further developed after some conceptual definitions regarding terminology relevant to environmental tax discourse, and follows to briefly analyse the Energy Taxation Directive. Just before the conclusions the thesis overview is presented, after which follow the criteria for the case selection and the methodology used.

1. OVERVIEW OF THE ARGUMENT OF THE DISSERTATION

Both environmental taxes and pollution taxes or environmentally-related taxes are raised on tax bases able to cause directly or indirectly environmental damage and highlight inefficiencies in abatement and opportunities for technological progress by putting a positive price on pollution, hence raising awareness and sharing responsibility. However, they are substantially different. Their underlying normative tax design is different following the different objectives they pursue. Consequently, they lead to different levels of environmental effectiveness.

The more environmentally targeted tax design of environmental taxes makes them perform better than pollution taxes as instruments of environmental policy, producing stronger and quicker environmental effects than environmentally-related taxes raised on the same tax bases. However, following their demanding design requirements, environmental taxes are due to enjoy a less generalised use than pollution taxes. The main theoretical framework underpinning pollution taxation fails to provide guidance to understand such difference.

Environmental taxes are regulatory instruments of environmental policy. As such they aim only or primarily at fulfilling precise environmental objectives via behavioural change and technological progress (adoption and development) and must be ruled by environmental criteria. Their design induces behavioural change by promoting tax awareness and tax avoidance, as well as by adopting a ‘forward looking’ approach provided by its reference to the opportunity for improvement, without leaving essential needs unaddressed or ceasing productive activities, rather than mere pollution amounts. Therefore, environmental taxes must be raised on specific polluting emissions or a proxy for them. Their tax rate needs to be referred to pollution abatement costs or relative polluting impacts taking into account a specific pollutant, and be set at the level required to induce the behavioural change necessary to attain the environmental objective pursued. And they must be charged to polluters who control the cause *sine qua non* of pollution and still did not explore all their opportunities for environmental improvement.

Environmentally-related taxes are revenue raising instruments that pursue cost internalisation among other objectives, including non-environmental ones. Following the price they put on the environment some behavioural change is expected but the tax design is not based on its attainment. To compensate society for external costs caused by polluters the tax rate should be based on the costs of pollution to society and the subjective tax incidence should be as comprehensive as possible ideally covering all the causers of the kind of pollution addressed.

In the taxes analysed different degrees of inclusion of the design features of environmental taxes resulted in different levels of environmental effectiveness. Medium compliance with these guidelines in the Danish waste tax and medium-low compliance in the Swedish CO_2 tax led to strong and fast behavioural change referred only to specific waste streams and in specific sectors, respectively, whereas low compliance with such guidelines in the Portuguese energy tax and the Swedish energy tax resulted in weak environmental results (and sometimes negative environmental results) only visible in the long run. In the Portuguese energy tax and in the Swedish CO_2 tax and energy tax, environmental criteria were little used in the tax design, and tax awareness and tax avoidance as well as precise environmental objectives were absent.

The tax base of the Portuguese energy tax and the Swedish energy tax kept a close linkage to environmental damage but was unable to allow tax payments to mirror the evolution of such damage. This failure was also experienced in the Danish waste tax in some cases due to unaddressed institutional filters that cut the linkage between the tax payment and environmental damages. In all four taxes some polluters able to prevent pollution were sheltered from the price signal provided by the tax and rates were unable to always set environmentally correct *effective pollution prices*. High environmental effectiveness was experienced in the cases where these failures did not occur, namely in the waste tax in some heavy waste streams entailing low recycling costs and in the CO_2 tax in the district heating sector, evidencing the relevance of charging prices able to induce the required behavioural change to polluters able to avoid pollution.

The strong and fast environmental results obtained in the Swedish sulphur tax and NO_x charge, especially in terms of technological progress, were due to good inclusion of the features of environmental taxes in their design. These taxes evidenced the relevance for environmental tax effectiveness of a tax design aimed at precise environmental objectives and led by environmental criteria, without exemptions for big polluters with opportunities for improvement. In both taxes, tax payments mirrored accurately the evolution of the environmental damage, following the tax charge on specific polluting emissions or a good proxy for these at rates based on pollution abatement costs, and reached a level sufficiently high to induce environmentally correct behavioural change.

The comparative analysis of the Swedish sulphur tax and the NO_x charge, on the one hand, and the Danish waste tax, on the other, shows the importance for environmental tax effectiveness of having tax awareness and tax avoidance instituted in the tax design. In the Danish tax, the transference of the tax to institutional practice in waste management by the municipalities eliminated to a great extent the tax awareness and tax avoidance instituted in the tax design by delaying, hiding and flattening the price signal provided by the tax. This led to fiscal illusion and reduced significantly the reward of tax avoidance strategies included in the law with a negative impact on the environmental effectiveness of the tax. Tax awareness was high in the sulphur tax and the NO_x charge due to their incentive to adopt emissions measurement and their targeting at polluters where the financial loss following from the tax was expected to be the highest. The linkage between payment and pollution was further enhanced in the NO_x charge by revenue recycling. The reward of tax avoidance strategies by allowing tax payments based on actual emissions with refunds provided for abatement efforts, together with the high tax awareness, accounted largely for the good environmental performance of the sulphur tax and the NO_x charge.

The Swedish energy taxation system, where the general energy tax partially cancelled the signal provided by the CO_2 tax and the sulphur tax towards the environmental hierarchy of behaviours in some sectors, was especially useful to test the importance of systemic coherence in environmental tax design. Better results in terms of emissions reduction would have been expected if the CO_2 tax and the sulphur tax had been based on environmentally correct relative *effective pollution prices* and compensated reductions in the energy tax. However, the comparative analysis between the Portuguese energy tax and the Swedish energy taxation system showed that the accumulation of environmental taxes and environmentally-related taxes is more beneficial for the environment than the isolated use of general energy taxes. The negative impact on the environmental effectiveness of national energy taxation systems of the misguiding leadership provided by the Energy Taxation Directive (2003/96/EC) is thought to have been increased due to such systematic effect.

Environmentally effective tax designs depend on compliance with environmental criteria, which adds technical complexity to the tax design, and the diversion of environmental tax design and management away from non-environmental concerns. Therefore, environmental tax powers should not be allocated exclusively to the Ministry of Finance. Under an institutional design where the Ministry of Environment plays a dominant role caution is still required to avoid bias towards revenue raising (fiscal bias).

Both taxes with a more technically demanding design, performing better from an environmental point of view and with low revenue stream, namely the Swedish sulphur tax and the NO_x charge, were designed by the Ministry of the Environment. The taxes presenting the tax design closer to the traditional excise duties, including just a few design features of environmental taxes and performing well as revenue raisers, namely the Portuguese energy tax and the Swedish CO_2 tax and energy tax, were within the ambit of the Ministry of Finance.

The need to divert the allocation of environmental tax powers away from non-environmental interests to avoid fiscal bias in the tax design was especially evident in the Danish waste. It is reasonable to expect that the tax design would have been made more environmentally effective if the entity designing and managing the tax, i.e. the Ministry of Environment until 1993 and the Ministry of Finance from then onwards, had not kept revenue interests in its tax base. The same reasoning applies to the entity controlling *de facto* the application of the tax in its transference to institutional practice, i.e. the municipalities, who kept a desire to fully exploit available capacity in public infrastructures dealing with waste.

2. Preliminary conceptual analysis of environmental taxes

Environmental taxes are understood as policy instruments aimed at conditioning the behaviour of economic agents towards more environmentally sustainable patterns. The attainment of this effect depends on very precise design features, which are seldom found in institutional practice, and in the absence of which taxes raised on polluting bases are not able to deliver relevant environmental results in the short to medium term, i.e. approximately within the five years following the adoption of the tax, but only revenues. As regulatory instruments of environmental policy, the effectiveness of environmental taxes must be assessed against precise environmental goals. Such effectiveness, though not limited by cost-benefit analysis, is constrained by the price system according to which they work and enhanced by competitive market structures following such dependence on the price system.

2.1 Environmental taxes are regulatory instruments aimed at environmental policy goals

As instruments of environmental policy environmental taxes are embedded with a regulatory nature (Hoerner, 1998: 11) aimed at protecting the quality of the natural environment. In the sense that they perform public control over private sector behaviour they have a regulatory nature (Vogel, 1996: 9; Ogus, 2004: 2) as opposed to a fiscal nature (i.e. orientation towards revenue raising). Regarding environmental policy, regulatory instruments are “institutional measures aimed at directly influencing the environmental performance of polluters by regulating processes or products used, by limiting the discharge of certain pollutants, and/or by restricting activities to certain times, areas, etc.” (Opschoor and Vos, 1989: 12).

‘Policy’ is a purposeful concept in the sense that the fulfilment of a conscious purpose is essential to it. It is “the content of policies (the paradigms of action, the objectives and the policy instruments), the legal and administrative structures that have been established to oversee them, and the dominant style in which policy is made and implemented” (Jordan and Liefferink, 2004: 1-2). Therefore, goals must be pervasive to environmental taxes as they are to environmental policy as ‘policy’ (“a coherent and consistent number of actions, grouped under a general orientation and with the aim of reaching certain objectives", Krämer, 1995: 44).

As instruments of environmental policy these taxes are part of the "courses of action to regulate polluting activities, to regulate the occupation of space, and to regulate the extraction of raw materials, all with the purpose to prevent the deterioration of, to maintain, or to improve, the quality of the natural environment" (Lundqvist, 1996a: 16; Jans and Scott, 2003: 325), i.e. courses of action aimed at environmental goals (for example, "contribute to preserving, protecting and improving the quality of the environment, protecting human health and prudent and rational utilisation of natural resources" – Article 191 (1) TFEU).

Environmental taxes must prioritise anticipatory action, since a touchstone of environmental policy at the European Union level is anticipatory action. This has been prioritised over remediation since the first European Community environmental action programme (1973). Preference for prevention over remediation is not only required by commonly agreed content of the polluter pays principle, the principle that environmental damage should as a priority be rectified at source, the prevention principle and the precautionary principle, but it is also a principle of EU environmental law and policy in itself (Art. 191 (2) TFEU). Principle 15 of the Declaration of the United Nations Conference on the Human Environment (Rio de Janeiro, 1992) also refers to the 'prevention principle'.

These instruments must accurately communicate the environmental hierarchy providing incentives to switch to less environmentally damaging behaviours (OECD, 2001g: 45-46, 112). As economic instruments, they stimulate response to price signals through reduction of overall consumption of the good, substitution of an environmentally harmful version of the good towards a less environmentally harmful one (Ekins and Dresner, 2004:2) or adoption of clean technology. The latter effect induces increased technological development in the industry (dynamic efficiency effect).

As regulatory market-based instruments environmental taxes have their regime mainly set by environmental law, as the branch of law comprising norms directly aimed at the maintenance and protection of the natural environment, and economic law rather than tax law. Whilst the latter sets the legal system that governs revenue collection by public authorities, economic law rules the attainment of public goals through regulation of economic activities. This clarification is relevant in countries where compliance of fiscal taxes with the constitutional principle of equality is assessed via the principle of the ability to pay, as it is the case for instance in Portugal (Article 4 (1) Law 41/98, 4 August 1998).

In such countries environmental taxes do not undergo the constitutionality test with reference to a strict correspondence between the tax payment and the ability to pay of the payer. Though they cannot be charged on those who do not enjoy the ability to pay, their measure is referred to the proportionality (i.e. proportionality *stricto sensu*, adequacy and necessity) between the tax payment and the regulatory goal. Furthermore, strict compliance with the principle of legality that lays the disciple (as detailed as possible) of the main elements of the tax (i.e. tax rates, tax base, tax subjects, as well as taxpayer rights and tax benefits) in the competence of the parliament is imposed on fiscal taxes but not on non-fiscal (i.e. regulatory) taxes as environmental taxes are. Regulatory effectiveness requires speedy and periodical reaction to the evolution of the regulated situations. Therefore, the public administration enjoys more discretionary power whilst managing environmental taxes than fiscal taxes.

As economic instruments of environmental policy, environmental taxes are defined by their effects on costs and benefits of alternative actions with the effect of influencing behaviour in an environmentally favourable way (OECD, 1991b: 10). Though there might be a number of ways in which different actors evaluate environmental taxes (for instance, exchequers might evaluate their revenue raising capacity whilst families assess their equity), assessing the effectiveness of environmental taxes as instruments of environmental policy refers to the causal link between policy action and its intended impact on human behaviour and the environment, i.e. the comparison of actual effects with desired environmental objectives. This is different from evaluating their cost-effectiveness, which requires comparing the effects of a measure with the costs of implementing it (EEA, 2001: 9).

There is possibly a wide variation in the grade of linkage of environmentally-related taxes to environmental protection. There are several taxes without any direct linkage to the protection of the environment and nature, but still with some kind of environmental impact (SEPA, 1995: 8). The qualification of a tax as environmental follows from the assessment of its direct causal relationship with the fulfilment of environmental goals. The latter follows from the tax design according to precise environmental criteria and goals undisturbed by other kinds of interest. This is attained with a tax payment mirroring the evolution of the environmental disruption caused in the amount necessary to steer behaviours in the desired direction and imposed on those able to avoid pollution.

Following the analysis of institutional practices in pollution taxation in the selected country case studies, it is argued that environmental taxes are instruments of environmental policy with tax awareness and avoidance instituted in their design and aimed at preventing environmental damage via increased awareness and shared responsibility and targeted at individuals able to perform it. Environmentally-related taxes fulfil this predominantly preventative function when designed as regulatory instruments with a 'forward looking' approach expressed in the reference to opportunity for improvement rather than pollution amounts.

Fiscally biased taxes raised on polluting bases, though with a more generalised use due to the less demanding tax design and mix of objectives pursued, are able to share responsibility but perform poorer behavioural steering (i.e. less strong and at a lower pace) than regulatory taxes. This is so since increased tax awareness over the price signal and tax avoidance strategies are neglected in their design for impacting negatively on the instrument revenue raise capacity. Such environmentally-related taxes are suited for damage restoration and compensation but not for damage prevention.

Different levels of environmental effectiveness follow from regulatory taxes (environmental taxes) and fiscal taxes (environmentally-related taxes or pollution taxes)

The environmentally-related tax design can aim at either raising revenues, in which case a fiscal rationale underpins the tax, or changing behaviours towards less polluting patterns and clean technological progress (adoption and development), in which case the tax is ruled by a regulatory rationale. Taxes led by a fiscal rationale and aiming at revenues to be applied in environmental programmes fulfil environmental goals only indirectly. In such cases environmental goals are directly fulfilled through the measures included in those programmes rather than the financing instrument itself, regardless of the latter's fulfilment of the shared responsibility proposed by the polluter pays principle.

Taxes only achieve environmental goals directly when their design is regulatory. Only then is it the charge of the tax itself that prevents the deterioration of, and maintains or improves the quality of the natural environment, by steering behaviours towards less environmentally damaging patterns whilst dissuading pollution through price signals. Regulatory taxes perform predominantly preventative action, leading to behavioural conditioning and technological progress that are *conditio sine qua non* for prevention. Remediation through regulatory tax measures occurs indirectly when environmental investment is stimulated by the non-charge of the tax. This can only be done marginally within a tax regime via tax benefits, which are exceptions to the tax rule.

Environmental taxes cannot be underpinned by revenue-oriented theories, for example the double dividend argument, since their design must be anchored to specific environmental objectives. The double dividend argument suggests that the use of environmental taxes can create an environmental dividend (first dividend) as well as an economic dividend (second dividend). Behavioural change induced by such taxes would produce environmental benefits and the application of their revenues in the reduction of pre-existent tax distortions would, in turn, yield an economic efficiency gain (see Section 1.4 of Chapter I).

However, there is a contradiction between technological flexibility and product substitutability, both required for behavioural change, on the one hand, and fiscal objectives, on the other, since those two variables lead to the erosion of the tax base with the consequent loss of revenue. Therefore, a fiscal rational diverts the tax design away from regulatory features able to prevent pollution. Even when there is a close linkage between the tax base and the environmental impact, as for instance in the case of a tax raised on energy products, environmental results and fiscal outcomes follow from different kinds of tax design.

Fiscal concerns advocate tax systems that are as neutral as possible, not affecting the decisions of the economic agents, by broadening tax bases, flattening rate structures and integrating or aligning different tax rate structures to avoid arbitrage opportunities (OECD, 2003c: 17). Environmental effectiveness, on the contrary, requires tax designs that lead economic choices by promoting tax awareness and tax avoidance.

Therefore, the OECD definition of environmentally-related tax ('any compulsory, unrequited payment to general government levied on tax-bases deemed to be of particular environmental relevance'[1]) does not fully overlap with that of environmental tax, since an environmentally-related tax can comprise either a fiscal rationale or a regulatory rationale.[2] Alternatively to 'environmentally-related tax' the expression 'pollution tax' can also be used when the tax base refers to polluting substances.

Despite the normative distinction between fiscal and regulatory pollution taxes and the potentially better environmental results following from the latter, environmental taxes are not common institutional practice, neither at a national level, as follows from the empirical data collected in the three case studies presented in Chapters II to IV, nor at the European Union level, as follows from the analysis of the European Taxation Directive briefly presented in Section 4 of this chapter. In the absence of theoretical guidance to set the environmental tax design, in practice pollution taxes got pushed towards the interests of the policy-entrepreneurs. The ministry of finance often pushed them through the budget process conditioned on their fiscal bias.

Signs of fiscal bias spread throughout the design of environmentally-related taxes include price inelastic tax bases with an indirect connection with the environmental damage instead of products for which there are cleaner substitutes and with a direct causal relationship with environmental damage or a close proxy for the latter. Regarding the tax rate, revenue interests have been nurtured through tax rates too low to push behaviours towards more environmentally sound practices. As far as the subjective tax incidence is concerned, the focus has been on the smaller rather than the larger polluters or on the politically less powerful groups rather than those best able to control pollution. Favouring one group vis-à-vis the other has often been done either through the granting of substantial exemptions or the setup of low tax rates.

In the medium and long term, this fiscal bias of pollution taxes can threaten the credibility of environmental taxes. There is a connection between the acceptance of the instrument and its successful design (Santos *et al*, 1999: 451). Just as proofs of successes of policy instruments can lead to increased awareness of such tools in policy in general (Hahn, 2000: 394; Speck and Ekins, 2002: 88), disappointed expectations create obstacles in the form of lost credibility that can have long-term political ramifications (Milne, 2003: 25).[3] For the instrument's acceptance and effectiveness, the connection between policy action and results has to be clearly acknowledged (Meade, 1978: 19; OECD, 1991a: 24; 'transparency principle' – Määttä, 1997: 39).

Critical appraisal of the concept 'environmental tax' in the literature

The distinction between 'pollution taxes' or 'environmentally-related taxes', such as taxes raised on polluting bases, and 'environmental taxes', such as instruments of environmental policy, is not clearly set neither in the literature nor in the political discourse (Opschoor and Vos, 1989: 18-19; Hertzog, 1991: 17; OECD, 1994b: 18; Määttä, 1997: 49). In the absence of a commonly agreed definition of 'environmental tax' (SEPA, 1995: 8; OECD, 1997c: 29), a conceptual benchmark to guide policy making is missing.

Different criteria have been used to define environmental taxes. The most frequent ones are the design of the tax itself (with emphasis on the polluting characteristics of the tax base or the use of tax revenues)[4]

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[1] OECD/EU database on environmentally-related taxes, available on: www.oecd.org/env/tax-database.

[2] Joint statement by the Austrian, Danish, Dutch, Finnish and Swedish delegations on draft annotated outline for a Communication on the use of market-based instruments for environmentally-related policy in the internal market (June, 2004).

[3] ENDS 12 October 2002, quoting Hans Christian Schmidt (Danish Minister of the Environment) at EEB Conference, Brussels, 10 October 2002: "We put up taxes, they changed their behaviour, and we put them up again. They don't believe us any more!"; *Advance for Britain's green fiscal framework*, ENDS 28 November 2002; phone interview with Sven Gunnar Persson, Representative for the environment of the Swedish Christian Democratic Party, 15 September 2003, commenting on the Swedish 'environmental tax reform': "There has not been so much compensation for 'green' but extra burden on 'non-green'".

[4] Documents defining environmental taxes based on the characteristics of the tax base or the use of tax revenues includes for example COM(1997) 9 final, 26 March 1997, 4, and Eurostat, ATW-Research, 1996a:Vol. 2, 5-6;

and the *ratio* underpinning the instrument.[5] Most recurrent defining characteristics are associated with the understanding of environmental taxes as environmental quality improving instruments which use price signals to attain their goals. This logic has its roots in Pigouvian theory and played a strong influence in the polluter pays principle (see Section 1.2 of Chapter I). It is based on the belief that internalisation of external environmental costs is able to bring polluters to consider the complete economic consequences of their actions and avoid such costs by preventing pollution. However, this definitional element, i.e. 'prevention', is not frequently found in institutional practice.

Environmental economic instruments in general are said to have broader functions than environmental protection and their use is considered relevant in administrative, political and constitutional terms as well as environmental ones (Elworthy and Holder, 1997: 306). Environmentally-related taxes tend to be associated with gains in mainly four key areas: environment, innovation and competitiveness, employment and the tax system. Often environmental taxes are described amorphously and include simultaneously multiple objectives (EEA, 1996: 7-8; Hoerner, 1998: 11; Ribeiro *et al*, 1999: 181-183; Holzinger, 2003: 189). Fiscal and environmental concerns are often simultaneously used to explain their rationale (even when the potential conflict between these two goals in the long-run is acknowledged – OECD, 2001g: 39)[6], whereas little literature on environmentally-related taxes contrasts fiscal taxes to regulatory taxes.[7]

Objectives attributed to pollution taxes are not always fully compatible and cannot therefore be simultaneously achieved (*e.g.* Speck and Ekins, 2002: 100). However, this is not broadly acknowledged in the literature. Confusion between fiscal and regulatory goals is more frequent regarding taxes which can affect simultaneously various aspects, like energy taxes (Brännlund and Kriström, 1999b: 16). These taxes have traditionally been aimed at raising revenue, with regulatory components, especially environmentally-related ones, making their way through their design only recently (IEA, 1993: 15; Määttä, 1997: 219).

2.2 Environmental tax effectiveness must be assessed against precise environmental goals

Successful use of regulatory instruments involves an informed choice between different kinds of instruments according to the objective pursued (Pildes and Sunstein, 1995: 52) and a clear communication strategy. Environmentally-related taxes often miss precise environmental objectives and consequently fail to deliver relevant environmental effects, though they are still communicated as environmental tools ('environmental taxes', *e.g.* OECD, 1998: 30). This raises concerns since an ineffective regulatory intervention and a unsuitable communication strategy can involve important credibility costs (political implications) and efficiency costs (policy implications). Moreover, the absence of precise environmental objectives flaws judgement on whether the expected objectives of the policy measure have been achieved (effectiveness assessment).

As instruments of environmental policy, environmental taxes require precise objectives. 'To protect the quality of the natural environment' is a broad statement served by more or less detailed goals. In the effectiveness assessment process, the environmental objective used as a reference should be the one stated by the legislator. The distinction between objective elements defining the tax and the legislator's words ('explicit versus actual goals' – Määttä, 1997: 9-10; Messerschmidt, 1986: 124-125; Herrera Molina, 2000: 56) is already part of the effectiveness assessment process to check whether a certain legislative goal got lost whilst designing the tax (Hansmeyer, 1989: 50, 55, 59; Herrera Molina, 2000: 56; Barde and Braathen, 2005: 121). This does not mean that a tax that proved unable to deliver the objective stated by the legislator is environmentally ineffective. A tax is environmentally effective as long as it delivers clear and measurable environmental improvement.

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whereas literature with that same approach includes *inter alia* Osculati, 1979:1; Messerschmidt, 1986:27; Buñuel González and Herrera Molina, 2003:9.

5 For definitions of environmental taxes based on their rationale underpinning see for example Carbajo Vasco, 1995:977; EEA, 1996:13; Paulus, 1995:51-52; Hoerner, 1998:11; Kluth, 1998:191; Brännlund and Kriström, 1999b:30.

6 See for example European Commission, 1975: point 4, and 1996: point 5.2.; Osculati, 1979: 34-35; Hendler, 1990:581-585; Maglia and Santoloci, 1996: 2111; OECD, 2001g: 24.

7 See for example Lodin, 1995: 4; Paulus, 1995: 51-52; Määtä, 1997: 210; Kluth, 1998: 191; Herrera Molina, 2000: 56.

The environmental effectiveness of a tax is measured as the extent to which the tax delivers environmental improvement in terms of physical results, since society expects, "above all, physical environmental results from environmental policies" (Dietz and Vollebergh, 1999: 346, quoting de Savornin Lohman, 1994). The parameter to assess such effectiveness should be a behavioural response following from price incentives rather than revenues raised: "[i]nformation on the magnitude of the revenue from environmentally-related taxes should not be used to draw any inference about the 'environmentally friendliness' of the tax system (…) without more detailed information" (OECD, 2001f: 2-3). The methodology to use in such assessment should be the one usually employed to assess the effectiveness of regulatory instruments, i.e. predict what behaviour would be if the regulation in question were not present and then compare this behaviour with actual behaviour, the difference between the two being a measure of how regulation effects (or changes) behaviour (Edwards and Edwards, 1974: 445).

The effectiveness assessment cannot be based on mere comparison between the pollution price previous to the tax and the one following its introduction. After introducing a pollution tax there is often a positive environmental change compared to business-as-usual due to price mechanisms. However, this is not always the case. Behavioural changes even when started by price corrections (internalisation of externalities) depend on several other aspects such as for example the existence of substitutes for the polluting good (especially if it is used to satisfy a basic need) and relative prices (i.e. a tax rate level high enough to make pollution more expensive than its avoidance), as well as the perception by the taxpayer of a clear linkage between the tax payment and the environmental damage (tax awareness).

A tax shift from environmental goods into environmental harms might not be enough to get environmentally correct behavioural changes. For instance, if despite the tax shift the disposable income is not reduced and there are obstacles to a more friendly environmental behaviour, then the level of environmental harm attached to consumption will remain unchanged due to unchanged habits. For example, this might be the case when the market fails to visibly transmit the price signal provided by the tax to the ones able to prevent environmental damage. The introduction of environmental taxes needs to be accompanied by the removal of institutional filters with this kind of impact.

2.3 Environmental tax effectiveness is limited by the price system but not by cost-benefit analysis

Environmental taxes as regulatory instruments have their effectiveness limited by the price system according to which they work but not by cost-benefit analysis, in contrast to tax instruments that pursue cost-internalisation. These are not adequate to protect every kind of environmental element, since they work within the realms of a cost-benefit approach, which is unable to embrace all environmental values. They are suited to prevent damage to environmental elements assessed via traditional economic analysis. Many benefits of public health and environmental protection have not been quantified and cannot easily be quantified given the limits on time and resources, such as outcomes occurring far in the future.

Cost-benefit analysis seeks to translate all relevant considerations into monetary terms, which requires reducing life, health, and the natural world to monetary values providing (shadow) prices but not values (Heinzerling and Ackerman, 2002: 13). Environmental goods are valued according to their value of exchange, which is taken as their value of use (Schmitt, 1985: 49). This process is inherently flawed. Since there are no natural prices for a healthy environment, cost-benefit analysis requires the creation of artificial prices. And these can hardly express accurately the benefits of preserving priceless things such as for instance long life and good health (Heinzerling and Ackerman, 2002: 13). Furthermore, cost-benefit analysis calculates a value only for 'statistical' life and ignores life itself (ibidem).

Cost-benefit analysis involves the devaluation of future events through discounting. The use of discounting improperly trivializes future harms and the irreversibility of some environmental problems (ibidem). Therefore, highly damaging pollution whose effects only occur in the far future (for example due to an accumulation of persistent emissions over a long period of time) should be addressed through other instruments rather than taxes. The impact of environmental disruption on future generations and other living beings, as well as decisions about who should own a property right over the environment and whether environmental costs should be measured by an individual's willingness to pay cannot be dealt with via traditional economic analysis.

Environmental taxes as regulatory instruments are not more constrained by the limitations of cost-benefit analysis than traditional environmental policy instruments, since their amount is not based on the estimated costs of the actual or potential environmental damage. In the environmental tax process, a quantitative environmental objective must be first decided likewise it would have been if the instruments to use were prohibitions or administrative controls. Only after such an objective has been set can the tax rate be fixed. The amount of the latter refers to the pollution abatement costs or relative prices that allow the attainment of the environmental objective previously decided. Relative prices should be based on the environmental hierarchy of consumption (i.e. relative polluting impacts).

However, environmental taxes cannot guarantee a certain amount of pollution control. They are price signals that can only guarantee the cost of pollution control. Therefore, if there is a risk of irreversible damage, either due to a permanent loss of the renewability of natural resources or balance of ecosystems, public regulation cannot be through taxes. The time span of the reaction to taxes makes these instruments also unsuitable to address cases when sudden changes in the environment may cause irreparable damage. If for example it has been scientifically proven certain emissions cause cancer, taxes are not suitable policy instruments to address such kind of pollution.

Likewise their use is not recommended regarding polluting emissions with persistent effects or made of several substances, since these conditions add considerable complexity and uncertainty to such use. For the same reasons, if damage following from polluting emissions (e.g. persistent emissions or emissions concentrated in time or space) follows a non-linear pattern and becomes irreparable after a certain point, as well as when the same quantity of emissions may cause different marginal damage some of which irreparable due to aleatory factors (e.g. the direction of the wind), the use of taxes is also not recommended.

2.4 Competitive markets enhance environmental tax effectiveness

Environmental taxes work within a system of price signals. Therefore, the less distorted the system of price signals is, i.e. the closest it resembles a perfectly competitive market, the more effective these instruments are. Perfectly competitive markets are described as ones where no participants are large enough to have the market power to set the price of a homogeneous product. This requires a large number of buyers and sellers, absence of transaction costs and entry and exit barriers, perfect information and a profit maximization rationale (Hanley et al, 1997: 22). This rationale leads companies towards cost-minimization. Under competitive market conditions polluters will reduce emissions while the marginal abatement cost is lower than the tax level.

In institutional settings with low responsiveness to market prices the price signal provided by the tax will tend to be filtered and there is hence expected a low behavioural impact following from environmental taxes. The possibility polluting producers enjoy in the presence of monopoly markets or economies of scale of passing forward to consumers the increased costs imposed on them by the tax may shun their reaction to environmental taxes (Oates and Strassmann, 1984: 32). For instance public companies, which often operate in concentrated markets (monopolies or oligopolies) and are not run according to a profit maximization rationale, may react poorly to price signals provided by taxes, unless these are accompanied by other kind of measures (e.g. instructions to managers).

Since competitive markets are the institutional processes most suitable for environmental tax effectiveness, tax designs impairing competition, such as the ones sheltering some economic agents from the price signal, might harm environmental effectiveness. Exemptions or tax reductions for some polluters not only raise issues of environmental effectiveness and equity regarding the charged industry, but may also hamper the environmental effectiveness of the tax in other industries.

If a maximum ceiling is set for the market price for polluting goods, the market for clean products is constrained. This problem is increased if companies enjoy tax benefits regarding the polluting emissions they produce following their consumption. Companies producing those goods will miss an interest in investing in clean technology since they might be unable to internalise the positive externalities thereof through the sale of clean goods due to the market price restriction applied to polluting goods that shelters the latter from competition.

3. The Design Features of Environmental Taxes

To steer towards less environmentally damaging behaviours through price signals, environmental taxes must have their tax design led by environmental criteria and precise environmental goals and refer to improvement potential rather than mere pollution amounts. Environmental taxes cannot result from the mere application of environmental economics to environmental objectives following from political compromises. For example, a tax that achieves an emission reduction target politically agreed that according to state of the art scientific knowledge is unable to avoid environmental damage cannot be considered environmentally effective.

The 'forward looking' at polluting impacts results from the institution of tax awareness and avoidance in the tax design and having specific polluting emissions or a good proxy for them as tax base and setting the tax rate according to abatement pollution costs or relative polluting impacts taking into account a specific pollutant in order to guide behaviours towards the environmental hierarchy of consumption. Furthermore, the tax payment should be imposed on the group of polluters able to avoid pollution. To get institutional practice to mirror this normative design it is useful to assign environmental tax powers to entities with environmental expertise and not holding an interest in the tax base.

3.1 Environmental taxation should be ruled by environmental criteria and aimed at precise environmental objectives

Environmental taxes as instruments of environmental policy hold a direct causal relationship with the fulfilment of environmental goals. Environmental tax design cannot be the mere application of environmental economics to environmental objectives politically decided. In environmental taxation the objectives and the design have to abide by environmental criteria rather than political compromises. This requires the presence of science at the centre of the environmental tax process. The absence of clear and precise environmental goals at the outset of the environmental tax process prejudices the reference of the tax design to scientific data, misguiding it towards political compromises and away from environmental effectiveness, as well as the reference of all the design features of the tax to one same purpose enhancing the systematic coherence of the tax and consequently its environmental effectiveness.

The earth can be understood as a series of sources from which resources are drawn, namely soil, water, biotic resources (forest and species diversity) and fuel and nonfuel mineral resources, and sinks, into which human waste and effluent goes, namely water, atmosphere and soil (Harper, 1995: 72). Waste and effluent can change the environment, hindering (temporarily or permanently) its balance and renewability, damaging natural resources and sinks. This damage caused to natural resources and sinks are understood as pollution (Stiglitz, 1993: 643).

The minimum level of pollution control should be set with reference to the concept of sustainable development, which is *jus cogens* and has been adopted as a legally binding national goal in most countries. Sustainable development requires prevention or abatement of any damage threatening the balance of the ecosystem, in the sense that economic growth cannot cost a reduction in well-being. This is assessed by reference to the potential consumption per inhabitant, which is conditioned by the production potential in the future and, consequently, by the whole amount of capital available (i.e. natural resources and human-made resources). Human, social and natural capital left to future generations has to allow them at least the same opportunities and potential consumption per inhabitant enjoyed by present generations. These should leave to future generations a stock of natural capital equivalent to what it took over, and sustain its consumption only on the dividend (Pearce, 1989).

The targets pursued with environmental taxes reflect a sustainability criteria if fixed on the basis of the carrying capacity of the environment. For example, the waste tax design should promote waste disposal according to the waste hierarchy, favouring reductions in waste amounts vis-à-vis recycling and the latter vis-à-vis waste disposal. A tax design that favours incineration over recycling stays short of complying with environmental policy objectives regardless of the adoption of such waste hierarchy as a policy goal following non-environmental interests and a genuine political process.

Once the environmental target has been set, the tax should be designed in order to allow its attainment taking into account scientific data available. For instance, to reach the target decided the pollution flux in a certain industry (i.e. amount of emissions of certain substances per unit of time) must be regulated taking into account the carrying capacity of the environment and the expected environmental damage following from non-residual emissions. If there is scientific uncertainty about the carrying capacity of the environment the precautionary principle should be applied.

The single purpose should be the rule since the environmental hierarchy of behaviours tends to vary depending on the kind of pollutant considered. Therefore, to base the tax rate on relative polluting impacts it is necessary to take into account a specific pollutant which the tax aims at preventing or abating. This pollutant should be the same as that considered in the tax base. To pursue simultaneously several objectives with one same tax is feasible when they are all served by the same environmental hierarchy.

If one same legal instrument aims simultaneously at several objectives making use of market segmentation, in the sense that two or more separate blocks of regulation are created, with different objectives, rates and products, in fact there will be more than one economic instrument. For example, an energy tax can be designed in such a way that it pursues reduction in sulphur emissions in motor fuel consumption and CO_2 emissions reduction in industrial fuel consumption, leading to two different environmental hierarchies of consumption. This design should only be used when there is perfect market segmentation according to products. Otherwise, i.e. if for example one same product is consumed both as motor fuel and as industrial fuel, the tax might end up communicating (through the tax rate) an environmentally wrong relative position of such a product in one of the hierarchies.

3.2 The environmental tax design must be referred to improvement potentials rather than pollution amounts

Under a 'forward looking' approach, environmental taxes should be imposed according to consumption efficiency and not mere absolute values of consumption, being financial penalisation assigned not for using natural resources but to avoid their "environmentally-harmful squandering" (OECD, 2003b: 35). The environmental tax design must acknowledge that there are indispensable activities that inevitably produce emissions, but there is also some scope for discretionary choice regarding the choice between substitutes, as well as luxury emissions (from consumers' perspective) and inefficient production (from producers' perspective) (Kallbekken, 2008: 19).

The regulatory rational is embedded in the tax design by referring it to the opportunity for improvement rather than to mere pollution amounts. To steer behaviours towards pollution prevention, tax awareness and tax avoidance should be instituted in the environmental tax design. Tax payments should follow from tax bases that are specific polluting emissions or close proxies for them and tax rates referred to pollution abatement costs or relative polluting impacts taking into account a specific pollutant leading to environmentally correct relative *effective pollution prices*.

Such price signals should reach the ones able to control pollution that still have not explored all their opportunities for improvement. Different tax treatment for polluters causing the same unit of pollution is acceptable only if based on factors that are under individual control (discretionary input to creating the externalities). Therefore, assuming the same amount of pollution, selective subjective tax incidence and more favourable regimes can only be based on criteria affecting the level of control over pollution, such as technological aspects, and not on the amount of pollution.

3.2.1 Tax awareness and tax avoidance should be instituted in the environmental tax design

Since environmental taxes work through price signals providing financial incentives towards environmentally correct behavioural change, their effectiveness depends on taxpayers' awareness of such signals and understanding of how they work. For polluters able to avoid pollution to change their behaviour they need to realise the financial cost involved in the tax payment, understand it as avoidable and comprehend how it is possible to avoid it.

Tax awareness can be instituted in the tax design by making clear to taxpayers how much they pay. Design that hides the tax payment in another price, spreads it over time or delays it should be avoided for leading to fiscal illusion (Puviani, 1972: 41-42). Tax awareness also requires that polluters understand why they pay. They should be able to understand the linkage between the tax payment and the environmental damage. This can be clarified, for example by electing as tax event a polluting action. Furthermore, the polluters able to prevent pollution should be those actually paying, rather than transferring the tax burden to others. Taxpayers' awareness of pollution costs is further increased by informing them of the tax revenues collected.

Polluters must also be aware of the possibility of avoiding or reducing the tax payment by adopting less environmentally harmful options. Presenting environmental taxes as avoidable enhances their effectiveness (OECD, 2003b: 6). The environmental tax effectiveness depends on the effort the tax subjects put into looking for alternatives to the tax payment. There is a high probability of having a positive tax driven behavioural change when the regulated aspects belong to the core business of the company. This is so because in such cases both the financial impact of the charge and awareness about it by the management will be high. The adoption of tax avoidance strategies is the expected reaction of a cost-minimising company following such high impact and awareness.

Environmentally desirable tax avoidance is instituted in the tax design by referring it to the opportunity for improvement rather than mere pollution amounts and keeping a close linkage between the tax payment and the tax base (and consequently also the environmental damages). The first is done by setting the responsibility cut according to the variables that are under the discretionary control of the polluter. The polluters are only held responsible for the emissions they can avoid without leaving essential needs unaddressed (differentiating between subsistence emissions and luxury emissions) or ceasing productive activities (taking into consideration alternative technologies, including BATNEC, and processes). Making polluters pay according to their capacity to avoid environmental damage and the actual damage they cause allows and induces tax avoidance strategies that correspond to pollution prevention.

Tax avoidance should be allowed and induced in the tax design, which requires narrow tax bases, comprising goods with a polluting impact that significantly affects the wellbeing of society and for which there are cleaner alternatives (developed or potential). This allows taxpayers to locate their behaviour outside the tax base following the adoption of more sustainable behaviours. In economically rational individuals increased cost-awareness is likely to lead to the adoption of tax avoidance strategies. Therefore, tax rates should be sufficiently high to raise cost-awareness. Cleaner alternatives to paying the tax and/or strategies to lower the tax payment should be available and signalled in the tax design, for example through refunds and tax rate differentiation.

Instituting tax avoidance in the tax design is likely to enhance not only the environmental effectiveness of the tax but also its feasibility, since people oppose taxes that they perceive as unfair. Taxes raised on consumption with an environmental impact impossible to avoid have been perceived as unfair and opposed (OECD, 2003b: 84). For instance, charges on unmetered environmental damage or only slightly related to the amount of environmental harm caused by the taxpayer tend to be unpopular. For instance, Irish domestic water charges were abolished due to such opposition (OECD, 2003b: 85).

3.2.2 Tax payments must take a 'forward looking' approach at polluting impacts

The tax base is used to measure the tax payment obligation laid on each taxpayer following the tax event (Martín Queralt and Lozano Serrano, 1994: 330), the tax fairness and effectiveness depending on how well the tax base reflects the taxpayers' participation in the tax event (Cortés Domínguez, 1965: 1042). The tax linkage defines the degree of connection between the tax base and the tax goal, presenting a trade-off between tax effectiveness and administrative costs (Smith, 1995: 10). It might evolve along the life of the tax (OECD, 1993: 53) and be made clearer for the taxpayer via specific tax revenue allocation.

In environmental taxes, the tax base measures the environmental damage caused by the taxpayer. Private benefits from pollution tend to be proportional to the amount of pollution caused. Economically rational individuals decide on pollution by comparing the private benefits thereof plus the tax cost with the pollution abatement costs. Therefore, the price signal transmitted by the tax needs to be proportional to the amount of pollution to be effective in leading the taxpayer towards pollution prevention.

However, the intensity of the price signal should not express the cost caused to society by the pollution, but the level required to achieve the environmental objective pursued. This is attained by applying the tax rate to the tax base. The tax rate, set according to pollution abatement costs or relative polluting impacts taking into account a specific pollutant in the level required to attain the environmental goal pursued, measures the intensity of the regulatory intervention. This must be levelled according to the precise environmental objective pursued taking into account the price-elasticity of the tax base and relative *effective pollution prices*.

A. The tax base should be specific polluting emissions or a proxy for them

The introduction of the tax aims at manipulating the amount of the tax base via a price increase. Only if the amount of environmental damage proportionally follows from the variation of the tax base, acting on the latter is equivalent to acting on the previous, since a change in the price of the tax base will equivalently transfer into a price change in the environmental damage. The polluting demand will react to the new price following from the regulatory intervention either whether this targets the price of the tax base or directly the price of the environmental damage. Therefore, environmental tax bases must be positively correlated with the environmental damage, being specific polluting emissions themselves or a good proxy for them.

For the tax payment to vary according to the environmental damage caused, as required not only by damage prevention but also by equity concerns, the tax base should be the specific polluting emissions directly leading to the environmental damage addressed by the tax and the taxable event should be the behaviour causing them. The selection of a tax base corresponding to specific polluting emissions is required for successful behavioural steering, since the latter requires precise leadership on which is the environmental hierarchy of behaviours those regulated should follow and the prevention of different pollutants tends to be served by different hierarchies. If the tax payment does not refer to the direct cause of a specific environmental damage it must be to a close proxy for the latter. Hence, a second best solution is a tax base accurately mirroring the evolution of the environmental damage produced, i.e. which variation is directly linked to (i. e. positively correlated with) the amount of damage produced. General energy taxes cannot abide by this demanding design.

A positive correlation occurs when (1) there is a strong relationship between the two variables, i.e. a change in the size of the tax base directly leads to a change in the amount of environmental damage and vice versa and (2) there is a positive association, i.e. both items change in the same direction (they both either increase or decrease) (Babbies et al, 2003: 260). The closest such correlation is from the unit the more accurately the tax base mirrors the evolution of the environmental damage, in the sense that a unitary reduction of the environmental damage is likely to follow from the reduction of the tax base in one unit. In the case of a positive correlation the taxable event should be the one directly affecting the size of the tax base (e.g. if the tax base is vehicle kilometrage, the taxable event should be the act of driving the vehicle and not the vehicle ownership).

Some polluting impacts can be better addressed by proxies than others. For example, fossil fuel consumption is a good proxy for the amount of CO_2 emissions disrupting the environment. Since an effective scrub technology is not available yet, the amount of CO_2 emissions released following fossil fuel consumption is directly related to the CO_2 content of the fuel itself. To raise the price of fossil fuels (e.g. via the introduction of a tax) will then also proportionally increase the price of CO_2 emissions. And the reduction in fuel consumption following from a price rise is likely to be mirrored by a reduction in CO_2 emissions. The same is not true regarding NO_x emissions since these depend on the combustion technology used.

If emissions are produced along a complex process depending on several variables, the tax payment should be based on the application of the tax rate to measured emissions to guarantee that the price signal provided is proportional to the environmental damage (and consequently to the private benefit withdrawn from the production of such damage) and hence adequate to prevent it. For instance, in the power industry, to tax fuels according to their NO_x content is a bad proxy for taxing NO_x emissions. Due to the complex formation of NO_x throughout the combustion process, continuous monitoring of NO_x emissions is important to allow the tax to provide an adequate price signal.

When the tax is raised on a close proxy for specific polluting emissions and there are abatement technologies available, as is the case with a tax raised on fuel consumption aimed at targeting sulphur emissions, any abatement should be taken into account in the tax payment. This tax design stimulates investment in abatement technologies, improves the correlation between the tax base and environmental damage and, consequently, allows the tax payment to provide a price signal adequate to induce behavioural change and rewards the adoption of pollution abatement strategies.

Furthermore, taking into account abatement efforts in the tax payment creates an opportunity to stimulate self-measurement of emissions, which raises tax awareness and pollution awareness. The provision of exemptions or tax reductions in direct proportion to the amount of emissions abated depending on the evidence provided requires self-measurement of emissions. The latter is stimulated by setting presumptive emissions in a level substantially higher than actual emissions. Self-measurement of emissions may lead the polluter to adopt pollution control measures following a better knowledge of his polluting impact.

To leave outside the tax design causes of environmental damage, in the sense that there is only an indirect or weak linkage between the tax base and environmental damage, means less probable reductions in environmental damage following from changes in the tax base. A loose linkage between the tax payment and environmental damage, following the inaccurate choice of the tax base, leads to environmental ineffectiveness since under such a condition there is an increased difficulty in rewarding tax avoidance strategies as well as in financially penalising increased amounts of pollution. At the limit, if there is not a correlation between the tax base and environmental damage, only by coincidence environmental improvement will follow from shrinking the tax base, which is the effect immediately pursued with the tax charge.

A tax base *ad valorem* tends to keep a looser linkage with environmental damage than one measured in physical units, because prices are set and change according to several other variables than the ones relevant to environmental disruption. This is caused by the amount and kind of substances withdrawn from natural sources and released to natural sinks. For instance, higher environmental damage will not come from a price increase in fuel (e.g. due to political crisis in the Middle East) but from an increase in the litters of fuel consumed. Likewise, environmental damage will not depend on waste value (e.g. dependant on its marketability) but on waste amounts and weight, since it is positively correlated with the composition and amount of material flow entering the ecosystem.

The amount of air pollution following from the use of road vehicles depends mainly on the kind of fuel and technology used, as well as the intensity and characteristics of driving (e.g. speed, congested traffic versus free flow traffic). Regulation of vehicle circulation can address all these aspects, whereas regulation of vehicle purchase can only address the technological characteristics of the vehicle. The more the design of a circulation tax takes into consideration all these aspects the closer to one will be the correlation between the tax base and the environmental damage. Once this is guaranteed, taxing vehicle circulation will be more effective in controlling car traffic emissions than taxing vehicle purchase, because it will be a better proxy for taxing emissions following from vehicle use than the latter.

It is necessary to address institutional filters that disturb the price signal provided by the tax by cutting the direct connection the (tax) payment keeps with environmental damage. This is the case when a variable tax payment mirroring the environmental impact of behaviours reaches polluters as a fixed amount (e.g. a service fee) or linked with a variable with none or low causal relationship with the environmental damage. For example, this occurs when the waste tax is charged to waste producers included in a single price covering also the transportation and administrative costs of the waste company.

The tax base should be narrow enough to allow the adoption of tax avoidance strategies corresponding to pollution prevention or abatement and comprehensive enough to impede the adoption of such kinds of strategies when these do not involve environmentally preferable action. If due to the insufficient comprehensiveness of the tax base, behaviours at least as environmentally damaging as the ones taxed are less costly alternatives to the latter, the environmental effectiveness of the tax is threatened. This is the case, for instance, if the waste tax just covers waste directly delivered to landfills and not the one covered by municipal collection schemes. Though full comprehension is often impracticable, due to political, administrative or economic reasons, regulation should be justified based on its ability to address a relevant amount of environmental harm.

The tax base of general energy taxes is not a good proxy for specific polluting emissions

General energy taxes, as taxes raised on measured units (weight or volume) of fuels or energy products and usually using comprehensive tax bases, abide by the environmentally-related tax design but cannot comply with the demanding design of environmental taxes. Since their tax base is not a good proxy for specific polluting emissions, their tax payment cannot mirror the evolution of environmental damage unless one rate per fuel or energy product is used. Therefore, these taxes are able to influence total energy consumption through cost internalisation and consequently foster energy efficiency, especially if they are raised according to energy content and depending on the price elasticity experienced in fuel consumption, but perform poorly in leading fuel choice. They are blunt instruments to achieve reductions in specific polluting emissions.

Depending on the characteristics of the fuel and the combustion technology and process used, environmental damage following each unit of energy consumption will vary depending on the kind of polluting substance considered. For example, a litre of oil is likely to cause different amounts of environmental damage whether one considers CO_2 emissions or sulphur emissions. Therefore, unless there is strict market segmentation, general energy taxes cannot address simultaneously several pollutants, because different hierarchies among the same products will follow depending on the kind of pollutant considered. If such segmentation occurs, the tax needs to be assessed per segment of the market as if it was not one but several separate taxes, each addressing a specific kind of polluting emission.

But general energy taxes also cannot accurately address specific polluting emissions unless they communicate relative polluting impacts by using a single tax rate per product, which increases significantly the administrative costs of the tax and consequently reduces its feasibility. If the same tax rate is used for two or more products, the tax will hardly be able to communicate accurately one specific environmental hierarchy of consumption as required for behavioural steering. This communication will not be possible because each rate communicates a specific relative position in the hierarchy and in the hierarchy referred to a specific pollutant different products will tend to occupy different positions, rather than the same as communicated by a tax rate applied simultaneously to a measured unit of several different products. For example, a tax rate applied equally to coal and oil in a tax aimed at reducing CO_2 emissions wrongly communicates that both products have the same CO_2 content and therefore the consumption of one or the other is indifferent for the regulator.

Since motor vehicle combustion uses technologies causing different environmental impacts, general energy taxes raised on motor fuels are specially limited in their capacity to mirror the evolution of environmental damage following from fuel consumption in vehicles. For example, a litre of diesel causes different environmental impacts depending not only on the pollutant considered but also on the technology of the vehicle burning it.

The element to which the price signal is targeted is the one expected to accommodate a change following the market reaction. For instance, if the tax is charged according to a specific polluting content of fuels, demand is expected to shift towards less polluting fuels as far as that pollutant is considered, whereas a tax differentiation based on energy content is expected to lead towards energy efficiency. The comprehensive tax base of an energy tax component promotes energy efficiency as well as fiscal objectives by eliminating part of the scope for substitution in comparison to a tax on specific polluting emissions or a good proxy for these (e.g. a carbon tax or a sulphur tax), which has its tax base limited to those specific polluting emissions associated with each energy source (SGTC, 1997: 194).

The tax rate has to be higher when, for example, a CO_2 or sulphur emission reduction target is pursued through an energy tax component than when a tax on specific polluting emissions or a proxy for these is used (European Commission, 1992b: 41; Määttä, 1997: 285), because in the first case such reduction is operated via a lower level of energy consumption, which depends mainly on the price elasticity of energy demand that tends to be low in the short to medium term, whilst in the second one it follows from the substitution of energy sources, which depends on relative prices and technological aspects. Therefore, from an environmental effectiveness perspective, a shift of the tax burden from general energy taxes to taxes raised on specific polluting emissions is advised or, when sufficient room has been made available for a tax rise, an increase in the tax raised on specific polluting emissions or a proxy for these.

B. The tax rate should be referred to environmentally correct behavioural change

To best steer behaviours the tax rate should use as reference abatement costs or relative polluting impacts taking into account a specific pollutant. The cost internalisation proposed in environmental taxes is a means to stimulate behavioural change being set in the amount necessary to prevent environmental damage via increased awareness and shared responsibility. The exact amount at which it should be set depends on the precise environmental objective pursued taking into account the price-elasticity of the tax base and relative *effective pollution prices*. The tax rate should make it financially attractive for the polluter to explore abatement opportunities or shift consumption towards more environmentally sustainable patterns.

In any case the abatement level pursued and used as a reference to set the tax rate must take into account the distinction between subsistence emissions and luxury emissions and efficient and inefficient production attending to alternative technologies and processes available (including BATNEC). Such levels cannot mean leaving essential needs unaddressed or ceasing productive activities, since the tax aims at highlighting low-hanging fruit in abatement and opportunities for technological progress to lead towards pollution abatement and technological development and not at punishing or compensating society for the use of the environment.

In environmental taxes cost internalisation is a means

According to conventional economic theory, pollution taxes should reflect external costs imposed on third parties. For example, traffic-related taxes should vary between fuels covering different environmental impacts and other external costs related to motor traffic (Swedish Ministry of Finance, 1997: 192; Speck and Ekins, 2002: 99).

Setting the tax rate at the level required to internalise external costs faces two main problems, namely the difficult quantification of the externalities (e.g. Brornley, 1990, regarding climate change) and the political resistance to high tax rates. Valuation is required across space, from local to global, and time (intertemporal extemalities). To overcome the difficulties in implementing the valuation of external costs Baumol and Oates (1971: 43-46; 1982: 163-164) suggested the trial-and-error process. Targets are first set and then sufficiently high taxes are imposed so as to assure the targeted reductions. To assess whether tax rates are adequate abatement costs should be used as a reference.

This proposal meant a shift in the approach taking pollution taxation from the ideal full cost internalisation to the practical emphasis on damage prevention. Tax payments proportional to the amount of pollution produced only guarantee cost internalisation. This is enough to prevent pollution when all prices are correct and reach undisturbed the ones deciding on pollution and these are economically rational agents. In the absence of any of those conditions, to induce the behavioural change and technological innovation required to address preventative concerns, tax payments need to take a 'forward look' at the environmental disruption caused.

Environmental tax effectiveness requires a 'forward looking' approach

A 'forward looking' approach requires the reference of the tax rate to pollution abatement costs or relative polluting impacts taking into account a specific pollutant. In competitive markets, regulatory tax rates should be set at the level that allows the tax payment to match the abatement costs necessary to attain the precise pollution abatement pursued. A profit-maximizing firm chooses the abatement level to equate its marginal abatement cost with the level of the tax payment due in the absence of abatement, since that abatement level is the one that corresponds to the cost-minimizing abatement level of such a firm (Keohane et al, 2007:160).

The introduction of pollution taxes which put a positive price on pollution tends to highlight inefficiencies in abatement. Falling abatement costs for given emission intensity levels over time can be taken as an indication of innovations in mitigation technology (OECD, 2010: 23) or revelation of low-hanging fruit. If under this condition the tax rate is not revised downwards, the level of abatement will grow above the objective initially set, whereas to avoid decreases in tax burdens in real terms due to price inflation it is necessary to carry out periodical reviews upwards of the tax rates.

When abatement technologies are not available (for example, reduced CO_2 emissions can only follow from fuel switch or cuts in fuel consumption) or when they are but to reduce pollution it is also necessary to shift consumption towards cleaner products, the tax rate must be based on environmentally correct relative *effective pollution prices* in order to signal an environmentally correct hierarchy of behaviours.

The relative difference between rates applied to different products should be proportional to the relative difference in their polluting impacts taking into account a specific pollutant. For example, in a CO_2 tax the tax rate should guarantee a higher tax burden on coal than on biomass and as many times higher as the number of times the amount of CO_2 emissions following from a unit of coal is higher than the amount of CO_2 emissions following from a unit of biomass under the same use.

In absolute terms, the level of the tax rate should not be based on the damage caused to society by the pollution following the taxed consumption. It should instead be based on the precise environmental objective pursued taking into account the price-elasticity of the tax base and relative *effective pollution prices*, being higher the lower the price-elasticity of the tax base and the more demanding the objective pursued.

For the substitute effect to be properly activated the tax differential must reach a threshold level in which more environmentally-friendly products have the same price as (or a lower price than) alternative regular ones. And this effect will be higher the wider the price difference between the clean and the polluting goods is. The relevant relative prices are the ones following from all the components of the price system (*effective pollution price*). Therefore, the tax rate should be set taking into account the relative *effective pollution prices* reaching those able to control pollution. Tax rate levelling can help neutralise wrong incentives provided by other elements of the price system that may disturb the effectiveness of the environmental tax. Environmental effectiveness requires the removal of any institutional filters keeping the tax rate from leading to environmentally correct *effective pollution prices*, both in relative and absolute terms.

It is necessary to account for potential systemic effects whilst setting the tax rate

Since decisions on consumption follow from *effective pollution prices*, it is necessary to account for systemic effects when setting the tax rate. The coexistence within the same tax system of several taxes with overlapping tax bases providing different price signals, following their different rationales (for example, a fiscal rationale and a regulatory one) or different environmental objectives (for example, a tax aimed at reducing CO_2 emissions and another aimed at reducing sulphur emissions), enhances the effectiveness of each other when all the objectives are served by the same environmental hierarchy of behaviours (convergent price signals) and raises problems if that is not the case (divergent price signals).

The requirement for systemic regulatory coherence, basing the environmental tax rate on environmentally correct relative *effective prices*, avoids the annulment of the signal provided by individual elements (environmental taxes) of the taxation system when taxes aimed at reducing specific polluting emissions coexist within the same taxation system with taxes with a fiscal rational. However, this guideline is not helpful to address divergent price signals following from different environmental taxes.

A problem can occur when the environmental hierarchy of consumption changes according to the kind of pollutant considered. For example, fuel A is better than fuel B regarding CO_2 emissions but worse regarding sulphur emissions and the reduction of both emissions is pursued by the tax system with two different taxes. Likewise, systemic coherence might also be difficult to attain when in the taxation system taxes aimed at reducing specific polluting emissions coexist with taxes aimed at raising energy efficiency, since for example biofuels tend to be less CO_2 loaded but also have less energy content per measured unit (volume or litre). In such cases, a choice regarding environmental priorities is necessary.

This is so because consumer behaviour is not determined by partial price signals following from individual taxes, but rather by the *effective price* to the consumer, of which the whole tax burden (*effective tax*) is a component, as well as by relative *effective prices* to the consumer. For instance, the total tax burden on a fuel (*effective fuel tax*), as a component of the *effective price* of that fuel, affects the total consumption of such fuel, while the relative *effective prices* of all the fuels considered impact fuel choice.

In general energy taxes, tax payments can mirror relative polluting impacts within the tax base (with reference to a specific pollutant) if there is tax rate differentiation (see also Section 3.2.2A). In these the optimal tax design to shift demand towards fuels with a lower content of a certain pollutant is one where tax rates are defined separately for each fuel, in terms of fuel quantities (by volume or weight units), and relative tax levels on different fuels are set so as to equate the implicit rate of tax per unit of such a pollutant across fuels (Smith, 1999:507). Such detail in differentiation allows tax payments to mirror relative polluting impacts more accurately and, consequently, better behavioural steering than the use of average values (e.g. to charge diesel and petrol at one single rate taking into account their average CO_2 emissions). However, this technique entails high administrative costs and is not usually used in institutional practice. For example, the Energy Taxation Directive, which is addressed infra in Section 4, used one minimum rate for several products with different polluting contents. The same was observed in the general energy tax practice in Portugal and Sweden.

3.2.3 Capacity to avoid pollution should define the subjective tax incidence

To be able to steer behaviours towards less environmentally damaging patterns, the price signal provided by environmental taxes must reach subjects able to control the cause *sine qua non* of pollution that still enjoy opportunities for improvement. Institutional filters disturbing or delaying such communication need to be removed. This 'forward looking' approach intended at avoiding pollution serves better an environmental regulatory rationale than the 'backward looking' one intended at internalising costs found in Point 3 of the European Council Recommendation 75/436/Euratom that defines 'polluter' in the polluter pays principle as the 'best-payer'.

The introduction of environmental taxes is likely to highlight inefficiencies in pollution abatement and raise polluters' awareness of low-hanging fruit in abatement opportunities leading them to take tax avoidance strategies based on less environmentally damaging behaviour. According to the efficiency criteria (to reduce pollution at the lowest cost) and environmental effectiveness criteria (to reduce pollution where it is still possible, i.e. where price elasticity for pollution is still positive), environmental taxes should have their subjective incidence referred to individuals who have still not explored all their opportunities for pollution reduction.

Tax subjects should be chosen with reference to the power hold over the size of the tax base and the performance of the tax event. Taxpayers should be able to shrink the tax base by using less of the environment. If there is not an alternative to the performance of the tax event, for instance due to the lack of substitutes to satisfy a basic need, the tax base will be relatively inelastic. It is improbable to follow environmental improvement from taxing it until the subject runs out of income to satisfy the need. Likewise, if companies can only stop pollution by stopping their economic activity due to the lack of technologies or processes available to the company the tax must reach prohibitively high levels before becoming environmentally effective.

Narrow tax coverage focused on industries where large substitution effects are expected is likely to deliver high environmental effectiveness. The more homogeneous is the group of taxpayers the more targeted can be the regulatory tax design and consequently the higher will be its environmental effectiveness, since, for example, both the assessment of the pollution abatement cost and the price-elasticity of the tax base required to set the tax rate must be referred to the group of taxpayers. However, selective subjective tax incidence is likely to reduce the advantage tax instruments enjoy over command-and-control instruments in terms of cost efficiency in emissions abatement, since the latter is associated with the flexibility market-based-instruments assigned to polluters in their efforts to abate pollution (see Section 1.3 of Chapter I).

Two other design guidelines follow from the understanding that environmental tax incidence should not depend on pollution amounts but on polluters' relative capacity to improve their environmental performance. Both of which contrast with common institutional practice, which lays the pollution tax burden on the best-payers, namely households, and eases it on the biggest polluters following competitiveness concerns. First, more favourable tax treatments based on pollution amounts should not be provided since they are not underpinned by any environmental criteria and raise issues of environmental effectiveness, efficiency and equity. And, second, specific technological conditions might justify selective subjective tax incidence focused on the industry or on the biggest polluters when such conditions affect the control the taxpayers hold over pollution. Potential market distortions following thereof might be addressed with revenue recycling.

More favourable tax treatment based on pollution amounts is not justified under environmental criteria

Competitiveness concerns have led institutional practice towards two kinds of approaches regarding the biggest polluter: either the reference of the tax rates to them, keeping tax rates low to avoid imposing on them a significant financial burden; or setting the tax rate at the level required to induce environmentally correct behavioural change but provide exemptions and rebates to the biggest polluters. Both approaches omit a justification based on environmental criteria, harm the environmental tax effectiveness, raise equity issues and might be in breach of the polluter pays principle since the non-application of this principle must be a preparatory way to reach its full application in the long-term rather than a permanent restriction.

From a redistributive perspective, pollution payment should be made with reference to the costs imposed on society. From an environmental effectiveness point of view, different distributions of the tax burden should follow only from different capacities to prevent pollution. Exemption and rebates to the biggest polluters do not abide by any of those criteria. They provide a wrong price signal to polluters, 'locking in' polluting processes. This perpetuates harmful effects on the economy and threatens the dynamic incentive associated with pollution taxes. Exemptions lead to inefficiency in the pattern of environmental improvement undertaken if many low-cost improvement options available in the biggest polluting sectors fall outside the scope of the tax. Administrative costs associated with a complex system of tax rebates and refunds are also relevant (OECD, 2001g: 10, 51, 92, for example, estimated such costs at around 1-2% of tax revenues in the Danish CO_2 tax).

A grounding argument for implementing environmental taxes instead of traditional regulatory measures has been the belief that distributing these taxes equally across all polluters produces more efficient results than those measures (OECDf, 2001: 126). Exemptions based on polluting amounts restrain the utilization of cheap emission abatement efforts of the polluters to which they are granted (usually the industry). Increased emissions following those exemptions must be offset by more costly emission abatement options in the covered sectors (usually the household sector) to reach a given target (Kohlhaas, 2003: 6-7; Böhringer and Rutherford, 2002: 1; Speck, 2008: 43). This situation can lead to substantial 'excess costs' (Böhringer, 2008).

Some studies mention a potential 20% increase in the cost of attaining a certain environmental goal when potential exemptions are considered (Böhringer and Rutherford, 1997). For example, under the assumption that carbon leakage rates are low, a CO_2 tax with exemptions might be a 'blunt instrument' (Böhringer and Rutherford, 1997). Moreover, exempted sectors tend to attract capital investment with consequent expansion of the polluting industry.[8]

In regulatory pollution taxes, positive discrimination for more unsustainable behaviours does not breach the equality principle as long as there is no space for improvement. Exemptions based on pollution amounts under the individual control of the polluters (following luxury consumption or inefficient production) not only harm the environmental effectiveness of the tax but also raise equity issues (OECD, 2001g: 79, associates exemption of big polluters with efficiency and equity issues).

Environmental interest groups might perceive proposals to tax polluting bases that embrace several exemptions and offer polluters excessive flexibility to have low environmental effectiveness. Therefore, this kind of tax proposal may fail to capture enough support to get through the legislative process. This might help to explain, for instance, the weak enthusiasm shown by environmentalists towards the proposals made by the European Commission to introduce a carbon/energy tax during the 1990s (Jachtenfuchs, 1996: 147-148; Schlegelmilch, 1999: 3-8).

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[8] This effect is explained *inter alia* by Baumol and Oates, 1982: 207-229; Pearce and Turner, 1990: 107-109; Lewis, 1994: 35-37; Laffont, 1994: 39-66; Crocker, 1996: 61-86; Hanley, Shogren and White, 1997: 73-75.

Though opportunities for improvement tend to overlap with total pollution amounts, they do not always do so. Big polluters can still benefit from more favourable tax treatment. But this should only be based on their low improvement opportunities. Exempting the biggest polluters might harm the environmental effectiveness of the tax since worst performing plants tend to respond faster to the tax than plants that are already relatively environmentally efficient. However, if companies already make important efforts to improve their environmental performance, because for instance environmental consumption is part of their core business, there may be few opportunities to pick low hanging fruit in abatement and the only opportunity for improvement might be in technological innovation.

Technological criteria might justify selective subjective tax incidence among the polluters

Equality polluting subjects might display quite different capacities to avoid pollution. These may justify selective subjective tax incidence. The profit maximisation rationale makes companies especially sensitive to price signals and their organizational structure allows them to respond better than households to such signals. Often the industry has also more alternatives to pollute than households. Therefore, exemptions from environmental taxes provided to industries are due to be costly in terms of environmental effectiveness. Among the industry, the capacity of the companies to prevent pollution depends *inter alia* on how close the regulated aspect is to their core business and their technology and financial capacity, as well as the improvement efforts they already made in the past.

To raise the tax on specific polluting emissions it is necessary to rely on measurement and abatement technologies available. Technological aspects might lead towards the taxation of big polluters, based on feasibility arguments, taking into consideration abatement costs, or because they are the best able to adapt given the nature of the available abatement technology. Capital indivisibilities in technological options, high costs for the most effective types of technology and the higher technological and financial capacity of larger firms might argue in favour of a selective subjective incidence of the tax on big polluters with exemptions to small ones.

Under technological constraints, it might be preferable to initially focus the tax only on big polluters and then progressively extend it towards small polluters following any reductions in abatement or monitoring costs. Covering small polluters might cloud the environmentally successful public image of the tax if technological constraints keep the biggest emissions abatement in large plants. Extending the tax might be environmentally justifiable if it pressures small polluters to follow more environmentally effective development patterns. Such pressure might be especially useful when the consumption or production of the polluting resource follows a rising pattern fuelled by demand side factors.

Revenue recycling can address competitiveness distortions following selective subjective taxation of polluters

Pre-allocation of environmental tax revenue might raise the behavioural impact of the tax. Returning the revenue back to the sector where it was collected (revenue recycling) might help enhance environmental tax effectiveness by allowing selective subjective tax incidence on polluters able to prevent pollution and tax levels sufficiently high to induce behavioural change, as well as by increasing tax awareness among the covered polluters. However, there are issues of efficiency and shared responsibility associated with this tax design.

To address the effects of distorted competitiveness between the large regulated and the smaller unregulated plants might be useful to consider the recycling of the tax revenue back to the industry as a whole rather than to each taxpayer to avoid the annulment of the incentive provided by the tax. If revenues are recycled according to environmental criteria this aspect of the tax design might feed into the environmental effectiveness of the tax, since the linkage between the tax charge and the tax goal is highlighted to the taxpayers.

However, this design raises issues of efficiency and lack of compliance with the polluter pays principle regarding the emission charges refunded. Following the refund polluters do not pay the full environmental cost of the pollution their production causes, leading to a potential welfare loss to society due to distorted resource allocation. Too many productive resources are allocated to polluting production relative to cleaner production, since the cost minimizing output level of the regulated firms exceeded the social optimal output level (OECD, 2010: 10).

Another possible source of concern with revenue refund is the watering of the incentive provided by the charge. Compared with an emission tax of the same magnitude, a refunded charge may inhibit the spread of innovations among the regulated plants (OECD, 2010: 36). This is so, since "by keeping the knowledge about the innovation to itself, a plant is able to reduce its emission intensity and improve its position relative the other regulated plants, which will render it a higher net refund through the refund mechanism" (ibidem).

There are three cumulative conditions under which the refunded charge gives rise to approximately the same incentives to invest in emission control as a non-refunded emission tax of the same magnitude, i.e. regulated plants invest in mitigation until the marginal cost per unit of emission reduced is equal to the unit tax. First, there must be a single output upon which the refunding can be based and, second, each plant's output must be small enough relative to the total output by regulated plants to form a competitive situation (OECD, 2010: 9-10). To these conditions should be added that the single output on which refunding is based must not coincide with the tax base.

3.3 Environmental tax powers should be allocated according to environmental expertise and diverted away from non-environmental concerns

Some institutional designs seem to be more adequate than others to create and manage environmental taxes. The regulatory feature these taxes embed will be better served if their design and management lies with institutions that hold environmental expertise and do not hold an interest in their tax base. Market-based instruments, such as taxes, leave the decision on how to abate to those best informed about abatement possibilities, i.e. the polluters. Though the Ministry of Environment might hold conflicting interests that hinder the environmental tax effectiveness, once these are addressed its technical expertise might justify a leading role in environmental tax design.

Institutions committed to the tax base will tend to preserve it instead of leading behavioural change towards its reduction. Disturbing fiscal interests occur when the entity designing and/or administering the tax is the one keeping its revenues. In such cases, the tax design is likely to be influenced simultaneously by interests in constant tax revenue fluxes that require a stable tax base and decreasing environmental harm expressed in a shrinking tax base. Revenue interests might be neutralised if the pre-allocation of the tax revenues to the group of taxpayers or third entities is instituted in the tax design. Other kinds of interests might also need to be addressed, such as for instance the desire to fully exploit available capacity in public infrastructures dealing with waste and effluent.

The high technical complexity of the environmental problem, which is one of the conditions under which taxes might be especially useful, and the need to rule the tax design by environmental criteria to achieve environmental effectiveness, justify the environmental tax design by the Ministry of Environment. Market-based instruments, such as taxes, leave to the ones best informed about abatement possibilities, i.e. the polluters, the decision on how to abate. Institutional frameworks based on consensus-cooperation involve the risk of having the environmental tax design distorted following the interests of the polluters. This risk might be higher when the Ministry of Finance is in charge of the tax design than when the responsibility lies with the Ministry of Environment due to the bigger information asymmetry between the regulator and the regulated in the first case.

4. INSTITUTIONAL PRACTICE USUALLY DOES NOT INCLUDE THE DESIGN FEATURES OF ENVIRONMENTAL TAXES: THE ENERGY TAXATION DIRECTIVE

The brief overview presented next of the basic EU legal framework most relevant for environmental taxation, namely the Energy Taxation Directive (2003/96/EC)[9], which is also a cornerstone of the European Climate Change Programme, shows that this failed to accommodate the design features of environmental taxes, namely the ones regarding the tax base and tax rate. Consequently it was unable to lead national institutional practice towards such inclusion and consequent higher levels of environmental effectiveness, as evidenced by the Portuguese energy tax and the Swedish energy tax analysed in Chapters III and IV, respectively. Energy consumption is influenced by *effective pollution prices* rather than partial price signals following from individual taxes. Therefore, the tax rate structure introduced by the Directive is likely to have also (indirectly) disturbed efforts taken at national level to lead consumption towards cleaner fuels following the impact general energy taxes have on the price signal provided by national taxes raised on specific polluting emissions following from fuel consumption or proxies for them. The 2011 European Commission's proposal for the revision of this directive is also not expected to bring national energy taxation systems up to their full potential in terms of environmental effectiveness.

The tax base used in the Directive was not appropriate to lead towards cleaner consumption or energy efficiency, since it did not target the price (i.e. the tax) to specific polluting impacts or energy content of fuels. The Directive laid down minimum rates, based mainly on the volume of energy consumed (EUR/1000l), for products used in heating, electricity and motor fuels, arguably aimed at strengthening the incentives for a more efficient and less polluting use of energy. However, to accurately signal the environmental hierarchies of consumptions serving such objectives, taxes have to be raised on specific polluting emissions (or energy content) or a good proxy for these, since only then will tax payments be able to mirror the evolution of environmental damage (or the degree of efficiency in energy use). The polluting impact associated with a litre of each fuel depends on the specific pollutant considered and how the fuel is used (process and technology). The technology used is especially relevant to address for example the environmental damage following from vehicle fuel consumption (see Section 3.2.2A above).

The failure to abide by environmental criteria and consequently to lead energy consumption towards cleaner fuels was also evident in the tax rate structure adopted by the Directive. Its minimum levels were due to reflect the competitive position of the different energy products and electricity and as far as possible be based on the energy content of the products, except for motor fuels. However, the rate structure provided environmentally incoherent price signals to the market since, for energy products, it was set according to historical rates in the Member States and not based on abatement costs or relative polluting impacts taking into account a specific pollutant. This led sometimes to higher tax burdens on cleaner products than on polluting ones.

For example, according to the minima, coal was the least taxed and ethanol was the most taxed and diesel was taxed at a lower rate per litre than petrol in spite of its higher energy and CO_2 content per volume (Annex I of the Directive). Since renewable energy sources were taxed at the same rate as the energy source they were intended to replace (for example, biodiesel was taxed the same as diesel) and rates were based on volume, rather than energy content, products with lower energy content such as renewable energy sources carried a heavier tax burden compared to some more polluting fuels they were competing with.

Furthermore, equally polluting energy uses were sometimes taxed at different rates depending on non-environmental criteria. There were different minimum levels according to the use of the energy products and electricity, for example business use versus non-business uses (Arts. 5, 7 (2) of the Directive), allowing that energy products used as motor fuels for certain industrial and commercial purposes and those used as heating fuels were normally taxed at lower levels than those applicable to energy products used as propellants (Point 18 of the Directive).

The Directive also failed to abide by environmental criteria whilst allowing most Member-States to provide tax benefits for waste oils which were reused as fuel, either directly after recovery or following a recycling process for waste oils, and where the reuse was subject to duty. The waste management hierarchy was

[9] Council Directive 2003/96/EC of 27 October 2003 restructuring the Community framework for the taxation of energy products and electricity, OJ L 283/51, 31 October 2003.

not reflected in the Directive, being irrelevant for the concession of the tax benefit the precedence established in waste oils management, namely first reuse, second recycling and third recovery (COM(1996) 399 final, 30 July 1996). In the case of waste oils this requirement demanded regeneration to take precedence over any other form of waste management, namely waste oils recovery as fuel.[10]

The Directive gave Member States flexibility to define and implement policies appropriate to their national circumstances and introduce or retain different types of taxation on energy products and electricity. And they could comply with such thresholds by taking into account the total charge levied in respect of all indirect taxes which they had chosen to apply (excluding VAT). Fiscal arrangements made in connection with the implementation of the Directive were a matter for each Member State to decide. However, in any case national tax rates had to comply with the minimum taxation thresholds introduced by the Directive.

Therefore, national taxes aimed at steering consumption towards cleaner fuels (or energy efficiency) could have overcome the disturbance introduced by the Directive if they were based on environmentally correct relative *effective pollution prices*. This would have allowed them to compensate for the wrong price signal provided by the tax rate structure introduced by the Directive, making sure that the whole tax burden raised on each fuel correctly signalled the environmental hierarchy of consumptions. However, this was not common institutional practice, since as far as business energy consumption was concerned Member States tended to stick to the minima to protect the competitiveness of the national industry.[11]

The 2011 European Commission's proposal for the revision of the Energy Taxation Directive[12] tried to improve the current legal framework by including the split of energy taxation between a CO_2 component and an energy component. However, it failed to acknowledge the importance of other pollutants than CO_2 emissions and environmentally correct relative *effective pollution prices*. The split would allow part of energy taxation to become more compliant with the environmental tax design since the CO_2 tax rate component would mirror this specific polluting impact of energy products (except regarding biofuels for which exemptions from the CO_2 element were proposed despite their still positive CO_2 content – Art. 1, Point 14 of the Proposal).

However, tax rate structures set according to energy content, though stimulating energy efficiency, fail to mirror relative polluting impacts taking into account a specific pollutant and lead towards cleaner fuel consumption. The price signal provided by the energy tax component is likely to disturb the one provided by any other tax component mirroring specific polluting impacts of fuel consumption (including CO_2 emissions) that is not based on *effective pollution prices*. According to the regime set in the Commission's proposal, to be able to effectively steer consumption towards cleaner fuels Member States need to raise fuel taxes at rates that compensate for the disturbance introduced by the general energy tax component and go over the minimal thresholds set in the proposal, as was already the case in the 2003 version of the Directive.

5. THE THESIS STRUCTURE

The structure of the dissertation is as follows. In this chapter, the reader is led into the theme of the dissertation with an overview of its argument followed by a preliminary conceptual analysis likely to clarify grounding ideas to the argument. Next the design features of environmental taxes on which their environmental effectiveness depend are analysed with further detail. A brief overview of the failure of the Energy Taxation Directive to include the design features of environmental taxes helps to demonstrate and explain the scarce use of the environmental tax design in institutional practice, which is restated by the

[10] However, the Court of Justice of the EU has already declared that by failing to take the necessary measures under Article 3 (1) of the Directive on the disposal of waste oils (75/439/EEC) to ensure priority to regeneration Member States who were allowed to provide the referred tax benefit had failed to fulfil their obligations under this directive. Judgment of the European Court of Justice 15 July 2004, C-424/02 *Commission v. United Kingdom* [2004] ECR I-7249.

[11] Information provided by the OECD/EEA database on environmentally-related taxes, available at http://www2.oecd.org/ecoinst/queries/.

[12] European Commission (2011), Proposal for a Council Directive amending Directive 2003/96/EC restructuring the Community framework for the taxation of energy products and electricity, COM(2011) 169 Final, 13 April 2011.

evidence provided by the national cases presented further down in the thesis. After presenting the thesis structure and before the conclusions, the case selection and methodology are explained.

After this introductory chapter, Chapter I describes the theoretical framework underpinning pollution taxes, assess its frailties and its application in institutional practices leading to the 'environmental tax/fiscal reform' concept. This chapter aims at showing how theoretical incompleteness and failure to steadily guide towards the environmental tax design led to common institutional practice with environmentally ineffective pollution taxes by the end of the 1990s.

Chapter II describes the Danish waste tax since its inception in 1987 until 2011. However, following the availability of evaluation studies, empirical data collected focused mainly on the period between 1987 and 2001. This chapter presents detailed analysis of the tax design as well as of the institutional environment where the tax was applied, due to the relevance of the latter for environmental tax effectiveness, drawing conclusions thereof. In this chapter as well as in the following two, empirical data are often organised into tables representing periods of time to facilitate information analysis and comparison.

Chapter III uses the Portuguese energy tax to draw a clear line between environmental taxes and pollution taxes. The scarce evaluation studies available have significantly constrained the analysis of the causal linkage between the tax design and the poor national environmental results reported in official documents along the period 1990 - 2011.

Chapter V describes and analyses the performance of the Swedish energy taxation system, which included the CO_2 tax, the sulphur tax, the energy tax and the NO_x charge, during the period 1990 - 2011. It discusses in detail the tax design and uses the evaluation studies available to draw conclusions on the causal linkage between the inclusion of design features of environmental taxes in institutional practice and the environmental results reported by the impact assessments of the taxes.

The Chapter Conclusions summarizes the findings. Taking the empirical evidence collected along the three case studies final considerations are drawn on the theoretical proposal laid out in the present chapter. The conclusion is that empirical data confirm the hypothesis that environmental effectiveness is higher, with stronger and faster environmental effects, following the introduction of the tax, the more the tax includes the design features of environmental taxes. This shows the substantial difference between environmental taxes and environmentally-related taxes.

6. The Case Selection

This research aimed at assessing whether the level of environmental protection delivered by environmentally-related taxes depends on their design according to specific design features that turn them into regulatory instruments of environmental policy. Bearing in mind such a causal relationship, the case selection was targeted at instruments with different degrees of compliance with the theoretical proposal.

A high amount of 'environmentally-related tax revenues' according to the OECD/EEA database, which is the most complete official database on pollution taxes, was taken as a potential indicator of a strong use of the tax system towards environmental goals. Therefore, the country selection focused on Denmark, Portugal and Sweden, where the experience with pollution taxes was expected to be statistically significant as, during the 1990s, these three countries held high rank positions in the OECD/EEA statistics.

The association between tax instruments and environmental concerns came more actively into the political discourse and the tax law in the 1990s. Tax shift approaches, which put an important fiscal emphasis on environmentally related tax interventions and tend to put environmental concerns into second place, date back to then. The contrast between fiscal pollution taxes and regulatory pollution taxes, and the problems arising thereof, became then more evident. Therefore, this analysis focused on that period, though with references to relevant antecedents and updates (up until July 2011) regarding the instruments researched. The main results obtained in each country case and which supported the general conclusions presented in the critical analysis were still valid in July 2011 unless clearly stated otherwise in the relevant chapter.

Taxes raised on energy consumption were assessed both in the Swedish case (energy tax, CO_2 tax, sulphur tax) and the Portuguese case (energy tax) because these kinds of taxes are able to generate a significant financial impact as well as an important environmental impact depending on the specific design features adopted. They have traditionally played an important role in tax shift processes and help in understanding the difference between 'environmentally-related taxes' and 'environmental taxes'. The comparative analysis between the Portuguese case and the Swedish case allowed us to assess the relevance of systemic effects for environmental tax effectiveness, as well as to contrast the structure introduced by the Energy Taxation Directive (2003/96/EC) for energy taxation, to which the Portuguese case was closer, with that of the 2011 European Commission's proposal for the revision of the same directive, to which the Swedish case was closer.

Other taxes analysed, namely the Swedish nitrogen oxide (NO_x) charge and the Danish waste tax, were pollution taxes where the regulatory feature was expected to be more probable than the average due to a direct connection with pressing national environmental issues and characteristics usually responsible for making economic instruments interesting options. The environmental problems addressed in Sweden, i.e. acidification and eutrophication caused by deposition of nitrogen compounds, were supranational in its sources and effects. And the Danish case related to a change in environmental policy argued to have facilitated the use of economic instruments, i.e. the shift of the focus from production to consumption.

To assess the environmental effectiveness of environmental taxes in contrast to environmentally-related taxes, the case selection covered taxes that complied with normative environmental taxes to different degrees. There were taxes raised directly on measured emissions (the NO_x charge), on proxies for specific polluting emissions (the CO_2 tax and the sulphur tax) and taxes raised on tax bases with a close linkage to environmental damage but unable to work as proxies for specific polluting emissions (the Swedish energy tax and the Portuguese energy tax), as well as taxes whose tax rate was based on abatement costs (the sulphur tax and the NO_x charge) and others where their setting up process was unclear or ruled by fiscal interests (the Portuguese energy tax and the Swedish CO_2 tax and energy tax). Regarding the subjective tax incidence, the reference of the tax design to capacity to avoid pollution was only the rule in the sulphur tax and the NO_x charge and, though to a lower degree, in the waste tax.

The institutional design in each of the taxes analysed was also diverse allowing some conclusions on how this might be related with compliance of the tax design with normative environmental tax features. The Swedish NO_x charge and the Danish waste tax (until 1993) were designed and administered by environmental authorities. The Swedish sulphur tax, though administered by the Ministry of Finance was designed by the Swedish Environmental Protection Agency. All the other taxes analysed were designed and managed by tax authorities and entailed a stronger revenue capacity than the three previously mentioned.

7. METHODOLOGY

This dissertation argues that environmental taxes are substantially different from environmentally-related taxes. Their demanding tax design explains their less generalised use but also better environmental effectiveness. Environmental taxes are understood as regulatory instruments operating through price signals aimed at steering behaviours towards environmental damage prevention via increased awareness and shared responsibility. The environmental effectiveness of these tax payments depends on their reference to opportunities for improvement rather than mere pollution amounts and targeted at economic agents able to perform it.

The research done aimed at setting out this difference clearly following the failure of the literature on pollution taxation to do it. The analysis of the information collected through interviews, official and private documents, as well as book-based research was issue-targeted. Both the theoretical framework used to underpin pollution taxes, as well as institutional practice in pollution taxation in three countries, namely Denmark, Portugal and Sweden, were addressed to assess the referred substantial difference. A causal relationship between the way in which tax designs abided by the normative design features of environmental taxes and the different degrees of environmental effectiveness following from them was researched.

The methodology used to collect information on all the three cases included the collection and analysis of documents found in public and private databases, as well as on-line, telephonic and presential interviews with experts and politicians involved in the legal draft, implementation and evaluation of the instruments addressed. Access to private databases was facilitated by some of the interviewees and the institutions where the empirical analysis was carried out. This access regarded published and unpublished material, such as evaluation studies, political reports and expert analyses. All sources used are listed after the Conclusions as 'References' and referred within the text whilst directly quoted.

The interviews were carried out by the author in Portugal, between January and April 2002 and October and December 2003, whilst holding a lecturer position at the Portuguese Catholic University, in Denmark, between June and November 2002, during a visiting research position at the Danish Ministry of Environment (Policy Analysis Division) under the supervision of Professor Mikael Skou Andersen, and in Sweden, in May 2002 and between February and September 2003, during a visiting research position at the Swedish University of Agricultural Sciences (SLU) (Department of Forest Economics) under the supervision of Professor Bengt Kriström.

Conclusions reached were constrained by results provided by studies on the effects of pollution taxation which were not always straightforward. Reasons lay in *ex ante* evaluations and counterfactual analyses, which took as baseline a scenario without taxes (*e.g.* NUTEK 1994; Larsen and Nesbakken 1997; Munksgaard *et al*, 1998; Boom 1998), as well as in the exclusion of some relevant sectors, like transport (*e.g.* SEPA, 1995: 1, 8, 55), whose non-inclusion may turn a 3-5% reduction in CO_2 emissions attributed to a tax into a 60% reduction (Andersen *et al*, 2001: 59). Furthermore, some evaluations did not assess directly the relevant tax, such as for instance the CO_2 tax, but rather stressed trends in energy consumption and looked at the total effect of the CO_2 tax and the general energy tax (*e.g.* the 1997 SWTC evaluation – Andersen *et al*, 2001: 62-63).

CONCLUSIONS

This dissertation aims at proving the substantial difference between environmental taxes and environmentally-related taxes expressed in different rationales and designs that lead to different levels of environmental effectiveness. Environmental taxes are understood as regulatory instruments aimed at steering behaviours towards pollution prevention whose environmental effectiveness depends on the inclusion of specific design features. After briefly presenting the thesis argument in Section 1, Section 2 of this introductory chapter clarifies some concepts that are preliminary to it.

Section 2 first analyses taxes as instruments of environmental policy with a regulatory rationale guided by precise environmental goals that stands in clear contrast to a fiscal rationale. The literature overview shows how this approach fulfils a gap in the literature. Second, it explains how the limits set by cost-benefit analysis to cost internalisation do not apply to environmental taxes, being the effectiveness of the latter limited only by the price system according to which they work. And, finally, it puts forward the importance of competitive markets for environmental tax effectiveness.

Section 3 presents the design features of environmental taxes. First it explains why these must be ruled by environmental criteria, including precise environmental goals. Next it presents how the tax design can refer to opportunity for improvement rather than mere pollution amounts. This requires design features that institute tax awareness and avoidance in the environmental tax design, as well as tax payments led by a 'forward looking' approach to polluting impacts and imposed on subjects with capacity to avoid pollution. Such 'forward looking' approach depends on raising the tax on specific polluting emissions or a proxy for them at rates referred to abatement costs or relative polluting impacts taking into account the specific pollutant considered in the tax base in an amount able to induce environmentally correct behavioural change. It is also argued that institutional practices are likely to follow more closely normative environmental taxes if the entity holding environmental tax powers has technical expertise in environmental matters and does not keep an interest in the tax base.

To give evidence and to analyse the causes for the scarce inclusion of the design features of environmental taxes in institutional practice, not only at national level as explained in detail in Chapters II to IV but also at the European Union level, Section 4 presents a brief overview of the EU legal framework most relevant to environmental taxation, namely the Energy Taxation Directive, concluding that its regime failed to guide the Member States towards environmentally effective design in energy taxation and may have disturbed their efforts in that direction. After presenting the thesis structure in Section 5, and following its strong empirical basis, some detail is put in Sections 6 and 7 of this chapter to explain the case selection and the methodology used, respectively. This Introduction ends with the present concluding section.

The following chapter (Chapter I) presents the main theoretical framework underpinning pollution taxes. It aims at showing the failure of the literature to clearly distinguish between environmental taxes and environmentally-related taxes and guide towards the inclusion of design features of environmental taxes in institutional practice. The next chapters provide empirical evidence on how, following the demanding environmental tax design, environmental taxes are more environmentally effective and less common in institutional practice than environmentally-related taxes. The theoretical hypothesis proposed in this dissertation is tested against and confirmed by empirical evidence withdraw from three countries, namely Denmark, Portugal and Sweden (Chapters II, III, and IV, respectively), to which the summarising and concluding chapter follows.

CHAPTER I

THE THEORETICAL FRAMEWORK UNDERPINNING POLLUTION TAXES AND INSTITUTIONAL PRACTICE LEADING TO ENVIRONMENTALLY-RELATED TAXES

This chapter analyses the main theoretical framework underpinning pollution taxes, namely the theory of internalisation of externalities, the polluter pays principle (hereafter also PPP), the least abatement cost argument and the double dividend argument, presents its weakness in terms of policy guidance and gives an overview on its application in institutional practice. The vagueness and non-consensual policy guidance it provides regarding environmental tax design issues led to the 'misapplication of good theory'. This process has simultaneously fostered the transference of the idea of green taxation into institutional practice and diverted it away from the economic approach based on regulatory intervention aimed at damage prevention to a more politically appealing one that emphasises damage restoration and revenues.

The input from economic theory, which was the root of the environmental taxation idea, to shape the environmental tax design has decreased over time. Its approach has also evolved from internalisating externalities (Pigou) in the 1970s to shifting the tax burden from 'goods into bads' (double dividend discourse) in the 1990s. A legal dimension has complemented the economic roots of the polluter pays principle with liability and equity concerns. Theory has at most provided principles and different rationales have disturbed the consensus over precise policy guidelines. Therefore, it was unable to constrain the bureaucrats in policy design and this became shaped by real-life politics.

Stakeholders with an active role in the policy making process, which led to institutional practice in pollution taxation, shaped the theory along their fiscal interest whilst translating it into policy proposals. Fiscal interests and win-win discourses in the double dividend argument and the ecological modernisation theory made the green taxation idea appealing to policy makers. This has broadened the concept of environmental tax, creating the 'environmentally-related tax' and feeding into the processes of 'environmental tax reform' and 'environmental financial reform', both more fiscally focused than regulatory oriented.

1. MAIN THEORETICAL APPROACHES AVAILABLE

The roots of environmental taxation are in the principle of externality taxation, an idea introduced in the 1920s by welfare economics. This work started by Pigou was not systematically addressed by economic literature until the 1960s, when Coase disputed its conclusions.[13] But the idea of reducing pollution through tax instruments only gained momentum after the 1970s, following the development of a whole branch of microeconomics evolving around externality taxation (Andersen, 1994: 32).

The initial idea introduced by Pigou was thereafter repetitively used to legitimise policy action (Jobert, 1994; Faure *et al*, 1995). Other ideas came along the process and acted as catalytic elements of change in the policy instrument 'externality tax' (Sabatier, 1987: 649-650; Hall, 1989: 7, 10). All these ideas have then worked as 'cognitive frames' (Lau and Sears, 1986; Vowe, 1994; Schön and Rein, 1994: 34-36; Jachtenfuchs, 1996: 25-26), which have structured tax policy options in several countries especially since the 1980s.

There are four main theoretical contributions to support the use of pollution taxes, namely the theory of internalisation of external costs (Pigouvian theory), the polluter pays principle, the least abatement cost argument and the double dividend argument. These contributions can be organised under two major currents of academic thought, the mainly Anglo-Saxon field of environmental economics and the theory of

[13] Pigou (1929: 194-7) was the first to suggest to impose on the causer of pollution a tax in an amount equivalent to measured emissions aimed at correcting externalities following thereof. Coase (1960) argued that, in the absence of transaction costs, the Pigouvian tax is not necessary because bargaining will lead to an efficient outcome regardless of the initial allocation of property rights.

ecological modernisation, which originated in Germany. The Pigouvian theory, the polluter pays principle, especially in its original version, and the least abatement cost rationale fit into the economic oriented academic school of thought, whilst the double dividend argument belongs to the more political and philosophical school of thought of ecological modernisation.

Pollution taxes have been associated with the need to integrate economic policy and environmental policy. Efficiency has been a recurrent argument to promote such instruments since focusing on efficiency is the most common way used to integrate economic development and environmental concerns (EEA, 1996: 18). However, there are different kinds of efficiency subjacent to each rationale: optimal efficiency in the Pigouvian model, improved allocation of resources in the polluter pays principle, improved cost-efficiency in regulatory intervention in the least abatement cost argument and improved efficiency in the tax system in the double dividend argument.

The process that opened a window of opportunity for the use of tax instruments within environmental policy was informed by all the rationales mentioned. The relative weight of these rationales in the political process evolved over time and was not the same all over Europe. The initial economic rational has progressively and partially given in to other rationales in the tax design, despite maintaining a central role in the political discourse on the use of these policy instruments. This process culminated in the adoption of environmentally-related taxes. This is a broad concept which feeds on contributions from the referred several theories and has roots in the incompleteness of those theoretical contributions.

These theoretical contributions have not been neither sufficient to provide a conceptual benchmark to guide policy action nor equally useful to underpin environmental taxes as instruments of environmental policy which, through allocation of external costs to the economic agents best able to avoid them, pursue primarily environmental damage prevention. The regulatory approach aimed at environmental damage prevention taken by environmental taxes is better underpinned by the polluter pays principle than by the double dividend argument, which requires pollution taxes with a fiscal rationale. The least abatement cost argument is useful to highlight the importance of having the definitional elements of environmental taxes flowing from the environmental objective pursued. However, neither the polluter pays principle nor the mentioned argument provide a precise explanation on how the tax should be designed. The Pigouvian theory helpfully explains how the price signals following from environmental taxes lead to environmental improvement. However, the Pigouvian proposal for unlikely feasible full cost allocation cannot guide the environmental tax design. In contrast to the approach suggested by this theory, the regulatory approach requires tax payments in the amount necessary to induce behavioural change.

1.1 The Theory of Internalisation of External Costs

Pigou (2002: 192) was concerned with welfare maximization and built up a theory of economic efficiency suggesting national dividend, and consequently welfare, would be increased to an optimal level if external environmental costs were fully internalised. The rationale behind this proposal was that a proper allocation of costs between those engaged in economic activities causing pollution, both polluters and pollutees, would allow equalising social benefits and social costs associated to those activities.

Pollution taxes would contribute to reach such a result by forcing equivalence between private costs and social costs per product or activity. This condition would be fulfilled if the costs of marginal uncompensated externalities were imposed via a tax on the agents causing them instead of burdening the society. A logic symmetric to the one developed for the external costs case should be applied when external benefits occur instead. In such case, the tax should be replaced by a subsidy to the economic agents.

Since the Pigouvian model aims at neutralising the difference between the marginal social net product and the marginal private net product (Pigou, 2002: 134), the tax rate should be set at the amount of the marginal external costs per unit of pollution (Cropper and Oates, 1992: 680; Bovenberg and Goulder, 1996: 985; Fullerton and Metcalf, 1998: 227; EEA, 2000a: 14). The Pigouvian theory focus on the internalisation of external costs. Therefore, according to such theory, the pollution tax rate should be calculated according to external costs rather than be referred to precise amounts of pollution abatement (Pigou, 2002: 193).

This theory aims at welfare maximisation rather than any specific environmental goal, with environmental improvement being a consequence of such general efficiency gains attained via full cost internalisation. Expected improvements in resource allocation shall occur as result of behavioural changes induced in the economic agents by the re-allocation of external costs. Cost internalisation is a first means which starts another means, behavioural change, in order to accomplish a specific objective, i.e. welfare maximization (COM(2005) 532 final, 14). Environmental results are secondary to the economic goal of increasing economic welfare by correcting market prices via cost allocation (Milne, 2003: 12, 'shadow prices' – Kneese *et al*, 1970: 84).

Effects of the Pigouvian tax are twofold: an abatement effect and an output effect. The first arises because of incentives to reduce emissions as long as marginal abatement costs are lower than the unit tax rate. The output effect is a result of increased production costs due to abatement costs for reduced emissions and tax costs for remaining emissions. Thus, following the adoption of a Pigouvian tax on emissions, two kinds of effects on resource allocation are expected: a direct improvement of the environment through reduced emissions and an indirect improvement through a structural shift in production towards less environmentally damaging goods.

The tax is the endogenous variable in a Pigouvian model. Regulatory intervention is explained by the absence of pollution control in the pre-tax moment and its occurrence in the post-tax moment. Polluters develop their decision-making process taking the amount of tax levied as a reference. A behavioural response is expected in a context where it is cheaper for the (rational, cost-minimising – Posner, 1992: 376-7) economic agent to control polluting emissions than to pay for the full cost associated to such emissions (i.e. private costs plus external costs internalised by the Pigouvian tax).

The Pigouvian approach is mentioned in some reference reports (e.g. EEA, 2000a: 14), but scarcely followed in institutional practices. For instance, even though in the early 1990s the Swedish Ministry of Finance claimed environmental taxes should theoretically be constructed as suggested by Pigou (Budget Bill 1991/92: 100, Annex I: 1.5, 5), this calculation method was scarcely used in Swedish practice (SEPA, 1995: 3). The UK landfill tax is viewed as an example of a Pigouvian tax (EEA, 1996: 28; Määttä, 1997: 224) as its rates were initially supported by externality evaluation using cost-benefit analysis (Powell and Craighill, 1997: 306; OECD, 2001g: 24-5). However, the 'pigouvian feature' was lost with the 1998 decision to increase the basic rate (OECD, 2001g: 111).

In 2004, New Zealand made use of Pigou's rationale to base the proposal for the design of a tax aimed at CO_2 emissions to be applied upstream from the point at which emissions occurred with price repercussion on effective polluters. The level of the tax for each product was made dependent (as far as it was practicable) on the emission factor of the product. The rate of the tax was intended to approximate the price of carbon dioxide-equivalent emission units on international markets in the period 2008-2012. Tax revenues should not be used to improve the Crown's fiscal position, but recycled into the economy through the tax system. But as the calculation method involved some problems the government could not specify at an early stage the basis on which prices could be adjusted (New Zealand Inland Revenue Department, 2005). This proposal was never adopted (NZPA, 2005).

1.2 The Polluter Pays Principle

The polluter pays principle (hereafter also PPP) emerged during the outset of modern environmental policy, in the early 1970s, when the government was asked to implement positive action to protect the environment (Stevens, 1994: 580). However, its vague formulation due to its evolving nature from an economic principle to also a legal one and expanding cost coverage referred to specific but unspecified environmental goals, kept it from providing effective policy guidance.

With ideological roots in economics, it expressed concern over international trade implications of a then new school of environmental economists which promoted as a policy principle an economic theory that called for internalisation of previously externalised environmental costs (Gaines, 1991: 465-466). It worked as a critical interface between environmental policy and international trade, introducing economic concepts into environmental policy, in line with the environmental economics political economy underpinning ecological modernisation ('economisation of the environment' – Murphy, 2000: 3). Countries adopting strong environmental protection measures were therefore especially enthusiastic regarding its adoption.

The PPP's vague formulation enabled it to have different meanings over time and such vague formulation was kept as its meaning was steadily enlarged (from prevention to remediation and from efficiency to equity). As a guiding principle to preserve equity in international economic relations, and not an absolute doctrine to be applied regardless of its economic or environmental consequences, the PPP gained great openness. It worked as a rule of reason ('economic reasonableness', Grabitz and Zacker, 1989: 439) which countries should follow whenever it was not socially, economically or environmentally unreasonable to do so (Gaines, 1991: 477).

The only consensual interpretation of the PPP is that it aims at ensuring that the costs of repairing environmental impairment shall not be borne by public authorities (i.e. the taxpayer), but rather by the polluter (Krämer, 1992: 244). This normative doctrine deems that the polluter should bear the cost of measures to prevent and control pollution, since the internalisation of environmental externalities into production and consumption costs is expected to allow the attainment of environmental goals with economic efficiency and a fair and equitable appropriation of pollution abatement costs.

Vague formulation failed to provide a 'belt of restriction'

The polluter pays principle was endorsed by several supranational organizations (e.g. the OECD, the European Union and NAFTA) and has been incorporated in many national and supranational documents (e.g. 1992 Rio Declaration, Agenda 21, 2002 World Summit on Sustainable Development Implementation Plan and successive Environment Action Programmes of the European Community), influencing the domestic law even of countries which have never codified it (e.g. the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA) in the USA).

However, the PPP is particularly vague (de Sadeleer, 2002: 60) failing to provide guidance (i.e. a reason that argues in one direction – Dworkin, 1977: 26). Different outcomes might result from its application since unlike rules, principles are relatively vague formulations referring to the attainment of general goals or values for society (Hart 1994: 260) and do not dictate any specific decision. Moreover, lack of legal clarity due to absence of a precise and generally accepted legal definition has led to discord over the PPP interpretation. EC law formally adopted this principle in Article 191 of the TFEU but did not define it. The OECD document which introduced it, i.e. OECD Recommendation C(72)128, is not binding international law since it was never ratified by any government.

The increasing number of international treaties and instruments and contemporary national legal systems where the principle can be found endorse it in various ways. For instance, the Commission's Green Paper on environmental damage considered the creation of collective compensation schemes an application of such principle to diffuse damage, contaminated sites and authorised discharges (Commission, COM(93) 47, 14 May 1993). Three other EU documents (Commission White Paper on environmental liability (COM(2000) 66, 9 February 2000), Commission proposal for Directive 2004/35 and Directive 2004/35, OJ L 143/56, 30 April 2004) which invoke the PPP as the leading and guiding principle vary considerably in the content assigned to it (Krämer, 2005: 133).

Deficient implementation is due not only to such ambiguity, which complicates the principle's applicability, but also to some features of the political process, namely the endowment of economic agents with the right to pollute (Tobey and Smets, 1996), and the difficulty to prove specific injury and causality (Seymour *et al*, 1992; Franck, 1995). Adoption of this principle is also hindered by its frailties regarding political feasibility, perception of fairness and ability to address existing political and economic asymmetries between the parties involved and thus to provide environmentally effective pollution control (Fischhendler, 2006).

The polluter pays principle is an economic and a legal principle

The expanding evolution of the polluter pays principle has not increased its precision. Economics has initially closely informed the content of the PPP with the theory of internalisation of external costs (Pigouvian theory) (OECD Recommendation C(72)128, 1972, Point a.2). The rational behind this first approach was deeply liberal. As originally propound the PPP's purpose was to promote efficient allocation of resources, irrespective of distributive issues, and avoid trade and investment distortions as a matter of economic policy. Its economic version ('no-subsidies') pursued free market internalisation of the costs of publicly-mandated technical measures as preferable to inefficiencies and competitive distortions of governmental subsidies. The costs of environmental policy should be imposed on polluters rather than on taxpayers.

Though emerging as an economic principle and being often defined with explicit endorsement of the criterion of cost internalisation (e.g. Principle 16 of the Rio Declaration), the PPP evolved into a legal principle (Smets, 1993: 340; 'principle of environmental law-making' – Krämer, 1991:point 3) and a general principle of environmental order. From this legal nature follows the duty of the state to implement it, using its guidelines to underpin the allocation of the costs of environmental policy.

The PPP's legal and moral reading went over the economic assertion, and claimed the polluter should pay all the social and economic costs of their conduct (Gaines, 1991: 470-1). It was then vested as a liability principle which demanded pollution sources to pay for damages caused and abate pollution to socially and legally acceptable levels at their own expense.

Though the PPP remains an economic principle (a principle of allocative efficiency), it has also turned into a 'legal principle of (just) distribution of costs' (a distributive principle) (Krämer, 2005: 134). It allocates the costs to the polluter not only for efficiency but also for equity reasons (Pearson, 1994: 563), since the problem of cost sharing calls for equity as well as efficiency (OECD, 1975: 25). This legal ambiguity, which is often overlooked (Verhoef, 1999: 206-7), raises a set of issues, among which the fact that the principle helps to justify policy decisions – whatever the decisions are (Krämer, 2005: 134).

In the PPP allocation of environmental costs aims at efficiency according to a particular conception of fairness ('*ubi commoda, ibi incommoda*'). The importance assigned to cost allocation varies depending "upon the point in time in the evolution of the principle, the institution espousing the principle, the words one chooses to emphasize in any given formulation of the principle and the legal instrument being used to execute the principle" (Milne, 2003: 5). Such definitional imprecision leads to different outcomes in terms of allocative efficiency and equity for instance regarding whether the polluter should pay.

The polluter pays principles has followed an evolving pattern regarding cost coverage

The economic rationale in the PPP assigns high relevance to efficiency, which claims for prevention (providing incentive) rather than remediation (redistribution). The preventative dimension of the PPP was clearly emphasised by both the OECD (OECD Recommendation C(72)128**)** and the (at the time) EC (EC Council Recommendation 75/436/Euratom) in their earliest policy statement and is broadly agreed in the literature[14] to be the final objective of the principle (OECD, 2002: 2; Bugge, 1996). Over time costs covered by the principle have expanded, but costs of pollution control equipment or other preventative measures at individual facilities still lie at the very heart of the principle (OECD, 2002: 2; Bugge, 1996: 77).

But the principle has evolved from supporting mainly preventative instruments to be strongly associated to redistributive tools. This has reduced its usefulness as an underpinning for tax regulatory instruments. By the end of the 1980s, the OECD underlined the importance played by the principle in the implementation of financial instruments to support environmental policy programmes (OECD, 1989: 33-5).

Cost internalisation is not an end in itself but rather a means to reduce the waste of natural resources (efficiency) and force the ones who cause social costs to support them (justice). As a cost allocation principle and not a 'principle of compensation for damage' (OECD, 1975) the initial version of the principle only covered the costs of pollution control and prevention. However, this analysis was not officially adopted by the OECD (Milne, 2003: 16).

The evolution has been from a partial internalisation principle to a full internalisation principle (Smets, 1993: 345, 363) with unanimity over its lower limit and complacence with its upper limit. The limited version of the PPP works well as a remedy when actors are easily definable. Its enlargement, which has been associated with the emergence of diffuse pollution as a major environmental problem, made the principle more demanding (Bergkamp, 2001: 259) and stronger despite its multiple exceptions (Gaines, 1991: 485) and brought it closer to the Pigouvian theory.

[14] For instance, Melli, 1989; Butti, 1990; Gaines, 1991; Tarantini, 1990; Smets, 1993.

In the 1975 OECD Recommendation full internalisation was not inconsistent with the PPP and the inclusion of restoration costs was accepted (Pearson, 1994: 556-7). The OECD suggested a first enlargement of the concept OECD in 1992 (OECD, 1992b: Foreword, paragraph 1.1(c)). In the 1991 OECD Recommendation of the Council on the Use of Economic Instruments in Environmental Policy it suggested the internalisation of damages (OECD, 1991a). Posterior inclusion of restoration costs and costs of residual emissions brought the principle to a full cost internalisation principle (Rehbinder, 1993: 61, and Hoerner, 1998: 1203, understand the principle as not requiring full internalisation). The broad version of the principle was adopted in Principle 16 of the Rio Declaration on Environment and Development, adopted by the United Nations Conference on Environment and Development (14 June 1992).[15]

Inclusion of compensation for damage in the PPP was done, among others, in European Union documents, namely, in 1989, the Commission Proposal for a Council Directive on Civil Liability for Damage Caused by Waste (OJ C 251/3, 4 October 1989, 4), in 2000, the Opinion of the Economic and Social Committee on the White Paper on Environmental Liability (OJ C 268/19, 19 September 2000, 19) and the White Paper on Environmental Liability presented by the European Commission (COM(2000) 66 final, 9 February 2000, 13-14), and, in 2002, the Commission Proposal for a Directive of the European Parliament and of the Council on Environmental Liability with Regard to the Prevention and Restoration of Environmental Damage (COM(2002) 17 final, 23 January 2002, 3, 32, 34).

Both two latter documents, as well as the Commission's technical paper on the application of the polluter pays principle (European Commission, 1999: 5), imposed the payment of restoration costs on the polluter. The European Community has also associated environmental taxes with a version of the principle which includes costs of residual emissions and restoration, in the 2001 Communication on Tax Policy (COM(2001) 260 final, 23 May 2001) and the Proposal for a Directive on Special Tax Arrangements for Commercial Diesel (COM(2002) 410 final, 24 July 2002). The EU has endorsed the obligation of the polluter to pay also for preventative measures in the Waste Shipment Regulation (Council Regulation (EEC) 259/93, 1 February 1993, OJ L 30/1, 6 February 1993). In the Green Paper on Integrated Product Policy (COM(2001) 68 final, 7 February 2001, Point 4.1), the Commission also referred to the broad version of the principle.

1.3 The Least Abatement Cost Argument

Pollution taxes can also be underpinned by another economic rationale that follows a least abatement cost argument. Support is awarded to these instruments based on their feature of cost-effective instruments for achieving specific environmental goals. The 'least abatement cost' rationale develops along a coherent series of statements evolving around economic efficiency and leading from a premise to a conclusion. However, it lacks a body of theorems presenting a concise systematic view of a subject, being a collection of arguments (rather than a theory) leading to a single conclusion ('taxes allow pollution abatement at least cost') but taking different starting points and explanations to reach the same result.

The efficiency rationale underlying this argument is based on an understanding of the tax as a regulatory measure. Economic efficiency is expected from the behavioural change it induces. After considering their cost function and the tax rate, some producers will take pollution abatement as the most efficient option while others will not. Therefore, the first will change their behaviour towards a more economic efficient and environmentally preferable direction (structural change).

Each entity will choose the most profitable strategy for itself and such reaction will allow the desired level of pollution to be attained by society at a lower total cost (static efficiency).[16] By not imposing the same level of pollution abatement on every polluter regardless of cost effectiveness (uniform solution), the tax approach allows a least abatement cost strategy (aggregate change in behaviour via individual reactions

[15] "National authorities should endeavour to promote the internalisation of environmental costs and the use of economic instruments, taking into account the approach that the polluter should, in principle, bear the cost of pollution, with due regard to the public interest and without distorting international trade and investment".

[16] On the static efficiency of pollution taxes see *inter alia* Osculati, 1979: 51-54; Baumol and Oates, 1982: 167-173; Stewart, 1993: 2087-2088; EEA, 1996: 18; and Hanley *et al*, 1997: 111.

to price changes) (Buchanan and Tullock, 1975: 140-2; Ackerman and Stewart, 1985: 1341). The result expected is the achievement of set environmental objectives at minimum cost.

Moreover, a tax raised on pollution provides a *permanent and continuous* (González Fajardo, 1987: 169; Milliman and Prince, 1989: 247-65) economic incentive to develop more efficient means of addressing environmental problems (dynamic efficiency).[17] Therefore, environmental taxation is a good instrument to implement the ideology of ecological modernisation, since the latter is mainly concerned with finding more sustainable means for the development of the economic activity through technical innovation and new production methods (Barry, 2003: 200-1).

Another line of the same argument is that tax instruments can represent an especially efficient way of dealing with pollution due to the preventive action they induce (Opschoor and Vos, 1989: 9) and to the fact that implementation of economic instruments requires only a benefit analysis whilst administrative measures demand a cost benefit analysis (Posner, 1992: 378-9). The risk of inefficiency due to too restrictive pollution control objectives will, thus, be lower with taxes than with command-and-control measures.

The explanation for the use of pollution taxes vis-à-vis other instruments under this argument, i.e. a higher efficiency in the attainment of precise environmental goals, shall affect the tax design. All the decisions about the elements of the pollution tax should flow from the decision on the level of environmental quality desired (Surrey, 1973: 157; Rehbinder, 1993: 66). There shall be a coherent relationship between the governmental objective and the definitional nucleus of the pollution tax, namely the tax rate, the tax base and the identity of the taxpayers. These elements shall be chosen in order to allow the tax to achieve the desired environmental objective (Surrey, 1973: 160; Johl, 1997: 716). But no further explanation is provided regarding the way how this should be done.

This argument tries to make economic growth compatible with environmental protection as required by ecological modernisation. The double dividend argument (analysed next) follows the same logic, though it takes a broader, more systematic and elaborated approach. The least abatement cost argument emphasises the regulatory moment, trusting the capacity of the market to function efficiently as a result of the stimulus provided by pollution taxes. There is no precise statement concerning how to use revenues obtained with pollution taxes. The revenue raising function is secondary in the tax design (Rose-Ackerman, 1977: 399). The double dividend argument, on the contrary, emphasises the use of the pollution tax revenues and the consequent tax shift.

1.4 The Double Dividend Argument

A fourth rationale supporting the use of pollution taxes is the double dividend argument. This argument develops as a 'revenue neutral approach' to environmental taxation. It suggests that the use of environmental taxes can create an environmental dividend (first dividend) as well as an economic dividend (second dividend). Behavioural change induced by such taxes would produce environmental benefits and the application of their revenues in the reduction of pre-existent tax distortions would, in turn, yield an economic efficiency gain.[18]

This reasoning takes the improved efficiency of the tax system, which yields non-environmental economic benefits (Milne, 2003: 11), and the environmental improvement as equally important variables supporting the decision-making process leading to the adoption of pollution taxes. It justifies the use of these taxes but does not provide guidelines for their design, giving only references on how to use tax revenues. This might be considered the first systematic approach to the use of the revenues from pollution taxes, since, except for a scant approach by Pigou's work, "modern externality theory hardly deals with the question of what should happen to the revenues from externality taxes" (Andersen, 1994: 37).

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[17] On the dinamic efficiency of pollution taxes see *inter alia* Stewart, 1981: 1261, 1279, 1285-1286; OECD, 1996a: 12; EEA, 1996: 18.

[18] See for example Terkla, 1984: 108; Lee and Misiolek, 1986: 339; Pearce, 1991: 940-941; Repetto *et al*, 1993: 14; Goulder, 1995: 158; Oates, 1995: 916-918; Parry, 1995: S-64; Bohm, 1997: 110-111; Fullerton and Metcalf, 1998: 221; Bovenberg and der Ploeg, 1998: 395; De Mooij, 1999: 1-2; Parry and Bento, 2000: 67.

The precise tax design will depend on the rationale followed by the pollution tax itself, which can be full internalisation of external costs, achievement of a specific environmental objective or allocation of precise external costs. In any case, under the double dividend argument a pollution tax has to fulfil two specific conditions. On the one hand, it has to be able to raise revenues to support a tax shift, thus such revenues cannot be earmarked, and, on the other, it must deliver an environmental gain.

The double dividend argument expresses a political rationale vested in an economic rationale, with a normative, rather than positive, approach. The motivation behind this argument is maximising gains from public intervention whilst minimising its political costs, which is different from demanding an efficiency gain. The discourse used to pass to the public the idea behind the double dividend argument is, however, an economic one. It builds up on the behavioural impact of taxes raised on pollution coupled with the specific use given to their revenues. The latter is an essential characteristic of the double dividend rationale since the occurrence of the argued second dividend depends on such use.

The revenue capacity of pollution taxes is a central element of the double dividend debate, whilst the level of cost internalisation performed by the tax has only a second rank importance. The Pigouvian tax is adopted as the starting point by several of the models testing the double dividend hypothesis (e.g. McCoy, 1997: 203; Goulder, 1995: 172-173; De Mooij, 2000: 14). However, there is not a theoretical objection to the use of less than full cost internalising taxes to pursue double dividend policies (e.g. OECD, 2000: 19-27). In fact, the double dividend argument is usually used by politicians regarding taxes raised on polluting bases which are aimed at a long-term process of market price correction, i.e. taxes that aim at steering economic choices via price signals without full cost internalisation.

The introduction of this rationale in the environmental taxation debate seems to be closely linked to the socio-political context where it is used. Environmental tax reform or tax shifting is a common context framing the discussion about the use of environmental taxes and the double dividend argument often arises within this debate (as explained in Section 4 below). This linkage made the use of this argument common practice by national governments (OECD, 2000: 51-52) and the OECD, in spite of some resistance from professional economists to the argument[19] voiced in some suspicion of the OECD itself (OECD, 2001f: 37-39).

[19] See for example Bovenberg and Mooij, 1994: 1087-1088; Bovenberg and Goulder, 1997: 60, 78, 2002: 1502; O'Riordan, 1997: 77; Flip de Kam, in OECD, 2003b: 20.

Table 1.1 - Summary of economic theories underpinning pollution taxation

Economic theory underpinning environmental taxation	*Basic concern*	*Environmental goal*	*Tax base* *Tax rate* *Tax subject*	*Tax revenue*
Pigouvian theory	Private versus societal allocation of external costs	Optimal welfare	Activity causing negative externalities Marginal costs of the environmental damage Externalities producer	Open-ended
Polluter pays principle (OECD, 1972)	Private versus governmental allocation of external costs Avoid international trade distortions Encourage efficient use of scarce natural resources	Full internalisation of external costs in the long-run	Polluting activities Tax rate set according to required&feasible Economic agents with decision power	Open-ended
The least abatement cost argument	Cost effective environmental protection	Level of pollution externally selected	Elements chosen in coherence with the public goal set	Open-ended
Double dividend argument	Improved efficiency in the tax system	Improvement of environmental conditions	Open-ended	Revenue recycling to reduce general tax burden

Adapted from Milne, 2003: 23.

2. INSUFFICIENT POLICY GUIDANCE TOWARDS ENVIRONMENTALLY EFFECTIVE TAX DESIGN

The least abatement cost argument and the double dividend argument are a series of statements evolving around economic efficiency and leading from a premise to a conclusion but lacking a systematic view. Therefore, among the main theoretical framework underpinning pollution taxes, only the Theory of Internalisation of External Costs (Pigouvian theory) and the normative doctrine of environmental law posed by the polluter pays principle could be useful to policy design since only they are close to be a theory, i.e. a group of general propositions used as principles of explanation of the integrated whole that is environmental taxation.

Together with the tax rationale, which follows from the objectives and criteria ruling the tax design and corresponds to the definitional content of the mentioned theoretical underpinnings themselves, other elements are paramount to the environmental tax design, namely the amount of the tax payment, the identity of the taxpayer, the admissibility of especially favourable regimes and subsidies to polluters and the application of the tax revenues. The Pigouvian theory and the polluter pays principle provide vague and at times contradictory guidelines regarding these other definitional elements of environmental taxes. Therefore, the conclusions both theories allowed regarding such definitional elements were insufficient to lead policy-makers surely towards environmentally effective tax instruments (i.e. regulatory pollution taxes).

The joint application of the PPP and the Pigouvian theory is also not sufficiently clarifying to provide policy guidance. These theories provide different solutions on the level of cost internalisation, for instance, regarding the need for full internalisation of emissions below the assimilative capacity of the environment and payment for pollution damages. Likewise, there is not an unequivocal identification of the individual who should pay. Charges on pollutees are rejected under the PPP but accepted based on a theory of efficiency as the Pigouvian one. Especially favourable treatment to polluters is also differently accommodated by each of the theories. And, in contrast to the double dividend argument, both theories tolerate several revenue uses but do not suggest a particular one.

Assessed next is the guidance provided by the PPP and the Pigouvian theory regarding each of the definitional elements of environmental taxes in order to demonstrate their insufficient policy guidance for good institutional practices in pollution taxation. Following their analysis it remains unclear at which amount the tax rate should be set and who should pay, as well as whether and how especially favourable tax regimes should be assigned to polluters and which revenue uses are suggested.

2.1 The Amount of Tax Payment

Referring the tax level to specific but unspecified environmental objectives is equivocal

Cost allocation in the PPP does not ultimately aim at internalising costs or assigning blame for environmental harms but rather at fostering environmental protection via increased awareness and shared responsibility (likewise on environmental taxes, Duff, 2003: 7). From an international perspective, trade distortion avoidance is also a relevant motivation supporting cost allocation. Full cost internalisation might not be necessary to achieve any of these objectives. But there seems to be an agreement over the inclusion of costs of measures to reduce (OECD, 1989: 27), prevent and control pollution (OECD, 2001f: 16), as well as costs of environmental cleanup and associated monitoring and compliance costs. However, precise and uniform guiding rules are not available leaving the issue to political decision.

On top of the evolving nature of the Polluter Pays Principle and its definition of costs covered, making the costs to be allocated to the polluter dependent on the specific environmental goal set by public authorities further hindered the policy guidance capacity of the principle. According to the version of the PPP in the 1972 OECD Recommendation and the 1975 European Community Recommendation (European Council, 1975: 2), the tax level should be referred to specific environmental objectives and keep a connection with the amount of pollution (European Commission, 1975: Point 4.b). Ideally, internalisation should be in the amount necessary to reach the behavioural response able to fulfil the intercalary objective aimed by the regulator (e.g. a certain level of emissions control).

This provided insufficient policy guidance because it was not specified to which publicly-mandated pollution control measures referred the cost of compliance polluting sources should bear. It was up to public authorities acting in the society's interest to set such standards. Using open concepts to indicate the level of payment authorities should pursue, of which the most relevant has been 'acceptable state' of the environment (Points a.3 and 4 of Annex to OECD 1972 Recommendation), did not bring further precision to the principle. Specific objectives could then still lead to different tax rates for instance depending on whether the tax was keyed to achieve a precise environmental quality standard at the individual economic agent level, at the aggregate level of the whole industry or at the level of the relevant ecosystem.

Referring the tax level to specific environmental goals differs from imposing on the polluter an amount with reference to a precise regulatory function, consisting of damage prevention, remedying or compensation. It might be necessary to raise a tax either higher or lower than prevention/control/remedy costs to attain the desired environmental quality level, for instance due to the interaction between the tax and other pre-existent regulatory instruments (Smith, 1999: 509-511). Compliance with the PPP should be inferred whenever the tax itself is "the means of achieving the specific goal with the guarantee that the polluters are paying the cost of meeting that goal, even though the cost of meeting the goal may be higher than the actual marginal prevention and control costs assumed by those who do not pay the tax" (Milne, 2003: 16).

Multiple competing levels of cost internalisation

According to the polluter pays principle, and following its expanding interpretation, there are several categories of costs that can be imposed on the polluter. These can be costs of pollution control, prevention and restoration, as well as the associated administrative costs and the costs of residual emissions. The principle also pursues the apportionment of administrative and other ancillary expenses of a public environmental protection system, to the extent that such costs can be linked to pollution control and preventative measures (European Council, 1975: 3; OECD, 1975: 25).

Concerning the amount of potential costs to allocate to the polluter, there are three main categories possible: 1) costs of pollution control and prevention, either a) costs of pollution control and prevention at individual facilities, or b) costs of collective measures on behalf of a whole group of polluters, 2) costs of pollution control, prevention and restoration of damages not prevented, and 3) all of the costs just mentioned plus the costs of any residual damage resulting from less-than-full prevention and control. In all mentioned hypotheses, the associated administrative costs are included as well.

The kind of costs allocated to the polluter, as well as the instruments used to perform such allocation, are influenced by the debate on the identification of the individuals to which the costs might otherwise be allocated, i.e. 'the polluter should pay instead of whom?'. The answer to this question depends on whether the costs are being shifted from the government or the society as a whole (Milne, 2003: 8-9). On the first hypothesis, which requires a lower cost allocation than the second, and going from its narrowest to its broadest version, polluter payments can replace a) direct governmental subsidies to polluters or b) public expenditure expressed in environmental actions taken by public authorities at taxpayers' expenses.

If polluters should pay instead of the government, only pollution prevention, control, and reparation costs can be imposed, and such cost allocation requires the previous existence of a legal duty to take pollution preventative or remedial action. On the second hypothesis, polluter payments substitute implicit societal subsidies arising from individual use of the environment. Under this approach polluters can also be forced to take the costs of residual emissions, as well as the costs of levelling the playing field for environmentally friendly products (European Commission, 1993), either by means of economic instruments or, for instance, regulatory requirements and liability regimes.

Under the narrowest version of the first hypothesis the polluter pays principle does not require pollution taxes but withdrawal or forbearance of direct governmental payments, since the idea is to make the polluter pay instead of getting a subsidy. The broadest version of the first hypothesis can be pursued by tax instruments of either a fiscal or a regulatory nature. A fiscal tax can implement the principle as a revenue recycling mechanism, compensating for public expenditure caused by the polluter. For instance, due to technical reasons, it might be more efficient to develop a publicly financed multimunicipal system of wastewater treatment than to require the investment in individual cleansing systems. In such context a tax can be used to comply with the principle if it is raised on polluters according to the costs they cause and earmarked to finance the public system.

Joint application of the polluter pays principle and the Pigouvian theory is insufficient

Both the Pigouvian theory and the polluter pays principle take cost internalisation as a means to attain increased efficiency. Efficiency, expressed in welfare maximisation in the first case and fewer international trade distortions and better use of natural resources in the second, is expected as a result of behavioural changes induced by external costs allocation. Such different understanding of efficiency leads to different tax designs.

Only the broadest version of the polluter pays principle (the one requiring the polluter to pay instead of the society as a whole) allows full internalisation of pollution costs, since only that version allows the allocation to the polluter of the costs of residual emissions (i.e. restoration of damages not prevented plus the costs of any residual damage resulting from less-than-full prevention and control); whilst the Pigouvian theory always requires full internalisation, since it aims at maximizing total welfare, requiring an accurate allocation of external costs. Unlike the Pigouvian theory, the PPP does not impose a pure cost internalisation philosophy and neither does it require bringing pollution to optimal levels (Gaines, 1991: 473). Therefore, also unlike the Pigouvian theory, it does not require payment for pollution damages.

'Pollution' is understood within environmental policy as a use of environmental resources which at the margin has higher costs than benefits for the society. Reduction of environmental damage is only socially desirable when it increases pollution victims' welfare more than it reduces polluters' welfare (Revesz, 1997: 3). This means pollution control is only socially desirable when it involves a welfare gain. Polluting emissions below the assimilative capacity of the environment do not involve external costs (Pezzey, 1988: 196-242; Hahn and Noll, 1990: 359; Hodge, 1995: 91). Therefore, the Pigouvian theory does not require their internalisation. However, the polluter pays principle in one of its versions requires the polluter to pay also for residual pollution.

Understanding the polluter pays principle as not intended to punish but avoid future pollution leads to setting the amount paid by polluters with reference to prevention requirements rather than environmental damage caused. However, a broad economic theory of environmental protection, such as the Pigouvian one, which considers cost internalisation a technique for maximisation of social welfare, requires payment for pollution damages. Such maximisation from an economic perspective does not require zero pollution but rather the equalization at the margin of social benefits and social costs. For economists the ideal level of pollution is not zero but the one maximising social benefits. This requires economic agents to decide on their pollution level taking into consideration not only prevention and control costs but also costs of pollution damage.

The Pigouvian theory suggests the tax should be levied at a rate equal to the marginal environmental cost caused by the activity (Pigou, 2002: 193). However, such a theoretical model is difficult to implement in practice due to incomplete information. To overcome this problem, some have suggested an iterative process ('trial and error approach') attending to price elasticities, where the tax would be set at a certain level and then progressively adjusted according to the effects obtained until the desired pollution level was reached (Baumol and Oates, 1971: 43-46, 1982: 163-164). This is still a vague guideline since the desired pollution level is left up to the decision of the public authorities without any restrictions.

2.2 Who should pay

Regarding the identity of the taxpayer, clear guidelines from the theory are also missing. The 1975 European Council Recommendation (75/436/Euratom, Point 3) defined 'polluter' based on an economic rationale, with reference to two criteria, namely economic and administrative efficiency and ability to internalise the costs. The 'polluter' should be the person who directly or indirectly damaged the environment or created the conditions leading to such damage. In a pollution chain, the 'polluter' should be the 'best-payer', i.e. the point of the pollution chain easier to control, with lower resistance to the charge and best able to provide effective pollution control and easier-to-avoid market distortions.

Identifying the polluter with the 'best-payer' allows pollution prevention when acting with a restricted number of 'polluters' who are in a privileged position to avoid environmental damage. Furthermore, the result from applying such a rule might be unfair. For instance, big polluters might be totally or partially exempted from paying the costs caused due to administrative charging difficulties or their strong power to oppose the charge. The OECD position did not lead to increased environmental effectiveness or fairness, since this organisation has taken a neutral stance on pass-through of costs from the polluter to its customers (OECD, 1975: 239).

The efficiency criterion provided by the Pigouvian theory does not help further. The payer should be the one whose charge leads to efficiency. If pollutees can avoid pollution at a lower cost than polluters the former should pay rather than the latter to stimulate the adoption of pollution control. This might involve, for instance, the relocation of the pollution victims (Kolm, 1973: 323 and 334-335). If it is not obvious who incurs the lowest cost to avoid pollution, the payer should be the one best able to negotiate (Coase, 1960: 28-39; Michelman, 1971: 649-658; Ellickson, 1973: 725). However, this rule is unfeasible in most cases due to the high number of subjects involved.

2.3 Especially favourable regimes to polluters

While scholars disagree in their approach regarding the amount of cost internalisation, most agree that the cheapest way to achieve objectives of environmental policy is through a uniform tax, i.e. a tax imposed on all the polluters according to the same criteria without especial regimes (Speck, 2008: 36; NERI, 2007: 22). However, vis-à-vis this theoretical vagueness institutional practice has often relied on special treatment provided to the individuals most affected by the tax to set tax rates at levels which are high enough to be effective (Hoerner and Muller, 1997: 153; Määttä, 1997: 178). This practice has been justified based on political reasons ('potentially big taxpayers' have enough power to block or disturb the implementation of the tax), equity reasons (the tax has a regressive impact on income) and/or economic reasons (important national economic sectors competing in the international market are badly hit by the tax). Exemptions and mitigation regimes have been especially relevant in energy taxation due to the socio-economic impact of energy prices.

The Pigouvian rationale allows subsidies to polluters that fulfil economic purposes similar to the ones reached through pollution taxes. In the short or medium term, the economic rationale expressed in maximisation of social welfare is compatible with the assignment of subsidies (Alder and Wilkinson, 1999: 184). From an economic perspective, the optimal level of pollution is not zero and subsidising the polluter can help the society reach the point where social benefits equal social costs. However, in the long term, this might not be the case due to expansion of the polluting industry induced by the attractiveness created by the subsidy.

According to the polluter pays principle, to make someone else other than the polluter pay for pollution should be the exception rather than the rule and therefore only admitted under very strict conditions (Gaines, 1991: 476). These might be socio-economic reasons (e.g. Art. 192(5) TFEU) or limits to the principle's rationale itself (for instance, European Council Recommendation 75/436, Arts. 6 and 7). Vague guidelines have allowed each European Union country to develop its unique re-interpretation of this principle to justify its subsidy schemes as compatible with it (Gaines, 1991: 479).

Colliding public interests may justify exceptions to the PPP if its application leads to higher social costs than its waiving. Avoidance of trade distortions (environmental dumping), provision of more rigorous treatment of new firms, concerns with national competitiveness, employment and income distribution impact are the most commonly accepted arguments. Likewise, arguments related to income distribution impact (regional development interests) are common in countries with asymmetric regional development and strong energy needs, like for instance Sweden, to justify exceptions to energy taxes. The 1975 European Council Recommendation accepted exceptions whenever (1) full application of the principle could cause a severe economic crisis and (2) contradiction of the principle resulted in indirect benefits from other common policies.

Restrictions to the principle are accepted under some conditions. They should be selective, transitory, applied within clearly defined periods of time and conditioned to the adoption of some kind of effort by the beneficiaries to comply with the principle ('efficient mitigation measures' – OECD, 2001g: 72). Moreover, they should cover only partially the costs caused by the polluter (tax reductions rather than tax exemptions) and not be successively prorogued. Furthermore, any limits to the polluter pays principle should be necessary, adequate and proportional to the interests on which they are based.

Apart from these general rules, there are some others set in OECD documents (such as Recommendation C(74)223) and, with substantially more detail, in Community guidelines. The EU legally binding body, which has been directly influenced by the OECD *soft law*, builds on the general regime for State aid established by Article 107 TFEU and has been developed in several documents, the most relevant being COM(97) 9 final (26 March 1997), and 1994, 2001 and 2008 Community guidelines on State aid for environmental protection (European Commission, 1994, 2001, 2008). Procedural rules can be found in Article 103 TFEU and some OECD and EU documents.

2.4 The application of tax revenues

There are no clear guidelines in the theoretical framework underpinning pollution taxes on how to use the amounts paid by the polluter. Revenues from pollution taxes can be either channelled towards the public budget, earmarked to special expenditure programmes or recycled back to the payers. Political programs endorsing the double dividend argument often associate pollution taxes with revenue recycling programmes to lower labour and capital taxes, as well as to environmental projects.

In the meeting of OECD Environment Ministers in 1974 there was some debate on earmarking. The German delegation proposed earmarking, whereas other participants understood the latter did not need to be associated with the polluter pays principle (OECD, 1975: 72-3, 80-1), reflecting the traditional resistance from economists to such techniques. The second position has prevailed in the end, but the use of a 'principle of revenue recycling' as complementary to the principle has been suggested (OECD, 1977: 52).

The European Council's 1975 Recommendation Regarding Cost Allocation accepted the use of revenues from charges reflecting the polluter pays principle to finance collective measures taken by public authorities and suggested that the remaining amounts in excess of the necessary to cover such measures should be diverted to environmental projects. By carefully defining the circumstances in which the income from charges could be used to finance private pollution control measures (p. 3), the European Council implied though that charges dedicated to the general support of private pollution control did not conform to the principle (Gaines, 1991: 475). Since its foundation the principle has continuously banned any use of revenues to subsidise prevention and control measures imposed on the polluter.[20]

The European Commission tends to associate the use of pollution taxes with a tax neutral approach via the double dividend argument (for instance in COM(2001) 260 final, 13). This position supports dissociation between the polluter pays principle and earmarking since it requires the freedom of public authorities to decide on the use of tax revenues. There is a unidirectional relationship between the two concepts rather than a bidirectional one. Dedication of resources can be compatible with the principle (Vos, 2002: 284), but the latter does not require earmarking.

Earmarking might serve efficiency concerns in what would be an application of the theory of internalisation if revenues paid by polluters were allocated to repair the damages they caused to the society. Pigou endorsed the kind of earmarking that aims at repairing the damage caused by the taxed behaviour rather than at funding new projects or compensating victims. For instance, he considered the British road tax a "very incomplete and partial" application of the theory of internalisation since the revenues were used to build new roads rather than to fund reparation of damages caused by drivers (Pigou, 2002: 193). Thus, drivers obtained from their payment an additional service useful to them rather than to the general public. Economic efficiency would have required road pricing revenue to be used to benefit society rather than refunded to users in proportion to how much they paid.

3. Theoretical vagueness led to misapplication of 'Good Theory'

One of the dangers that could potentially arise from converting economic theory into legitimate economic policy is misapplication of 'good theory' (Christoffersen and Svendsen, 2002: 361). This as well as delayed application[21] can occur following a gap between theory and reality due to either (in principle good) economic theory that is non-operational by nature or the ignorance of facts.[22] Theories underpinning pollution taxes were non-operational. Sometimes they were unfeasible (e.g. full cost internalisation suggested by the Pigouvian theory) other times they did not put forward clear criteria for bureaucrats to design and identify environmental taxes. This incompleteness allowed their misapplication. Real-life politics set the actual design of these taxes.

Furthermore, the environmental economic theory proposal was just a starting point (Barthold, 1994: 145; Hahn, 2000: 376). Environmental economists in general paid little or no attention to the institutional context within which pollution taxation was to be introduced (Baumol and Oates, 1982: 157; Crandall, 1983: 69; Helm and Pearce, 1990: 14; Andersen, 1994: 33, 1995: 63). However, the set of formal and informal rules surrounding the practical use of pollution taxes operates as a constraint to the optimal/correct application of the concept of environmental tax built by the theory (Andersen and Sprenger, 2000).

Stakeholders intervening in the political decision-making process were able to use the gap between theory and reality to develop rent-seeking behaviour, which led to (good) theory misapplication. Among other

[20] For example, OECD's 1972 Recommendation; European Council's 1975 Recommendation; European Commission's 2001 Guidelines on State Aids.

[21] On delayed application of theory see for example Schelling, 1983:xiii; Mendelsohn, 1984: 202; Frey, Schneider and Pommerehne, 1985: 62; Opschoor and Vos, 1989: 127; Hahn and Stavins, 1991: 42; Smith, 1992: 52.

[22] On the gap between theory and reality see for example Krueger, 1997: 11; Christoffersen and Svendsen, 2002: 361; 'regulatory gap between researchers and legislators' – Schelling, 1983:xiii. Also Segerson, 1988: 96; Cordes, Nicholson and Sammartino, 1990: 343; Määttä, 1997: 15.

possibilities, a bureaucracy could construct an ideology dictating public sector growth and budget maximization (Christoffersen and Svendsen, 2002: 361). Externalities were promoted as new ideologies (Hajer, 1995: 26-27). Goals, expectations and actions in public matters were referred to them, using economic theory to legitimate economic policy. Once the new ideology had been established, empirical facts were ignored (Christoffersen and Svendsen, 2002: 361). This might explain why the incapacity of pollution taxes to deliver environmental results has been broadly ignored.

Actual design of economic instruments in general typically departs dramatically from their 'efficient' design.[23] This risk was noticed early on (Pigou, 1947: 99-100). However, problems raised by public choice in the late 1940s were not considered in the first version of the theory of internalisation of externalities, which assumed a dispassionate calculus of external costs. Pigou saw government as "a beneficent exogenous force, unaffected by special-interest demands for government favors" (Yandle, 2001: 593). But he ended up concluding corrective taxes could never be applied effectively in the real world, due to considerable administrative costs (Pigou, 1947: 99-100). Pigou's criticism of the bureaucracy included scepticism of the abilities of local authorities to undertake effective market intervention, due to lack of professional competence by politicians, disparity as far as the territorial dimension is concerned between the regulatory power and the evolving optimal operational base of public utilities, a short-time horizon for decision-making and vulnerability to pressure groups (Andersen, 1994: 38).

Overlooking the imprecision which actual application of the theory involved made it vulnerable to misapplication. Therefore, precise guidance regarding how to feasibly design environmentally effective taxes is especially relevant. For instances, establishing the rate of pollution taxes has been an 'imprecise process' (Surrey, 1973: 160) due not only to the influence played by the regulated agents in the decision-making process but also to difficulties associated with the measurement required (Pigou, 1937: 42-4; OECD, 1992a: 59-60; Weizsächer and Jesingham, 1992: 23).

According to economic theory, pollution taxes should only be based on the marginal environmental damage (marginal pollution damage at the socially optimal level of emissions – Smith, 1999: 509) and should not be manipulated attending to government revenue considerations or even to income distribution concerns (Proost, 1999: 335). This condition is hard to comply with in real world policy instruments. Often taxes are used to raise revenues rather than reflect optimal damages (Barthold, 1994: 136; Hahn, 2000: 506).

Moreover, *ex post* evaluations of the effects obtained with pollution taxes might also be complex due to difficulties associated with both the selection of a baseline and the use of measurement processes (Speck and Ekins, 2002: 88-9). On the one hand, there is lack of data on measurement of environmental benefits from pollution taxes (OECD, 2001g: 99). On the other hand, analysis of existing data can sometimes be ambiguous. And taxes are often part of a policy package hard to disentangle, thus creating difficulties to the individual assessment of each tax instrument (EEA, 1996: 9).

Furthermore, addressing multiple issues with the same instrument not only harms the effectiveness of the instrument but also prejudices the assessment of such effectiveness (EEA, 2000b: 4). However, a mixture of functions can often be observed in practice (EEA, 1996: 8). For instance, according to the European Union, the most statistically representative pollution taxes, energy taxes, have as central aims: simultaneously securing energy supply, protecting the environment and maintaining international competitiveness (Welfens *et al*, 2001: 79).

Another aspect to consider is that any deviation from 'good theory' can be successfully hidden following the time lag between the regulatory intervention and the occurrence of the effects. In the case of energy taxes the time span is said to be 10-15 years (EEA, 1996: 9). The degree of effectiveness reached by the instrument can increase over time. For instance, the regulatory energy tax adopted in 1996 by the Netherlands is considered to have become more effective over time, being able to decrease gas and electricity use at an increasing rate as time went on (Holzinger, 2003: 197).

The double dividend argument exemplifies the contribution of the literature to the misapplication of 'good theory'. Environmental taxes as instruments of environmental policy are legitimised based on

[23] Authors mentioning this problem include for example Buchanan and Tullock, 1975: 139-147; Hahn and Hester, 1989: 361-406; Barthold, 1994: 133-151, McMorran and Nellor, 1994: iv; Maloney and McCormick, 1997: 231-236; Keohane, Revesz and Stavins, 1999: 89-125; Hahn, 2000: 392.

environmental benefits. By diverting the citizens' attention to the double dividend argument, the emphasis is put on raising revenues rather than on obtaining environmental results. This misleads the citizens and their scrutiny of the policy measures. In the absence of a double dividend discourse (ibidem and Bovenberg and Goulder, 1997: 60), environmental tax interventions have to be actively justified on the basis of environmental benefits.

4. EXPLAINING THE CONCEPTUAL EVOLUTION

The way stakeholders' ideas and interests have translated into a selective use of theories explains the current understanding and use of environmental taxation. Contributions from reference stakeholders, such as the OECD, European Union institutions, government commissions and epistemic communities have influenced how policy ideas and economic theory translated into political proposals. The transference process, strongly informed by ecological modernisation (which "has become an embedded part of the discourse of environmental policy elites and policy-making" – Barry, 2003: 199), was highly creative due to lacunae in the theories themselves, culminating in 'environmental tax reform' and 'environmental financial reform' movements. These were shaped by 'political-costs-reduction' concerns and only weakly influenced by environmental engagement.

4.1 The idea of green taxation

Since the late 1960s, environmental policy-makers in most Western countries have been experimenting with various policy instruments and looking for workable policy ideas and solutions. The green tax epistemic community (Daugbjerg and Pedersen, 2002) gave its input into the process by recommending the use of charges as the best solution to deal with environmental damage. This community was comprised mainly of environmental economists, who tended to dominate the debate and were bound together by a common policy enterprise, i.e. to develop environmental instruments that could effectively reduce pollution at the least cost (efficiency) (ibidem).

This idea gained a relevant political dimension with the OECD definition and recommendation of the polluter pays principle (PPP), according to which the polluter bears the expenses of carrying out the measures laid down by public authorities to ensure that the environment is in an acceptable state (OECD, 1972). As several windows of opportunity opened up, progressively ideas diffused by the community throughout the Western world were seriously considered in many government circles and translated into real world practice since the early 1970s. However, it was not until the late 1980s that the concept made its first serious international breakthrough.

A greater use of taxes to deal with pollution was driven not only by the search for more effective and efficient environmental policies but also by other factors, namely regulatory and fiscal reform moves and public revenue interests (OECD, 1998: 15-16). The diffusion process was also facilitated by failure of command-and-control measures to control pollution, increasing environmental pressure and citizens' demand for environmental quality (Weale, 1992: 167-170; Young, 1993: 53-56), as well as exhaustion of traditional tax bases, such as labour and capital.

This evolution was not least an effect of developments within two major currents of academic thought, namely the mainly American and British dominated field of environmental economics and the more political philosophical thoughts of ecological modernisation originating primarily in Germany (Daugbjerg and Pedersen, 2002), both of which contributed actively to the wider acceptance and vague understanding of environmental taxation experienced in the middle of the 2000s.

Based in conventional mainstream economics, American and British environmental economists have stressed the importance of supporting environmental action on a proper cost-benefit analysis. These studies contributed to change the perception of environmental taxation from being an instrument of making polluters pay for environmental cleanup or protection measures to a possibly more efficient market-based instrument of environmental regulation. Regulatory mechanisms like taxes were considered useful to drive the process of industrial innovation “with environmental and economic gains realised as a result” (Murphy and Gouldson, 2000: 43).

Together with the argument of greater value for public money spent on environmental protection measures, this development introduced a double dividend argument with possible additional benefits to be derived from environmental taxation. While most taxes distort incentives, environmentally-related taxes correct distortions, namely externalities arising from excessive use of the environment. It was argued that if these taxes’ revenues were used to finance reductions in other incentive-distorting types of taxation, such as labour supply, investment or consumption, secondary gains could be created in addition to environmental benefits (ibidem).

The ecological modernisation paradigm, which has contributed to collective action towards the adoption of pollution taxes, has been developed mainly within the German and Dutch contexts (Weale, 1992: 79-88, 125-36; Barry, 2003: 191). In spite of the several understandings attached to the concept of ecological modernisation (Young, 2000: 1), it was commonly agreed that this strategy simultaneously aimed at the improvement of ecological and economical efficiency (Jänicke, 1988: 23).

It was argued that environmental protection and economic prosperity proceeded “hand-in-hand” in a positive-sum game (Hajer, 1995: 64; Dryzek, 1997: 145; Susskind, 2002: 291) and environmental policies were presented as a ‘win-win’ scenario (OECD, 1985: 10; Gouldson and Murphy, 1997: 74). The ecological modernisation rationale focused on reducing resource consumption and generating less waste, while creating employment and improving economic welfare (Gouldson and Murphy, 1997: 75). Debates arising from this conceptual approach, like the double dividend argument, represented a return to the cost-benefit analysis of environmental policy (Andersen and Massa, 2000: 340; Barry, 2003: 206).

Ecological modernisation implied partnership and participation (Young, 2000: 13 – ’more inclusive approaches’): “government, businesses, moderate environmentalists, and scientists co-operate in the restructuring of the capitalist political economy” (Dryzek, 1997: 144). In this regard economic instruments, such as taxation, provided the most powerful means of regulating pollution in a market economy, ensuring economic efficiency and forcing the development of cleaner technologies. Later a social economic policy of Ecological Tax Reform (ETR) was also to be associated with German ‘ecological modernism’ (Andersen and Massa, 2000: 341).

In the 1980s, the ideology of ecological modernisation became appealing to many members of the policy elite in European countries (Weale, 1992: 75-9; Barry, 2003: 194) and international organisations such as the World Commission on Environment and Development and the OECD. It gradually attained a degree of societal consensus both within some Member States, such as the Netherlands, Sweden, Denmark and Germany (Lundqvist, 2000: 21-32; Andersen and Massa, 2000: 339; Barry, 2003: 194), and EU institutions.

Its attractiveness was associated with its potential to break the political stalemate between environmental quality advocacy and economic feasibility advocacy coalitions (‘attempt to green capitalism’ – Barry, 1999; Young, 2000: 2). The balance of argument in terms of economic feasibility simultaneously tipped towards environmental protection (Weale, 1992: 78-9). This was a ‘supply-side approach to environmental policy’, which attempted to circumvent, downplay or avoid issues of social or distributional in/justice and in/equality, and considered the regulation of demand for goods as of secondary importance. (Barry, 2003: 201).

The importance of designing environmental taxes able to change consumption habits might consequently have been downplayed. The incapacity of ecological modernisation to articulate the full range of normative issues relating to social-environmental affairs (Barry, 2003: 208) might explain other aspects of real world pollution taxes. Broad acceptance and use of especially favourable regimes for intensive energy users as well as the placement of heavier energy tax burdens on households than on the industry, both usually motivated by international competition issues, raise equity issues. These were seldom addressed in political programs.

During the 1990s, ecological modernisation was embraced in milestone EU environmental and economic documents (for instance, the European Commission White Paper on economic growth, competitiveness and employment (COM(1993) 700 final, 5 December 1993); and the European Commission 1997 proposed Directive for Energy Taxation in the EU (COM(1997) 30 final, 12 March 1997)). This approach was maintained in the Council Integrated Guidelines for Growth and Jobs (European Council, 2005: 5 and 11).

Unlike the PPP, ecological modernisation might be considered an 'overarching set of ideas' which reduces the complexity of understanding the world "by giving short-cuts and schemata of interpretation which can help actors to order and evaluate information" (Braun, 1999: 15). This ideational dimension of politics, as 'a process of meaning which may subsequently engender choices' (Braun, 1999: 12), might help to explain the predominant fiscal feature in real world pollution taxes.

Ecological modernisation, which was the ideological wrap involving the resurgence and development of the 'green taxation' idea, was a lost opportunity to strengthen in the pollution tax design the preventative rationale emphasised by the PPP. None of the two political systems playing an important role in the development of ecological modernisation, i.e. Britain and Germany, was able to present environmental taxation as a determinant tool in the development of preventative environmental policies. In Britain the government resisted playing a steering role in the economy, and in Germany particular emphasis was given to other means of promoting environmental quality rather than to regulatory taxes.

4.2 Main driving forces shaping the concept

The ecological modernisation discourse signals a shift away from traditional bureaucratic top-down dirigisme (Mol, 1995: 362-3) to open-ended discussions between 'variegated set of interests' (Weale, 1992: 32), which amounted to "regulatory negotiations" (Weale, 1992: 175). Acknowledging that some environmental problems were manifestations of state failures created new opportunities for private stakeholders (such as industry and NGOs) to influence the development of policy ('heterarchy', Smismans, 2006: 3). This openness engaged a multitude of stakeholders in the diffusion of the idea of environmental taxation.

Ecological modernisation "has emerged by means of evolution as a societal practice that is *beneficial to all parties involved*" (Blühdorn, 2000: 200). And environmental regulation is a potential vote-winning issue (Barry, 2003: 211; Cole, 2000: 21). This context prone to the accommodation of multiple interests influenced how the idea of environmental taxation became part of national policy-makers' 'toolbox' as specific policy. The OECD, EU institutions and Scandinavian countries were among the major players in this transference process.

The fiscal crisis of the Scandinavian welfare states opened a window of opportunity for pollution taxes to enter the political agenda. For welfare states under fiscal strain, these instruments represented a substantial asset, since their revenue capacity allowed the reduction of strangling income tax rates without much public resistance. Thus, the greater fiscal pressure experienced in these countries compared to other European countries helps to explain the emergence and importance of these taxes in Scandinavia (Lohman, 1994: 56; Andersen, 1995: 56; Henderson, 1996: 106).

But in the process of securing new sources of income for the welfare state, some of the initial principles of environmental taxation were lost (Andersen, 1995: 58, 62). Two main factors diverted the measures adopted from the literature proposals, namely the fiscal focus of the discourse proposing the use of pollution taxes and the decision-making process, in which economic and political interests had a say over the final design of pollution taxes (OECD, 1995: 17).

Relevant epistemic communities

In the 1970s, using taxation as an instrument of environmental policy was a new idea in the political discourse and in institutional practices. The OECD relevantly played the diffusion role regarding this new idea whilst promoting the implementation of pollution taxes. In practice this culminated in the development of two instrumental ideas: 'policy integration' and 'environmental tax reform' (ETR). The latter being a development of the first.

Comprehensive restructuring of the tax system to achieve environmental objectives (environmental tax reform, hereafter also 'ETR') and the adoption of new taxes to deal with newly identified environmental issues or to replace or complement existing regulation became the two main approaches to environmental taxation. ETR commonly involves restructuring traditional taxes (such as energy and transport related taxes) and subsidies, making then more compatible with environmental goals. Policy integration together with the ETR discourse contributed to a vague concept of 'environmental tax', since the 'ETR' is not a simple or unequivocal concept, many 'wrong' ways of pursuing it might have damaged the economy and further reduced public confidence in the tax authorities (Weizsäcker *et al*, 1998: 204).

In 1996, recognising the absence of a clear definition and classification of environmental taxes and its necessity, in connection with the OECD, the European Commission took the first step in its attempt to clarify the concept aiming at "an operational definition of environment-related taxes" (Eurostat, 1996a: 2). It was also intended to develop a statistical framework able to allow data comparison. This work was commissioned by Eurostat. Both institutions, the OECD and the EU, joined efforts to set up a database on 'environmentally-related taxes, fees and charges'[24] and adopted the definition proposed by Eurostat without further investigation into the concept. The definition then adopted allowed the classification as environmental of a tax whose "tax base is a physical unit (or a proxy for it) of something that has a proven specific negative impact on the environment, when used or released" (Eurostat, 1996a).

In 1997, the European Commission expressed some concern regarding the provision of useful guidelines for Member States in designing, implementing and evaluating environmental taxes. Though it referred to the taxable base to define a tax as environmental, effects achieved were also accepted as a relevant criterion to such classification (COM(1997) 9 final, 26 March 1997, 3-4). The Communication of the European Commission laid the burden of proof on national governments: "(…) it is up to the Member State to show the estimated environmental effect of the levy" (ibidem). This approach was maintained in the 2001 Community guidelines on State aid for environmental protection (European Commission, 2001).

Contrasting rationales of different epistemic communities

Dominant and more or less institutionalised features and values in the policy process affect the types and degree of rationality being pursued in different national political systems. Regarding pollution taxes, the main epistemic communities involved in the policy process presented contrasting rationalities, which they managed to imprint in that process in different degrees. Following this contrast, along the tax policy process some stakeholders contributed to blurring the concept of environmental tax whilst others were concerned early on with raising awareness about this risk.

For instance, in Sweden, political parties and decision makers indiscriminately talked about 'environmental taxes' and 'environmentally-related taxes' when referring to pollution taxes. By taking the two notions as interchangeable, these stakeholders stimulated understandings and institutional practices that used the environmental label associated to fiscal instruments. However, environmental economists and academics involved in green tax commissions have tried to clarify the objectives and effects behind each tax instrument. Concern about clarification is evident in some literature which distinguished for the Swedish case between taxes with a clear environmental profile ("'pure' environmental taxes") and taxes environmentally-related with a mainly fiscal nature.

Such dissent is reflected in the debate regarding the energy tax and the electricity tax. In Sweden, until the middle of the 2000s, both taxes tended to be considered environmental by political parties and the government but not by environmental economists. The Green Party, opposing energy consumption *de per se*, took the position that the production of energy in itself had a negative impact on the environment and proposed steep increases of energy taxation.[25] However, economists argued that the energy tax was an inefficient tool to pursue environmental goals and the electricity tax was fiscal rather than environmental, as the source of externalities is not energy consumption but energy production (Brännlund and Kriström, 1999b: 33, 35). Hence, the quantity of energy consumed it is not relevant, but the kind of sources used for its production is. The same line of argument was used to exclude vehicle taxes from the environmental classification (idem, 33).

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[24] Database available at www.oecd.org/env/tax-database.

[25] On this discussion see Swedish Green Party Program, 1998: 14, and in the press, *e.g.* Jan-Olof Tedebrand, *Länsträff om energipolitik*, Västernorrland, 29 January 2003. Runar Brännlund, interview, 13 August 2003, Umeå.

There was a clash of opinions also regarding who should be taxed. While political parties and environmental NGOs focused on making the polluter pay and reducing the distributional impact tax interventions might have on lower-income earners, economists took a macro-economic approach and were concerned with the efficiency of the intervention as a whole (idem, 16). From an economic perspective, an efficient way of raising tax revenues is to tax consumption rather than production, as a tax on the latter is not as neutral as on the first. Therefore, Swedish government green tax commissions suggested shifting energy taxation into final consumption and the exemption of business, though the latter would still be required to pay environmental taxes, like CO_2 tax and sulphur tax (SOU 2003:38 and SOU 1997:11). This feature of the Swedish energy tax package has introduced some degree of inefficiency and environmental ineffectiveness in the system as different (and often contrasting) rationales have been used simultaneously to design the same tax instrument (see Chapter VI).

The broad concept of sustainable development in the Brundtland Report also raises issues regarding environmental taxation as it does not help to narrow its focus making it more useful in policy guidance. A tax tends to increase the price of goods or services. If there is positive (own-)price elasticity of demand[26], as it is often the case, this instrument will work in the more or less long term to pressure the level of consumption down, as expected from an environmental policy instrument according to the Brundtland Report. However, this leads to a definition of 'environmental tax' able to encompass most taxes raised on consumption.

Broadening the approach: The environmental tax reform idea

The assumption that citizens informed about the seriousness of environmental problems will support political action (Dobson, 2000: 112) is naïve (Garner, 2000: 215). To succeed as political ideology ecological modernization required support to face the resistance from economic interests which were expected to lose out. Therefore, industries that profited from environmental regulation had to be encouraged (Pearce, 1993: 12-3). In the political discourse through which ecological modernisation was communicated, benefits from environmental regulation were highlighted as economic in nature and connected with a general improvement in the quality of life (Garner, 2000: 215).

The environmental tax reform (hereafter also ETR) approach was related to this discourse. The idea of tax neutrality and the double dividend argument on which the ETR concept was built[27] were part of the strategy to get broader support for public intervention.[28] On the one hand, taxpayers would not lose, since revenues obtained from taxing pollution and resources were to be used in the reduction of the tax burden on socially desirable goods and services (tax shift). On the other hand, all the citizens would win thanks to welfare gains (double dividend). As a consequence, the notion of environmental taxation gained a broader sense than the one used at the beginning of the debate by economists, who favoured the narrower concept that identifies with regulatory taxes (Rodi, 2002: 1). In the 1990s, the focus of the debate had shifted from the results to the revenues obtained with the tax charge.

By the end of the decade, 'environmental financial reform' (EFR) had succeeded 'ETR' in the political discourse. However, the idea behind both concepts is essentially the same: increase economic efficiency in a broader sense, which includes environmental improvement, by changing the public financial burden on the economy 'from goods to bads'. Both ideas, ETR and EFR, are 'business friendly policy tools to address environmental challenges' (Hontelez in OECD, 2003b: 64). They intend to tackle the issue with revenue neutrality. The difference between the two approaches lies in the broader meaning associated with the term 'environmental financial reform'. Whilst the ETR focuses only on using pollution tax revenues to reduce other taxes (tax shift), the 'EFR' requires also a reform of subsidies, with a reduction of environmentally perverse subsidies (OECD, 2003b: 64).

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[26] The (own-) price elasticity of demand is a measure of the responsiveness of demand to a change in price, defined as the percentage change of demand per percentage change of the price. It reflects current preferences (consumer demand), technology (producer demand of intermediate goods) and availability of substitute goods. OECD, 2001g: 100.

[27] Pearce (1991) introduced the term 'double dividend' as a result of environmental tax reform (Kratena, 2002: 1).

[28] 'Tax neutrality' is here understood as an environmentally-related tax intervention which leaves the total tax burden unchanged due to a complete recycle of the revenues back to the economy. The linkage between the two concepts, i.e. 'tax neutrality' and 'double dividend', involves a revenue recycle through the reduction of the tax burden on labour.

Since the Pigouvian theory gained momentum in economic literature in the 1960s, pollution taxes have been largely guided by financial objectives. In the 1960s and 1970s, they were often viewed as cost-covering taxes and earmarked user charges. In the 1980s and 1990s, they evolved to a combined rationale of behavioural stimulus and fiscal concerns. Since the 1990s, these instruments became directly connected to the idea of 'environmental tax reform' (EEA, 1996: 8) and 'environmental financial reform' (OECD, 2003b), with special importance being assigned to their revenue capacity (Schlegelmilch, 2000: 4-6). Eroding traditional tax bases, increasing public expenditure due to a growing ageing population and welfare state, a general trend towards reducing the cost of labour in order to reduce unemployment and strong social concern expressed by green parties help explain the emphasis on revenues. A particular strategy of 'piggy-backing' of pollution taxes on existing legislation as a procedure for minimising adjustment costs also accounts for their departure from the normative 'Pigouvian tax' (Määttä, 1997: 348-9).

In 1993, a major change occurred in the European Community policy. From being the goal of the developing European environmental tax policy, in the Fifth EAP, environmental protection became only an expected spin-off of such policy, in the White Paper 'Growth, Competitiveness, Employment' (chapter 10: 150). The Community has then explicitly entered the track of financing pollution taxes, in contrast to its preference for regulating ones expressed in its 1975 recommendation (Deketelaere, 1995: 173, 193).[29]

'Fiscal neutrality' and 'tax shift' were also expressly associated with the Commission's carbon/energy tax proposal (Article 1 of the 1997 proposal, Boeshertz and Rosenstock, 2003: 154). Fiscal neutrality was brought in during the discussion of the first proposal of the Commission. By then the opinion of the European Social and Economic Committee included a 'counter-Opinion' highlighting the risk of the instrument proposed being seized by revenue interests, which received at least one quarter of the votes cast (OJ C 108/20-6, 19 April 1993, Appendix, point 3).

Simultaneously, though, on the one hand, the OECD played down the linkage between environmental and employment policies and put the emphasis on the environmental impact of tax instruments, on the other hand, it emphasised the importance of revenues obtained from taxes raised on polluting bases in the adoption of mitigation and compensation measures to offset or reduce the negative effects associated with the use of the tax system to protect the environment, namely competitiveness issues. This second aspect is associated with the term 'tax packaging' (OECD, 1996b: 55).

Political feasibility was one of the factors influencing the adoption of an integrated approach both in the EU and the OECD (OECD, 1998: 15 and 24). Through the adoption of environmental tax reforms, governments aimed at several objectives simultaneously, such as improving competition in markets, improving effectiveness of government intervention, meeting consumer needs more closely, reducing costs, and promoting technological innovation (OECD, 1996c; Majocchi and Missaglia, 2003: 345; 'the most recent example of integrated policy approach towards sustainable development is the so called "environment-employment double dividend policy"' – Carraro and Siniscalco, 1996:preface). National governments have followed multipurpose interventions particularly through energy taxation, with a combined adoption of pure energy and carbon taxes (Majocchi and Missaglia, 2003: 345).

Therefore, the double dividend rationale, which supports the ETR approach, became recurrent as an underpinning for pollution taxes during the 1990s (several studies referred to in EEA, 1996: 39-40). During that period, the energy and transportation sectors were greatly affected by the increased use of pollution taxes in Europe. There was a 50.8% pollution taxes increase between 1990-1997, the growth experienced in energy taxes being 10.1%, and, in 2000, transport taxes represented approximately 20%of environmentally-related tax revenues in the EU (EEA, 2000b: 2, 7). These figures were a consequence of the implementation of the ETR approach and the high tax revenue potential of the sectors involved.

This approach was supported and further induced by the environmental NGOs. Initially there was some resistance from these towards the use of economic instruments to promote environmental quality. This was mainly grounded on the uncertainty over the pollution control level when economic instruments were used and on some ethical concerns (Golub, 1998: 7; Majocchi, 1998: 147). In the 1990s, this approach

[29] COM(1992) 226, 30 June 1992, COM(1995) 172, 10 May 1995, COM(1997) 30, 12 March 1997. Other documents where the tax shift idea is directly associated to pollution taxes are, for instance, EU Commission, 1997, and EEA, 2002a: 128, at EU level; OECD, 1996b: 55, 1998: 12 in the OECD; and SOU, 1997:11:488, at national level.

was replaced by a pragmatic one characterised by the advance of tax proposals with especial enthusiasm for ETR (Weizsäcker *et al*, 1998: 205). This acceptance has eased the way for a more frequent use of tax instruments within environmental policy. However, the concept of environmental tax used by NGOs has often been foggy. And though its representatives have at times referred to the purpose of environmental taxes as being to act as an incentive to save natural resources (OECD, 2003b: 73), in an attempt to gain support from the ministry of finances to environmental taxation, NGOs have focused on taxes raised on polluting bases with high revenue potential. For instance, the EEB campaign on Environmental Financial Reform demanded an additional 10% shift in total tax revenue from labour to environmental use by 2010, at EU and national level.[30]

Institutional actors, in contrast with non-institutional actors, tend to have an opportunity structure that helps them to score better in the emergence and strategy level, with institutional rules being included here (Hooghe, 1993: 149). National institutional actors have had a major impact on the way the 'environmental taxation' idea spread. The ministry of finance acted as the gatekeeper. Thus, it was not until the latter became interested in environmental taxation that it became a credible option. For example, in Denmark there had already been a report on the possibility of using pollution taxes in 1975, the same happened in Sweden in 1978 (SOU, 1978:43). In both cases the government was reluctant in adopting these taxes as it did not view them as adequate instruments of public policy, *inter alia*, due to the difficulty in putting them into practice (SEPA, 1995: 5-6). The co-operation of the ministry of the environment and the ministry of finances was able to bring the issue to parliament, the latter often being the one to take the initiative (EEA, 1996: 39).

Ministries of finance developed a new interest in environmental taxation due to the fact that recently graduated environmental economists started entering the staff of the ministry and pinpointing the potential of these instruments. Financial policy-makers acknowledged the potential gain from spreading the label 'environmental' to tax instruments in order to reduce the political costs of raising revenues (Yandle, 1998: 127-128) and consequently these instruments gained a leading role in proposals for neutral tax reforms (e.g. SOU, 1997:11:475). Simultaneously, the awareness of the ministry of the environment of its own interests may have also increased. The possibility of getting earmarked revenue may have appealed to environmental bureaucrats (Svendsen *et al*, 2001: 496).

Simultaneously with the increased fiscal bias of pollution taxes, it has been recorded a loss of trust in environmental taxation in countries adopting 'environmental tax reform' approaches, of which Sweden and Denmark are historical examples (Petras Project, 2002: 14, Eurostat, 2005: 49-50). Acceptance has not been an obstacle to the initial adoption of taxes to pursue environmental goals in the early 1990s. 75% of the respondents to the 1995 Eurobarometer Poll expressed such support (EEA, 1996: 42), which reduced to 44% when it was assumed a slight negative economic impact (Gee, 1997: 83).

But the reality had changed considerably by the end of the 1990s, when only 5% of the European citizens were willing to contribute to the environment by paying taxes, being these the less-liked instrument of environmental policy (Eurostat, 2005: 49-50). By then lack of public support was an important deterrent to further developments in environmental tax reforms due to heavy suspicion and resistance associated to pollution taxes (PETRAS, 2002: 14). Levels of awareness and understanding of ETR varied among the countries. However, widely expressed beliefs were that ETR was just a way to raise more money; though the name was chosen to make people think that it was for the environment. People could not understand the point behind ETR as being about changing behaviour (PETRAS, 2002: 14; 'fiscal cow' – Määttä, 1997: 11; Deketelaere, 1996: 9; Gaines, 1991: 479).

CONCLUSIONS

Environmental taxation harnesses economic theory to protect the environment, using economics as a means to the end (Milne, 2003: 3). Although, economic theory, with its several contributions, is associated with the origins of environmental taxes and has served as their 'conceptual linchpin' (ibidem), the importance of economics in shaping the contours of these specific tax instruments has decreased over

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[30] Available at http://www.ecotax.info/platform.pdf; http://www.foes-ev.de/GBNnews1/2artikel4.html.

time as theory has been adapted to fit the reality. Since the idea of taxing pollution was first brought up by Pigou, the relationship between economic theory and taxes as environmental policy instruments regarding the implementation of theory in institutional practice has developed along two lines, namely the economic oriented school of thought of environmental economics and the more political and philosophical school of thought of ecological modernisation.

The contribution from economics to explain the adoption of environmental taxes has changed over time. And economic arguments have been progressively replaced or shaped by political ones in the design of pollution taxes. From an argument mainly based on the internalisation of external costs in the 1970s, the focus of the economic approach has moved to the double dividend argument since the early 1990s.

The incompleteness of the theory informing the public decision-making process in the face of empirical limitations and political obstacles might explain this evolution (misapplication of 'good theory'). Reference stakeholders with different profiles in terms of rationalities, motivations and information worked selectively and strategically on the theory. They translated the idea of green taxation into political proposals and used win-win discourses to communicate it. This process created a new and broad institutional practice focused on remediation and revenues rather than pollution prevention referred to as 'environmentally-related taxes' ('theoretically impure environmental taxes' – Milne, 2003: 4).

Ecological modernisation, which together with the PPP was the main component of the ideational dimension in politics providing ideas and paradigms, opened the way, supported and framed the adoption of pollution taxes. Its focus on means rather than ends, ignoring issues of socio-economic justice and the need to change consumption patterns, led to highlighting the 'dynamic efficiency' potential of pollution taxes and forgetting their preventative potential.

A further contribution from environmental economics to the fiscal bias of pollution taxes was the double dividend argument, which since the 1990s has often been used to underpin pollution tax proposals wrapped under the environmental tax reform idea. This argument based on efficiency concerns applied to the whole tax system and was strategically used by certain coalitions to advance reductions of the tax burden on labour as a means to promote higher employment rates via a tax shift idea. The fiscal bias of pollution taxes was accompanied by their loss of credibility.

Following this lack of principal convergence, agreement over a single set of design guidelines for environmental taxes has been missing and the environmental tax discourse has evolved towards non-regulatory concerns. The next chapters test the theoretical proposal based on the understanding of environmental taxes as regulatory instruments of environmental policy with precise design features aimed at pollution prevention, which was explained in Chapter Introduction, against empirical evidence from Denmark, Portugal and Sweden (Chapters II, III and IV, respectively).

CHAPTER II

THE DANISH WASTE TAX

Danish legal vocabulary and abbreviations

Betænkning (Bek.)	Report made by an official committee or commission
Folketingstidende	The Official Journal of the Danish parliament
Bekendtgørelse af lov (LBK)	Consolidated version of an Act and its amendments
Lovforslag	Bill or Law proposal (proposal of new legislation)

This chapter assessed the inclusion of the design features of environmental taxes in the Danish waste tax. The analysis focused on the period 1987-2000 since changes made to the categorisation of sectors in the Environmental Protection Agency information system (ISAG) made any direct comparison with pre-2001 data more difficult. It aimed at identifying the degree of such inclusion and the causal linkage it kept with the environmental effectiveness of the tax. It has been concluded that, though at its introduction the tax was aimed at reducing traditional disposal (i.e. landfill and incineration) by raising the incentive for recycling, it delivered strong and fast behavioural effects only in waste streams characterised by heavy weight and low recycling costs, namely construction and demolition waste, garden waste and bulky waste. This limitation seems to have been mainly rooted in the low level of the tax rates and the filtration by the institutional framework of the price signal provided by the tax, which strongly harmed tax awareness, the reward of tax avoidance strategies corresponding to pollution prevention and the linkage between tax payment and environmental damages.

There was not a strong inclusion of the design features of environmental taxes in this tax, which contrasted with what was expected since its introduction in 1987 was not part of the Danish environmental tax reform process initiated in the 1990s and its design and management until 1993 was carried out by the Ministry of Environment. The entity designing the tax held technical expertise, but its lack of interest in the tax base fell short of being proofed, since tax revenues were allocated to it. Furthermore, the tax management was *de facto* controlled by the municipalities, who displayed ambiguous interests with regard to waste reduction following their simultaneous role as indirect taxpayers, regulators and professional players in the waste service market. These aspects might explain, respectively, the diversion of the tax design, especially regarding the setup of the tax rate, and institutional practice away from normative environmental taxes.

The Danish waste tax had from its introduction precise, though unquantified until 1990, environmental objectives, namely to divert waste streams towards more sustainable waste management techniques according to 'the Danish waste hierarchy'. This did not strictly follow from environmental criteria, since reduction was not prioritised and energy recovery was made equivalent to incineration, though the latter is only one of the possible means of achieving it. Furthermore, no consideration was provided directly in the tax design to the kind of material flow entering the ecosystem, being only the amount of such flow addressed by the setup of a weight-based waste tax.

Tax awareness and tax avoidance were instituted in the tax design mainly via the tax charge according to weight of waste and the reward of recycling strategies. The latter was performed both by the tax refund assigned to recycling companies for the waste removed from traditional disposal plants and by the use of the tax revenues to finance the subsidy schemes for recycling projects until 1993. However, these features have been almost neutralised by institutional filters in their transference to institutional practice in the waste management system. The pricing system used in a vertically integrated market delayed, hid and flattened the price signal provided by the tax. Furthermore, it cut the linkage between tax payments and the amount of waste disposal (and consequently also of environmental damages), instituting tax illusion and reducing the availability of tax avoidance strategies. In industry and commerce, this reduced tax awareness is likely to having added to the negative effect on waste disposal cost awareness already caused by low tax levels.

The tax was raised on measured units (weight/tonne) of waste delivered to registered plants (landfills and incinerators), which was a good proxy for polluting emissions. At the introduction of the tax, its rate was referred to pollution abatement costs, namely the costs of waste recycling. This aimed at making recycling profitable vis-à-vis traditional waste disposal. After 1992, tax rate differentiation was used to favour some waste management techniques over others, exploring further the potentialities of the tax to lead towards the 'Danish waste hierarchy'. Recycling was preferred over incineration and this was advantaged over landfilling. Furthermore, more energy efficient incinerators were advantaged over less efficient ones, being energy efficiency measured according to heat and power generation capacity.

The tax performed worse in leading waste streams towards the waste hierarchy when it failed to communicate environmentally correct *effective pollution prices* and consequently to induce behavioural change. This was the case when the additional costs of recycling were not compensated by the level of the tax rates applied to traditional waste disposal, failing the tax to make recycling profitable vis-à-vis traditional disposal. And it was also the case when the lower municipal fees charged in landfills were not compensated by the higher tax rates on waste brought to those plants failing the tax to guarantee lower *effective disposal costs* for incineration. The reference of the tax rates to waste-intensive companies, though most opportunities for improvement were in waste-extensive ones, also made the tax unable to induce behavioural change in industry except regarding heavy waste.

The tax was raised on the waste treatment plants, which were the polluters with the best capacity to avoid pollution by influencing the design of waste management through the development of recycling facilities, on which the environmental effectiveness of the tax highly depended. Though the polluters ultimately taking the tax burden were the waste producers, the price signal was primarily addressed to waste treatment plants. The power they had to influence the waste management system was connected to the fact that, at the introduction of the tax, most of them were owned by the municipalities, which were the entities controlling such a system. However, this proximity has also harmed the environmental effectiveness of the tax, since the desire to exploit available installed capacity in traditional disposal has sometimes been reported as the reason for not adopting institutional designs that would have impacted positively on waste reduction and recycling, such as weight-based collection fees.

1. BRIEF OVERVIEW ON WASTE INTENSITIES: NOTION, CAUSES AND REGULATION

There is a trade-off between the quantity of goods produced and waste generated, since materials entering the production process (raw materials and intermediates) end up as produced goods or residuals (waste or pollutants). This trade-off is reflected in the waste intensity. The waste intensities are defined as the ratio between the waste amounts and the production (measured for example in value added terms or absolute quantities, for example gross domestic product) in the respective sectors and industries of the economy (Brix and Bentzen, 2008: 1, 3).

Waste intensity is not a constant term, being influenced by relative changes in the factor input prices and technological change (ibidem). Following the latter the number of units produced can increase while the volume of waste generated remains unchanged reflecting a more efficient use of material input and consequently reducing the ratio between waste and production (i.e. waste intensity). There is not a constant relationship between a firm's productivity and waste generation (Brix and Bentzen, 2008: 3). For example, the introduction of minimum requirements to the production technology regarding waste generation helps to stem the growth in the volumes of waste produced by the industrial sectors (Brix and Bentzen, 2008: 17), whereas a waste tax can induce efficiency in the use of material input.

Waste increases can be due to increasing waste intensities (the companies produce more and consequently generate more waste) or to less efficient production (the companies' gross value added production has decreased while the firms have maintained the same level of input in the production) (Brix and Bentzen, 2008: 14). They can also be due to a reduction in the incentive to produce less waste, which can be rooted for example in reductions in real waste generation prices, following the failure to update over time the amount at which the waste tax is fixed (Brix and Bentzen, 2008: 15). A fourth cause of waste increases might be the incorrect waste management system in place. All these causes were identified in the Danish waste system (ibidem).

How waste taxes work

The waste tax can help reduce total waste amounts when these are due to increasing waste intensities by inducing a more efficient use of the material input. But it is less effective in addressing increasing waste volumes when these are caused by activity or structural effects (Brix and Bentzen, 2008: 4). In such cases, regarding waste amounts, a waste tax can perform mainly cost internalisation by raising revenues rather than behavioural steering towards the adoption of waste reduction strategies. However, regarding waste management, the tax can still have a positive environmental impact. It can steer behaviours by diverting waste streams towards more sustainable waste management techniques. A waste tax is expected to lead to waste reductions mainly where the tax rates are higher than the marginal treatment costs of the waste management alternative to disposal (for example, recycling).

Following the various causes of waste increases, a waste tax is insufficient to address the waste production problem. Firstly, it cannot address activity and structural effects. Secondly, to affect the choice of materials and consequently reduce waste intensities a waste tax needs to be very high (politically unfeasibility high) since materials are only a small part of a product's cost. By the end of the 2000s, the price elasticity of waste production in Denmark was measured at 0.5, in the sense that the increase of waste taxes by 1% would lead companies to reduce their waste production by 0.5% (Brix, 2010: 57). And thirdly, if there is an economic incitement to waste production in the waste management system, the tax will lose its capacity to provide an incentive to waste producers to take investment in waste reduction (Brix and Bentzen, 2008: 17).

2. Brief overview of the Danish waste management policy

In the mid-1980s, Denmark responded to the waste disposal problem by adopting a comprehensive waste management policy. This evolved from focusing on waste reduction to pursuing the diversion of waste streams towards more sustainable waste management techniques to which the cost efficiency of waste management later joined. Danish waste legislation was characterised by a close interplay between EU regulation and national regulations (DEPA, 1999b). The Danish waste model was based on a combination of traditional administrative instruments (Acts, Statutory Orders and Circulars) and various other instruments, such as taxes and charges, subsidy schemes and agreements (DEPA, 1999b).

Until 1997 efforts in Danish waste management policy primarily concentrated on limiting waste arisings, increasing recycling and reducing landfilling (DEPA, 1999b). In the 1998-2004 Waste Management Plan special attention was paid to the quality of waste treatment and some waste fractions (aiming to separate them at source) (DEPA, 1999b). These were addressed through specific taxes such as the packaging tax. The 2005-08 Danish Government's Waste Strategy built upon three fundamental elements, namely prevention of resources loss and environmental impact from waste, decoupling of the growth in waste from economic growth and improved cost-effectiveness of environmental policies (DEPA, 2004).

After the merger between the Ministry of Energy and the Ministry of Environment into the Ministry of Environment and Energy in 1994, the Danish Environmental Protection Agency (hereafter also DEPA) became responsible for waste management, co-operating with industrial associations about waste return possibilities (DEPA, 1999b).

The Danish waste management system

Each municipality was responsible for managing, according to the waste hierarchy, the waste (from households, trade and industry) produced within its geographic boundaries, except for specific waste fractions covered by special regulations (Andersen et al, 1997: 31). Their responsibility for waste management included collection, transportation, recycling and disposal of waste, including hazardous waste (ibidem). Municipalities could choose to coordinate operational tasks themselves, assign them to an inter-municipal company (i.e. a union of municipalities) or delegate them to private companies (OECD, 2004: 33). Often local authorities got together for the provision of waste removal and treatment services, generating economies of scale (ibidem).

Municipalities were legally obliged to collect and dispose of household waste in areas of more than 1000 residences (Andersen et al, 1997: 31). In less populated areas where collection was not mandatory and for commercial institutions the municipalities were assigned the responsibility of issuing instructions as to where the waste could be disposed of (Sections 43-50a, Environmental Protection Act Consolidated Act N. 753, 25 August 2001; Statutory Order on Waste N. 619, 27 June 2000). Collection schemes for household waste were operated by both municipalities and private waste companies, whereas the management of industrial and commercial waste was usually carried out by private companies alone (Andersen et al, 1997: 31-32).

Municipal user fees were collected from all those connected to the municipal collection scheme. Waste management expenses of the municipalities regarding household waste, including costs of collection and disposal, could be covered using fees or taxes (Andersen et al, 1997: 88-89, 32), whereas costs associated with the management of waste from services and industry had to be covered with fees (ibidem).

In pricing waste collection municipal authorities were restrained by cost recovery limits (ibidem). The principle guiding the fixing of waste management fees was that for each user group fees should to the extent possible reflect the actual costs of the scheme for which they were charged (ibidem). In some municipalities this principle has been taken strictly in that all waste producers, including households, paid for the collection of waste by weight (Andersen et al, 1997: 92-100[31]). However, in the absence of legal constraints concerning the pricing system used, municipalities frequently fixed fees based on certain waste volume capacity (ibidem).

At waste plants, gate fees were collected from the deliverer of waste based on waste there delivered, weighed and categorised. Waste plants had to comply with the principle of full cost recovery and cross-subsidisation was in principle not allowed, but they were free to establish their own fee structure in terms of DKK/ton of waste (Andersen et al, 1997: 32; OECD, 2004: 101). This could reflect the capacity and options for waste treatment available at the specific plant, as well as the objectives and priorities of the specific municipal waste plants (ibidem). Typically, plants would apply a higher fee for unsorted waste than for sorted waste and different levels for different types of waste (ibidem).

Only a state/municipal company could establish solid waste landfills, though there were private landfills for inert waste (Andersen et al, 1997: 6). Therefore, most such plants were municipal, but there were also many small private plants. In addition, some companies had their own landfills or incineration plants. Both in the case of municipal and intermunicipal waste disposal, companies running plants receiving the waste normally used private waste carriers and collectors (ibidem). Companies which provided waste management services could also take part in the waste recycling process, establish waste recycling facilities and purchase waste to be recycled (ibidem).

Danish municipalities administered the flows of waste suited for incineration through waste disposal schemes. The majority of the Danish municipalities were either owners or co-owners of an incineration plant and basically the municipalities either collected the waste for incineration or referred it to their own plants for incineration (Odgaard, 2011: 4). Therefore, they enjoyed a quasi-monopoly status in waste incineration. Consequently, the choices open for waste producers in terms of waste treatment were limited by the municipal schemes (ibidem). For example, from 1997 onwards, in industrial waste suitable for incineration a company would either use the municipal schemes or transfer the waste to incineration plants (co-incineration or dedicated incineration plants that were also recovery plants) in another country (ibidem).

This allocation of responsibilities led to an economic incitement in the Danish waste management system (Brix and Bentzen, 2008: 15-16). For a firm to reduce its waste amounts it would need to make changes in the production processes and undergo the necessary investment. This decision would be economically rational if the potential economic gains from less waste more than compensated for the investment costs. However, this was not usually the case due to the insufficient level of the fees charged by waste treatment

[31] The report analysed developments in the decade from 1987 to 1996 in 10 Danish municipalities.

facilities plus the waste tax (ibidem). These facilities had no incentive to encourage the firms to reduce their amounts of waste since this would reduce their own production. Danish municipalities simultaneously played the role of regulator and owner of many treatment facilities. Therefore, they had ambiguous incentives (ibidem).

3. THE DANISH WASTE TAX

The Danish waste tax was introduced in January 1987 by the Bill to reform the Danish Environmental Protection Act (Law Proposal 176, 11 February 1986; Law 329, 4 June1986). Under the Danish constitution no tax could be collected but by law. However, this tax was not introduced through a genuine tax law but as part of a regulatory law. Therefore, until 1988, Danish Statistics did not classify it as such but as a fee[32], which involves an idea of bilateralism in the sense of a payment for a public service. With the introduction of a raw materials tax, the waste tax was transferred from the Danish Environmental Protection Act to a stand-alone Act on taxes on waste and raw materials (Law 100, 8 December 1989) (Andersen et al, 1997: 22). In July 2011, the tax on landfilling of waste was regulated by Law 529, 17 June 2008 as amended by Law 311, 1 April 2011 (Odgaard, Law Department (University of Aarhus), e-mail communication, 2 September 2011)

The background leading to the adoption of the waste tax

Conceptually, the background of the interest in economic instruments in Denmark had its roots in an OECD conference held in 1984 (OECD, 1984) and a subsequent study carried out by the Danish Environmental Protection Agency (Andersen et al, 1997: 22). A tax on waste was discussed in detail in a 1985 report from the DEPA on possible use of economic instruments in environmental policy (ibidem). At the national budget negotiations in 1985, political agreement was obtained on the introduction of a waste tax. The tax was adopted with broad political agreement in 1986 (ibidem).

The context leading to the adoption of the waste tax was marked by the need to find alternatives to incineration to deal with increasing waste amounts. In the mid 1980s, Denmark's per capita generation of waste was higher than many of its neighbours in Europe and projections showed that its landfill space was quickly running out (Andersen et al, 1997: 22). There were considerable problems in siting new landfills (ibidem).

However, at the same time, the dioxin debate in Denmark had focused on waste incineration plants as the source of diffuse dioxin pollution and consequently there was growing concern over air pollution from incinerators (ibidem). Limited options for waste management and growing per capita waste output made this a major concern, leading the country to adopt a strict waste management program (ibidem) and to recognise the need for new and more preventive ways in waste regulation (ibidem).

In Denmark, the allocation of tax revenues to individual ministries was not the rule. They were allocated means in the budget by a parliamentary law (Jens Hansen, Danish Ministry of Taxation, e-mail communication, 6 November 2002). Under budget rules a ministry could avoid budgetary cuts through suggestions of budget-improving measures (revenue-enhancing or expenditure-reducing) under other departments (ibidem). If the ministry used more funds than assigned in the budget it was obliged to come up with suggestions the following year on how to correct the imbalance (ibidem).

For the budgetary year 1987 the Ministry of Environment did not have sufficient means to finance initiatives it considered necessary (René Mikkelsen, Danish Ministry of Taxation, e-mail communication, 23 October 2002). Therefore, it was allocated revenue from several excises, as well as allowed to increase and introduce others (ibidem). Such excises were adopted by parliamentary law and the allocation to the Ministry was part of the budget as adopted by Parliament (ibidem).Those were

32 Apparently the tax was confused with a user fee. If a charge is introduced in a regulation law, the Danish Statistics will rule it as 'user fee' unless following some doubts the matter is further investigated. Jens Hansen, Danish Ministry of Taxation, e-mail communication, 12 November 2002.

environmentally related duties, namely an excise on packaging and another on waste (ibidem).The latter became the waste tax.

The political history of the waste tax after its adoption

Until 1993, all waste tax revenue was book kept under the Ministry of Environment and claimed to directly enhance the incentive provided by the tax charge (Jens Hansen, Danish Ministry of Taxation, e-mail communication, 12 November 2002). However, legally the revenues could be used for any purpose since from the budget law only a maximum expenditure limit followed (ibidem). In 1989, the Ministry of Environment tried to solve its budget problems by raising the waste tax and formally delivering it to the Ministry of Taxation but keeping the revenue. The Ministry of Finance accepted, but the Ministry of Taxation refused. Therefore, the Ministry of Environment had to propose a genuine tax law to bring the tax in agreement with the Danish constitution (LBK 882, 17 November 1993, Consolidated version of the Act on taxes on waste and raw materials and its amendments). Some changes were then introduced in the tax design regarding the tax base (ibidem).

In the beginning of 1993 a new centre left government took office and decided that a number of taxes should be transferred from the Ministry of Environment to the Ministry of Taxation, including the waste tax (ibidem). It was argued that the Ministry of Environment, which disliked having to refuse the requests of local authorities for tax credits, willingly gave the waste tax up, being compensated for the revenue loss (ibidem). The tax, which had for long been administered by local tax authorities, was formally transferred to the Ministry of Taxation (ibidem). Its revenues, which were channelled back to the general public budget and managed like those of any other fiscal tax, have been used as part of the green tax reform since 1993 (EEA 1996: 54).

3.1 The waste tax was mainly aimed at lowering traditional waste disposal

The 1987 Danish waste tax was introduced with the aim of reducing traditional waste disposal, i.e. the waste amounts going to incineration and landfill. This was to be attained both via (1) reduced waste generation, following the incitement to companies to apply low waste technologies and consequently reduce their waste intensity, and (2) increased recycling and reuse by households, manufacturing industry and the building sector (Law Proposal 176, 11 February 1986, 4425-4426).

Recycling was expected to increase following the financial interest of the municipalities, through their waste collection services, in establishing adequate treatment facilities (recycling and separation systems). A tax on waste final disposal was expected to increase the profitability of recycling because for each tonne of waste delivered for recycling such services would save the corresponding waste tax payment following the refunds assigned for waste subsequently removed from registered plants (Law Proposal 176, 11 February 1986, 4425-4426).

3.2 The waste tax was raised on measured units of waste

The waste tax was an excise duty levied on the final disposal (dumping, landfilling and incineration) of non-hazardous waste to registered waste treatment plants (incineration plants and landfills) (Andersen et al, 1997: 33). The tax was raised on measured units of waste, namely waste weight measured by tonnes of waste disposed (Andersen et al, 1997: 5). It has been extended several times to also cover waste going to other plants rather than only municipal plants.

Waste that was given another approved destination rather than registered waste treatment plants, such as hazardous waste, was exempted. The refund for waste subsequently removed from registered plants provided an incentive to recycling and avoided double taxation of waste that after treatment in an incineration plant was disposed of at another registered plant (Andersen et al, 1997: 23).

The waste tax base has been expanded several times. Until the end of 1989, the tax was limited to waste going to plants receiving waste from municipal collection schemes (ibidem). Waste not collected in municipal schemes was also taxable if it was delivered to such plants, which in practice were large, primarily municipally operated landfills and incineration plants (ibidem). Therefore, industrial and commercial waste delivered directly to municipal plants was also covered by the tax (ibidem).

To simplify the tax administration waste going to private landfills for inert waste or other private landfills was not included until 1990 (ibidem). The 1990 Statutory Order replaced the provisions of environmental regulation. It partly imposed an assignment obligation on local councils and partly extended the concept of municipal waste collection schemes (Andersen et al, 1997: 24). The tax base was then extended so that private landfills for inert waste were also covered (Law 838, 19 November 1989, Act on taxes on waste and raw materials). Therefore, with effect from 1990 onwards the tax base was extended to comprise all plants receiving collected or assigned waste. This meant that a large number of small, primarily private, inert waste landfills and companies with their own landfills were required to register (ibidem).

Several waste types were not covered by the waste tax due to their insignificance, the economic or environmental relevance of the activities involved or the use of other regulatory instruments, such as special collection requirements, deposit and return systems, agreements on recycling and taxes aimed at regulating the consumption of products (Andersen et al, 1997: 28-29). Most regulations were related to special and, by weight, mostly less important waste streams.

The tax bases of other taxes have overlapped with the waste tax base. The weight-based tax on packaging introduced in 1999 joined three others which provided an incentive to increase re-use and hence reduce the volume of waste (Dengsøe and Andersen, 1999: 5), namely a volume-based tax on new packaging for most beverages (since 1978), a tax on disposable tableware (since 1988) and a weight-based tax on paper and plastic carrier bags (since 1994) (DEPA, Waste Strategy 2005-08, 2004, Appendix B). In 1989, a tax on raw materials was also introduced, aimed at supplementing the incentive provided by the waste tax to recycle construction and demolition waste (Act on taxes on waste and raw materials, Law 838, 19 December 1989) (Andersen et al, 1997: 24).

3.3 Waste tax rates mainly referred to recycling costs

The waste tax rates have been continuously increased since 1987 with reference to the need to make recycling profitable and during the 1990s it more than doubled the cost of waste disposal. In 2001, the Danish waste tax was the highest in Europe (Andersen and Dengsøe, 2002: 28). Since 1992, have been differentiated according to the kind of waste treatment provided by the taxpayer, namely landfilling and incineration (Law 1071, 23 December 1992).

In 1987, the tax rate adopted reflected the need to ensure the profitability of recycling plants for construction and demolition waste and collection schemes for glass (Andersen et al, 1997: 22). The tax was uniformly charged at DKK 40/tonne on all taxable waste delivered to municipal waste facilities (ibidem). Over time several refinements have been introduced, especially concerning the cost difference between landfilling and incineration.

In the Finance Act of 1990, the waste tax rate was increased to DKK 130/tonne, which was estimated to produce an additional yield of DKK 340 million (Andersen et al, 1997: 24). This was taken in connection with a comprehensive plan of action to increase recycling to 54% until 1996 (Plan of Action for Waste and Recycling), of which the tax was an integral component (Folketingstidende, 89/90, 3056; Andersen et al, 1997: 14).

In connection with the 1990 tax increase a refund scheme for recycling companies was introduced. According to the scheme, these companies were granted a refund on their waste tax of DKK 90/tonne (Andersen et al, 1997: 26). The rate of the refund scheme was increased with effect from 1993 to DKK 120/tonne and DKK 155/tonne for incineration and landfilling, respectively (Bemærkning to Law Proposal 70, 28 October 1992, 1826).

In 1992, following a proposal from the Ministry of Environment, the tax was restructured to decrease the cost of waste brought to incinerators under the cost of that brought to landfills (Law 1071, 23 December 1992) (Andersen et al, 1997: 22). The purpose of the increase was twofold, namely to compensate for the abolition of the subsidy scheme for recycling projects from 1993 onwards and to increase the incentive to incineration compared to landfill. The tax differential was set at DKK 35/tonne (Andersen et al, 1997: 27).

In 1993, the tax rate was further differentiated according to efficiency in incinerators measured by energy (electricity and heat) recovery to promote combined heat and power production (Andersen et al, 1997: 26-27).[33] This change was part of the tax reform that shifted part of the tax burden from personal taxes to pollution taxes and involved also an increase of the waste tax rate with effect only from 1 January 1997 (ibidem). The tax was then restructured to lower the tax rate applied to waste brought to incinerators with electrical and heat recovery capacity, which by then accounted for most of the Danish incineration capacity, under the tax rate applied to waste brought to incinerators just with heat recovery capacity (ibidem). In the same year the Government's Plan of Action for Waste and Recycling for 1993-1997 introduced the obligation to incinerate waste suitable for incineration (Andersen et al, 1997: 25-26).

In December 1996, the waste tax was further increased by DKK 50/tonne for all three categories, namely waste delivered to landfills, waste delivered to incinerators with electrical and heat recovery and waste delivered to incinerators just with heat recovery (Law 493, 30 June 1993, as amended by Law 1224, 27 December 1996) (Marie Odgaard, Law Department (University of Aarhus), e-mail communication, 2 September 2011). From 1 January 1997 onwards, following the law changes approved in 1996, the tax rate amounted to DKK 335/tonne on waste going to landfill, whereas the tax rate applied to waste brought to incinerators just with heat recovery capacity was DKK 260/tonne (Andersen et al, 1997: 27). This meant a differentiation of DKK 75/tonne and a 72% increase for waste brought to landfills compared to 1992 levels. The rate applied to waste brought to incinerators with heating generation capacity and a minimum of 10% power generation was DKK 210/tonne (ibidem). This meant a tax rate differentiation of 125/tonne with reference to waste brought to landfills compared to the previous DKK 35/tonne.

In January 1999, a tax on heat produced from district heating systems via waste incineration was introduced (Law 1034, 23 December 1998), eliminating the tax advantage previously involved in applying waste for the production of heat compared with conventional fuels as waste incineration plants were exempted from the CO_2 tax (but charged with the SO_2 tax) (Nordic Council, 2006: 84). The tax introduced was often passed on by incineration companies to waste producers (ibidem). In 2001, the waste tax differentiation between incinerators with and without power generation ceased as all municipal waste incinerators in Denmark (still exempted from the CO_2 tax) were required to provide both heat and power (Law 1295, 20 December 2000) (ibidem).

In January 2010, the waste tax structure was changed (Law 461, 12 June 2009; Law 527, 12 June 2009). The weight-based tax levied on waste brought for incineration was abolished and replaced by a general energy and CO_2-tax in line with the tax structure for other fuels. This was charged on waste used as fuel for heat production in power plants (or attributable to heat production in combined heat and power plants) at a rate of DKK 31.8/Gigajoules (Decree Law 1292, 17 November 2010) (Odgaard, Law Department (University of Aarhus), e-mail communication, 2 September 2011). The purpose of the alteration was to change the waste flows so that to a greater extent the waste would be incinerated where it was cheapest, as well as to create incentives for a different distribution of the energy resources of waste between heat and electricity (Odgaard, 2011: 1).

3.4 The waste tax was raised on waste treatment plants

Registered waste treatment plants were the ones liable for the tax payment (Andersen et al, 1997: 31). They would next pass the tax burden to waste producers by charging those delivering the waste at the plant, namely (public or private) professional carriers or the waste producers themselves who being under assignment schemes decided not to hire professional carriers (ibidem). Regarding waste covered by municipal collection schemes, owners of real estate (and the companies regarding recyclables covered by the scheme) were legally imposed to be liable for the payment of waste collection fees (where the waste tax was integrated) (ibidem). They paid to the municipality, which in return settled its accounts with waste companies and treatment plants (Andersen et al, 1997: 88-89). Therefore, the economic burden imposed by the tax was ultimately laid on the waste producers.

[33] Law 493, 30 June 1993, approved for the first time such differentiation following the Law Proposal 304, 19 May 1993, but its rates never became effective because meanwhile it was amended by Law 1224, 27 December 1996. Marie Odgaard, Law Department (University of Aarhus), e-mail communication, 2 September 2011.

Initially certain waste treatment plants were not covered by the obligation to register. Therefore waste going to these plants was not taxable. Under the argument that it was necessary to increase recycling of construction and demolition waste which had previously been landfilled at small inert waste landfills and private landfills, in 1990 such sites also became liable for the tax payment (Andersen et al, 1997: 6, 24).

Table 2.1 Evolution of the Danish waste tax

01.01.1987 **Law Proposal 176, 11 February 1986**	The same rate on all taxable waste delivered to municipal waste facilities	DKK 40/tonne		
01.01.1990 **Law 838, 19 December 1989**	The tax base was extended to cover waste delivered to private landfills & other private facilities			
01.01.1993 **Law 1071, 23 December 1992**	The tax was restructured to decrease the cost of waste brought to incinerators under the cost of that brought to landfills	Waste brought to landfills	Waste brought to incinerators	
		DKK 195/tonne	DKK 160/tonne	
01.01.1997 **LBK 882, 17 November 1993**	The tax was restructured to lower the rate of waste delivered to incinerators with electrical & heat recovery capacity under the cost of waste brought to incinerators with just heat recovery	Waste brought to landfills	Waste brought to incinerators with just heat recovery	Waste delivered to incinerators with electrical & heat recovery capacity (with a minimum of 10% power generation)
		DKK 335/tonne	DKK 260/tonne	DKK 210/tonne
01.01.1999 **Law 1034, 23 December 1998**		Waste brought to landfills	Waste brought to incinerators with just heat recovery	Waste delivered to incinerators with electrical & heat recovery capacity (with a minimum of 10% power generation)
		DKK 375/tonne	DKK 330/tonne	DKK 280/tonne
01.01.2001 **Law 1295, 20 December 2000**	The same rate was charged on all waste incinerated	Waste brought to landfills	Waste brought to incinerators (all with electricity and heat recovery capacity	
		DKK 375/tonne	DKK 330/tonne	
01.01.2010 **Law 527, 12 June 2009**	Introduced a new tax	Waste brought to landfills	Waste brought to incinerators (all with electricity and heat recovery capacity	
		DKK 475/tonne	New tax	

Source: Marie Odgaard, Law Department (University of Aarhus), e-mail communication, 2 September 2011.

4. Evaluation and critical analysis of the waste tax

4.1 Evolution of the waste streams

Whilst analysing the evolution on Danish taxable waste amounts it is necessary to caution that data from 1987 and 1989 cannot be directly compared with data from 1990 to 1996 due to successive extensions of the tax base. Moreover, it is necessary to distinguish between gross and net delivered waste amounts. The latter are a better indicator of the developments in taxable waste amounts as this figure is corrected for several counts of the same waste and also represents the final impact of waste after the effect of waste separation and external sale of residues from waste treatment plants for recycling. Gross delivered amounts indicate total taxable waste amounts. Decreases in gross amounts may signal less consumption (or less activity in the construction sector) or increased efforts in recycling.

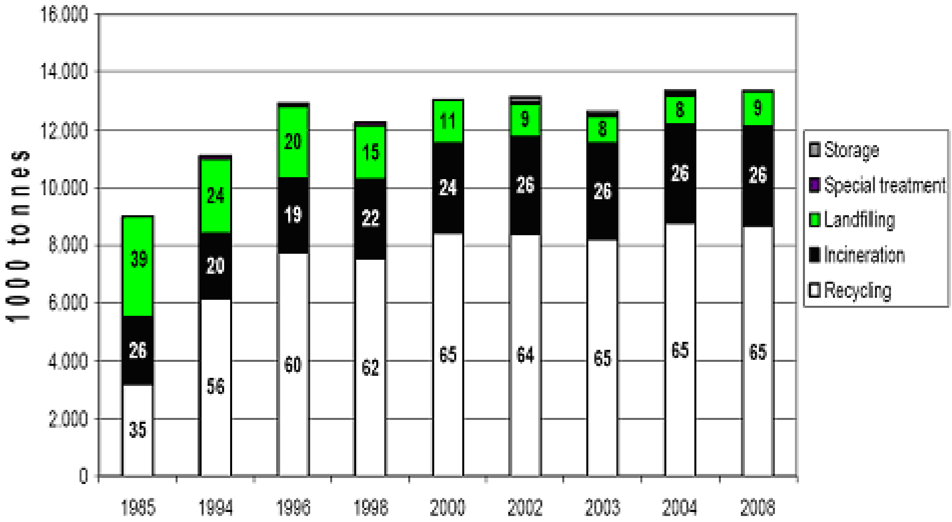
The tax was unable to reduce waste amounts in absolute terms (see Table 2.2). Reductions were experienced in waste amounts brought to treatment plants and took place especially between 1987 and 1993 (see Table 2.3) (Andersen et al, 1997: 6) indicating a fast behavioural reaction following the introduction of the tax. Since 1993, despite the increase of the tax rate, the amount of taxable waste did not decline further (ibidem). From 1994 to 1996, waste amounts have even increased slightly and from 1995 to 2001 there has been an increase in the volume of household waste (Andersen and Dengsøe, 2002: 27).

Table 2.2 Total waste production

	1000 tonnes							Development
	1995	1996	1997	1998	1999	2000	2001	%
Households	2.610	2.767	2.776	2.796	2.963	3.084	3.083	18
Domestic waste	1.628	1.655	1.621	1.702	1.665	1.676	1.701	4
Garden waste	326	401	443	438	464	519	458	10
Packaging waste							107	41
Others	38	72	125	83	163	158	138	259
Institutional/trade and offices	834	851	861	955	955	1.119	1.307	57
Manufacturing	2.563	2.632	2.736	2.783	2.653	2.948	2.616	2
Building and construction	2.559	3.088	3.427	2.962	2.968	3.223	3.391	33
Wastewater treatment plants	1.195	1.212	1.248	1.251	1.379	1.476	1.123	-6
Slay, fly ash, etc (coal)	1.099	2.332	1.770	1.409	1.299	1.170	1.211	-29
Other	6	30	34	18	15	5	37	481
Total	**11.466**	**12.912**	**12.857**	**12.233**	**12.233**	**13.031**	**12.768**	**11**

Source: DEPA 2003.

Table 2.3 Treatment of waste 1985-2004



Source: DEPA 2006.

Accordingly, the tax yield more revenue than was initially expected. Its revenues estimated ex ante at DKK 120 million annually, in 1998 were already reaching DKK 889 million (see Table 2.4) (COWI, 2001). This development might be explained both by increased marginal recycling costs in the remaining waste fractions and, regarding household waste, the positive economic development experienced in Denmark since 1993 (ibidem). The latter might have followed from the approximation of full use of installed capacity in recycling or increased financial costs of new recycling projects. In 1993, the subsidy scheme for recycling projects was abolished (Jens Hansen, Danish Ministry of Taxation, e-mail communication, 12 November 2002). Though after 1993 part of the revenues kept by the Ministry of Finance were still used to finance subsidy schemes for recycling, waste tax revenues were also used to finance clean technology projects to increase the incentive provided by the tax (Betænkning, 16 May 1986, 1779), as well as the Danish tax shift initiated in 1993. Therefore, necessary new recycling capacity might not have been made readily available or it might have involved the need to use more expensive financing sources than before.

Table 2.4 Revenue from the waste tax (million DKK)

	1990	1995	2000	2002	2005
Waste tax revenues	404	619	999	1090	983

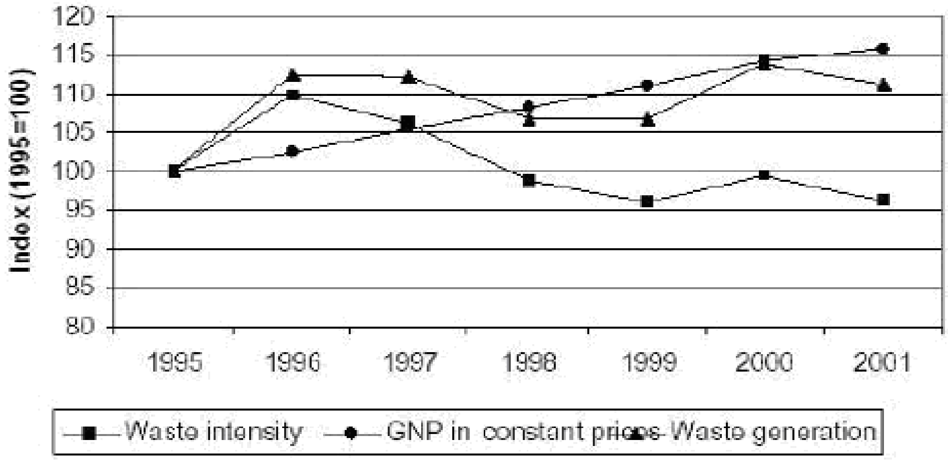
Source: Nordic Council, 2006: 84.

The development at municipal plants, which were the first to be subject to the waste tax, showed a 17% reduction from 1987 until 1993 and a 13% increase from 1994 until 1996 in gross delivered amounts (Andersen et al, 1997: 43). With regard to small private landfills for inert waste and other private landfills, a decrease of 39% in waste amounts was observed from 1990 to 1996, with a 27% reduction in landfills for inert waste (idem, 42). This was a result of either waste diversion from private sites to municipal plants (DEPA, 1999a: 99) or recycling of construction and demolition waste (Andersen et al, 1997: 108-121). And in net amounts, from 1996 to 1998, taxable waste in municipal waste treatment plants and inert waste landfills decreased by 0.5% and 23%, respectively (Dengsøe and Andersen, 1999: 5). This evolution is thought to have been related with the fact that reuse and recycling of waste offered for collection rose from 21% to 50% between 1985 and 1993 (EEA 1996: 54) and this increased further to 61% by 1995 (Ekins 1999: 50). The increased amount of waste removed for recycling comprised around one third of construction and demolition waste (ibidem).

When developments in waste amounts were related to a baseline (representing developments in waste amounts without regulation or taxes), it was possible to notice that without the efforts to recycle construction and demolition waste, residual waste amounts could have increased considerably in the period 1986-1996 (Andersen et al, 1997: 54). By contrast, the change in the net production index followed quite closely the change in industrial and commercial waste (ibidem). However, for households there was a slight de-linking between waste amounts and developments in consumption (ibidem). However, though householders reduced their waste considerably, such a drop might have been overestimated, since part of this waste may have come in the statistics as part of the industrial and commercial waste increase (Andersen et al, 1997: 47).

The long term trend does not point in the direction of decoupling growth in the volume of waste and growth in gross domestic product (hereafter also GDP) (Dengsøe and Andersen, 1999: 72-74). There was rather an approximately constant relationship between them (ibidem). The next graph (Table 2.5) shows the relative changes in GDP, waste generation and waste intensity measured by the relationship between the relative change in waste generation and the relative change in GDP (ibidem). Total volume of waste increased more rapidly than economic growth in the period 1995 to 1996 (ibidem). This was followed by a decline in the volume of waste, which might have been related to the sharp tax rate increase experienced in January 1997 and again in 1999 to 2001 (ibidem). Waste intensity declined until 1999 (decoupling) and has subsequently been relatively constant (ibidem).

Table 2.5 Development in GDP in constant prices waste generation and waste intensity



Source: Waste Statistics 2001, DEPA 2003, Statistics Denmark.

In Denmark, by the mid 2000s technological change had not led to a more efficient use of material input (Brix and Bentzen, 2008: 3). In 2005, the same amount of economic activity was associated with more waste than 10 years previously (ibidem). The intensity effect mainly explained the changes in waste production in the manufacturing sector (Brix and Bentzen, 2008: 16). In other sectors waste intensities were increasing more than what could be explained by economic activity and growth indicating an inefficient surplus in the amount of waste the Danish firms generated over time (ibidem). Less efficient firms, the interest of the entities responsible for waste management in keeping high waste flows and the insufficient level of the waste tax were mentioned as the causes of such development (Brix and Bentzen, 2008: 14-15).

4.2 Waste streams most affected by the waste tax were heavy ones entailing low recycling costs

Analysis of the waste streams allows the evaluation of which ones were affected by increased separation and recycling. Most significant reductions in residual waste amounts occurred in heavy waste fractions for which the costs of disposal for recycling were low, namely construction and demolition waste, garden waste and bulky waste. These were also the waste streams for which most municipalities set voluntary collection schemes following profitable recycling.

In total, the most significant reduction took place for construction and demolition waste (EEA, 2000a: 49) (total recycling rate was estimated at around 79-80%, Andersen et al, 1997: 50), for which collection 71% of the municipalities created voluntary schemes (Andersen et al, 1997: 75). The second largest share was the composting of garden waste, for which 82% of the municipalities adopted voluntary collection schemes in 1996 (ibidem). Also in 1996, bulky waste was collected under voluntary schemes in 87% of the municipalities (ibidem)

The Danish Statutory Order on Waste did not establish any requirements for recycling of organic waste, but many waste companies and local administrations (50% of them in 1996, ibidem) paid special attention to this waste fraction, which evolved positively. However, even though recycling increased, it must be assumed that some garden waste was previously burned locally, so that the reduction in taxable amounts hardly accounted for more than half this amount (Andersen et al, 1997: 57).

Though iron and scrap metal is a heavy waste type sensitive to a tax on waste, the tradition of collection was established long before a tax was introduced (Andersen et al, 1997: 49, 83). Interviews in the iron and metal sector showed that scrap iron had a positive economic value that promoted collection (Andersen et al, 1997: 49). By 1996, 80% of the municipalities had adopted voluntary collection schemes for this kind of waste (Andersen et al, 1997: 75).

The mandatory collection of glass and paper only accounted for a minor part of the reduction in taxable waste amounts (Andersen et al, 1997: 36-57). Following the requirements of the Danish Statutory Order on Waste, the collection of glass and paper increased between 1986 and 1994. However, the rate of recycling of these materials did not increase proportionally. Only approximately two thirds of glass were collected and for paper only half the potential was collected (Andersen et al, 1997: 49). The increase in glass collection was also relatively moderate from 1986 to 1999, especially considering the extensive effort of establishing bottle banks (ibidem).

4.3 The waste tax did not influence to a decisive extent Danish waste management systems

The tax is understood as having contributed to shifting the financial balance towards recycling, without being totally decisive (Andersen et al, 1997: 6-7). The behavioural steering experienced in diverting waste from traditional disposal to recycling followed not only from the waste tax but also from other taxes that increased the price of waste disposal as well as from the recycling policy adopted and other more general political reasons. The waste tax enhanced the environmental effectiveness of such policy by increasing the number of waste streams (heavy and easily disposed ones) where recycling was profitable and by providing revenues to increase the availability of non-mandatory recycling schemes, especially until 1993, when the subsidy schemes for recycling were discontinued. The fact that municipalities kept a direct interest on the waste service market might help to explain their relatively low reaction to the tax as far as the design of the waste management system was concerned.

The quite extensive collection schemes introduced by the municipalities exceeded the requirements of the 1983 Danish Statutory Order on Waste (Statutory Order 568, 6 December 1983). In a questionnaire study carried out among local administrations, in general both political (i.e. the desire of the local council to increase recycling) and economic reasons were mentioned to explain the offer of facilities for reuse and recycling (Andersen et al, 1997: 76).[34] The intent to reduce total treatment costs as well as the desire to comply with the objectives of the Government's Plan of Action for Waste and Recycling were mentioned as the two second-most important factors, after the political reason, to introduce such schemes (ibidem).

Regarding the design of recycling schemes for heavy waste fractions, a 30% of respondents attributed "some" impact to the tax, “but all in all the result seems to be that the waste tax did not influence such design to a significant extent” (Andersen et al, 1997: 83-84). After 1997, some local authorities changed to

[34] Andersen et al, 1997: 73, do not provide precise reference on when the questionnaire, which got responses from 189 local administrations and was carried out in connection with the 1997 study, was replied to.

weight-based fee systems for household waste (Dengsøe and Andersen, 1999: 72-74). However, the reduced number of municipalities doing it led to a low impact of such systems on total waste figures (ibidem).

The tax mainly influenced municipalities to introduce collection schemes for some of the heaviest waste fractions regarding which there was no mandatory collection but which recycling became profitable following the introduction of the tax, such as bulky waste, garden waste, construction and demolition waste and organic waste. For these waste streams about 70-80% of local administrations with collection schemes mentioned the waste tax as being important for the economy of such systems (Andersen et al, 1997: 8).

The active role municipalities played in the waste service market might help to explain the low influence of the tax on the design of waste management systems. The waste tax a municipality paid depended not only on the amount of waste it recycled and incinerated compared to the one it landfilled, but also on the efficiency of its collection and recycling operations and whether it relied more on the operation of incinerators or landfill (both of which were operated by public authorities when dedicated to handling household waste).

The main aspects professional players in the waste service market considered whilst deciding on waste management included the cost of operating a recycling plant *versus* the sales price of collected and processed material; the cost of operating an incineration plant plus the revenues from the sale of power and heating from such a plant *versus* the cost to operate a landfill; the cost for the waste company to collect waste *versus* the possibility and cost to subcontract totally or partially this task; the possibilities of minimising waste amounts through in-plant recycling or sale of by-products, and transport costs related to recycling *versus* transport costs of landfilling or incineration when the recycling plant was located in another region (Andersen et al, 1997: 7, 34-35).

4.4 There were considerable differences between the various recycling rates for industry

Following the less filtered price signal reaching the waste producers covered by assignment schemes and consequently their relatively higher tax awareness, the best results in terms of behavioural steering towards more sustainable waste management techniques, namely recycling, were expected in the industry. However, except for the construction sector, recycling has not been significantly increased in industry. This was mainly due to the low importance of waste disposal costs in the production cost structure. The low level at which the tax was set did not help to raise awareness of waste costs in industry.

The data are inconclusive regarding whether taxed waste quantities would have increased in industry and commerce in the absence of waste policy and waste taxation. The most important motivation for recycling mentioned by the respondents to an enquiry carried out by Andersen and Dengsøe (2002) was the possibility of revenues from the sale of waste and residual products (ibidem). The desire to reduce costs of waste disposal came second (ibidem). Only two out of the eight companies that mentioned the reduction of costs as an important factor argued that the waste tax had a direct impact "to a large extent" or "to some extent" in relation to other cost components (Andersen and Dengsøe, 2002: 28).

In general, there was not a very precise knowledge of actual costs of waste disposal in the sector, including the effect of the waste tax as a cost component in relation to other cost components, since responsibility for physical disposal was often separated from financial responsibility (ibidem). The modest attention paid to economic aspects of waste management is related to the relatively low level of waste disposal costs (ibidem). These normally did not exceed 0.5% of companies' turnover ranking lower in cost consciousness than energy, whose costs rarely exceed 2% of turnover (Andersen et al, 1997: 72).

Attention to waste management was highest in waste-intensive sectors (Andersen and Dengsøe, 2002: 27). Accordingly, following the 1997 tax increase, considerable differences between recycling rates for industry were found, ranging from high (up to 90%) for traditional 'chimney industries' (such as iron and metal, shipyards and breweries) to low (down to 5%) for the plastic and pharmaceutical industries (Dengsøe and Andersen, 1999: 5).

4.5 The design features of environmental taxes were fairly included in waste tax institutional practice

The design features of environmental taxes regarding the use of environmental criteria, the reference to precise environmental objectives, the institution of tax awareness and tax avoidance in the tax design, the use of a good proxy for polluting emissions as tax base and the tax incidence on polluters able to avoid pollution were relatively well instituted in the waste tax design. The same kind of inclusion has not been so accurate as far as the reference of the tax rate to behavioural change is concerned. This might have been a consequence of the interest that the entity administering the tax, namely the Ministry of Environment, kept on its revenues, which were allocated to this Ministry until 1993.

However, this slight positive image gets lost when the analysis focuses on the waste tax institutional practice. This is explained by the overlap of the taxpayer with the entity holding *de facto* tax powers and of both with the main professional player in the waste service market, namely the municipalities, which hence kept ambiguous interest regarding the tax base. Following this institutional framework, both the pricing system and the waste management system as a whole have come to play a decisive role in the (un)success of the waste tax. These factors provided institutional filters that have significantly disturbed tax awareness and tax avoidance instituted in the legal waste tax design.

The behavioural impact of the waste tax on waste producers was influenced by the *effective waste disposal price*, both in relative and absolute terms, and the waste producers' perception thereof. Furthermore, it depended on the availability of avoidance strategies. The *effective waste disposal price* depended on its several components, the main ones of which were the tax component, the charge on waste collection and the gate fee at treatment plants. The share of the tax component in relation to other elements depended to a large extent on the kind of waste fractions considered. However, for industry and commerce, in general such a component has been too low to lead to positive behavioural change, and for households it has been too filtered to provide an effective stimulus toward the environmental hierarchy of waste.

The institutional framework sheltered waste producers from the price signal provided by the tax. Following the rules and agreements on collection and fees, waste producers were unable to estimate the price relationship between the three options available for waste disposal (i.e. landfilling, incineration and recycling). Since the transaction (waste disposal) was arranged through an intertwinement of different players with diverse price components (Andersen et al, 1997: 7, 34-35), the waste tax was only one of the components of the *effective waste disposal price* and arrived at the waste producer intermediated by the waste bill. This flattened, hid, delayed and changed the relative terms of the price signal provided by the tax.

Institutional filters in the waste service market, namely the market structure and the billing system, tended to blur the price signal provided by the tax, which affected the sensitiveness of waste producers to such a signal. Institutional filters operated by cutting the linkage between the payment and the damage (turning the tax into a fixed price), by hiding the price signal provided by the tax (making it an indiscriminate component of the *effective waste disposal price* or a component of the tax bill, the rent or the contribution for the residents' association), by delaying the price signal provided by the tax or by changing it relatively, in the sense that the hierarchy of waste management techniques communicated by the tax did not always arrive to waste producers undisturbed.

4.5.1 Environmental criteria were relatively well embedded in the waste tax

The waste tax had from its introduction a precise and direct environmental objective, namely to influence waste management techniques. However, the hierarchy of these techniques followed from environmental criteria only up to a certain extent. The tax aimed at diverting waste streams towards more sustainable waste management techniques by increasing reuse and recycling and reducing landfilling and incineration ('the Danish waste hierarchy', DEPA, 1999a: 96). In the successive tax rate changes reference has been made to action plans for waste and recycling. No quantitative targets were introduced in the law until 1990, when a 15% reduction in taxable waste was mentioned in the Finance Act (Andersen et al, 1997: 24).

The use of environmental criteria in the waste tax design was only fairly good. The 'Danish waste hierarchy' did not accurately match the sustainable management of waste following from environmental criteria. This prioritises reduction over reuse and this over recycling, which is preferred to energy recovery, disposal being at the bottom of the pyramid (Art. 3 Council Directive 75/442/EEC, 15 July 1975, OJ L 194/39, 25 July 1975, as amended by Council Directive 91/156/EEC, OJ L 78/32, 18 March 1991, and Council Directive 2008/98/EC, 19 November 2008, OJ L 312/3, 22 November 2008). The hierarchy adopted in the waste tax did not consider reduction as the first objective in waste management, since stemming the increasing waste amounts was a means to achieving reduction in traditional waste disposal. According to motivations used at the inception of the tax, apparently this was not expected so much to lead to waste reduction across the board but to have such an effect where the marginal treatment costs of recycling were lower than the waste tax rate (DEPA, 1999a: 96).

Furthermore, it identified energy recovery with incineration, though this is only one of the possible ways of doing it. The prioritisation of incineration over landfilling might have been related to the size of the country. In densely populated countries like Denmark, at least 50% of all waste tends to be incinerated because these countries' incinerators benefit from economies of scale and incineration reduces the volume of household waste by 90% (OECD, 2004: 27). Furthermore, in 1999 energy recovery seemed to have been downgraded in the 'Danish waste hierarchy' following the introduction of a tax on heat produced from district heating systems via waste incineration, which eliminated the previous tax incentive to use waste, rather than fossil fuels, to produce heat (Nordic Council, 2006: 84).

Environmental damage is correlated to the composition and amount of material flow entering the ecosystem. The waste's weight is suitable to measure the amounts of such material flow. However, there was no reference in the waste tax design to the composition of such flow. Waste tax rates were only differentiated according to the kind of waste management technique used. They did not discriminate problematic waste streams. For example, plastic waste and electronic waste are considerably more problematic to recycle than ordinary household waste. This contrasted with the weight-based tax on packaging introduced in January 1999, which reflected the environmental impact from each material by working out a number of environment indices on the basis of 'cradle to grave' (life cycle) assessments (Dengsøe and Andersen, 1999: 5).

However, the use of several different instruments (including taxes) for different kinds of main waste in Danish waste management policy might have allowed the regulator to roughly take into account the composition of the material flow (Andersen et al, 1997: 118). Following the higher administrative costs further tax rate differentiation involves, it might have been considered more expedient to establish take-back schemes, perhaps supported by deposit systems, for problematic wastes, and to use the waste tax as a more general instrument aimed at total waste amounts, ignoring accurate cost internalisation of waste impacts (ibidem).

4.5.2 Tax awareness and tax avoidance were greatly disturbed by institutional filters

Tax awareness and tax avoidance were instituted in the Danish waste tax design by charging the tax according to waste weight and rewarding recycling, respectively. However, these features tended to be neutralised in their transposition to municipal institutional practice, which introduced fiscal illusion to a great extent whilst passing the tax to waste producers especially in collected waste. Therefore, tax awareness among polluters was on average very low, being the highest among waste companies and waste producers bringing their waste directly to treatment plants. Furthermore, the availability of tax avoidance strategies in practice depended on the interest of the municipalities in providing them, which has sometimes been challenged following the role municipalities played in the waste service market as professional players.

The clear linkage between tax payments and waste disposal was expected to allow polluters to realise the size of the financial loss it involved, to understand the cause of the payment and the means to avoid it. However, the municipal charging schemes instituted fiscal illusion in waste management institutional practice. Following the integration of the waste service market, the waste tax tended to be billed together with other items in assigned waste, which led to fiscal illusion. In collected waste, the delayed charge on a fixed base cut the linkage between tax payment and waste disposal (and consequently environmental damages), leading to fiscal illusion and low capacity of tax avoidance. Both these aspects are thought to have negatively impacted on the environmental effectiveness of the tax keeping it from reaching more substantial reductions even without increase of the tax level (Andersen et al, 1997: 8).

A. The integration of the waste service market tended to reduce tax awareness in assignment schemes

Waste producers were either covered by a collection scheme or an assignment scheme. Waste producers were under an assignment scheme when they were not covered by a collection scheme, but requested to deliver waste to a plant under the terms of municipal assignments, being in charge of transport to such plant (Andersen et al, 1997: 89). These were typically industrial or commercial companies (ibidem).

Collection and transportation of waste has brought about an independent market for purchase and sale of services (Andersen et al, 1997: 31). Whereas operators used to be mainly carriers, the market developed into vertical and horizontal integration with an increasing number of private waste companies operating as 'turnkey' contractors (ibidem). Through EU tenders, these companies competed with traditional municipal companies, which held regional and local monopolies, on delivering services in the waste market (ibidem). The largest of these companies also operated its own plant for disposal and treatment of waste, but in general, the latter were operated by the local council (ibidem).

Waste producers not covered by collection schemes usually contracted a professional waste carrier (Andersen et al, 1997: 88). The way in which the price signal from the tax was passed on these waste producers depended on the pricing used by carriers (ibidem). Most waste producers paid the *effective waste disposal price*, comprising both direct disposal costs (rent of containers, collection, transportation and treatment fee) as well as the waste tax, which was not usually specified separately on the invoice unless the company contracted waste transport and waste treatment to different entities (ibidem). In less frequent cases, the waste producer himself brought waste to the plant, being the waste tax calculated on the basis of weight, and visibility and simultaneity were achieved (Andersen et al, 1997: 89).

B. The charge method used tended to disturb tax awareness and tax avoidance in collection schemes

For households covered by municipal waste collection schemes the reward of tax avoidance strategies and tax awareness was disturbed to a great extent by the charge system used by the municipalities. The weight-based tax was often integrated into volume-based waste collection fees, due to financial arguments, complexity-avoidance arguments, fear of fly-tipping and lack of reliable technology (Andersen et al, 1997: 84). The local council could also, in part or totally, finance waste expenses via taxes on real estate (Andersen et al, 1997: 88-89). This pricing meant the absence of a direct linkage between waste disposal and tax payment with the consequent reduction of alternatives to payment. Furthermore, since waste collection fees were part of other prices, the fiscal illusion problem reported for assignment schemes in the previous section also occurred in collection schemes.

Householders, and others covered by collection schemes, had limited options concerning their management of the waste tax, as they could not individually reduce waste costs by reducing waste amounts (Andersen et al, 1997: 73). Their ability to separate part of the waste for recycling also depended on the extent of collect or bring systems in the community (ibidem). Especially for single-family houses, increased separation of waste only led to unexploited capacity in waste bins (ibidem). With volume-based fees (by bin or bag) fees were charged for normal rather than actual waste production (Andersen et al, 1997: 89).

Waste producers in blocks of flats had more opportunities for tax avoidance. By increased separation and recycling, the number of waste containers could be reduced, which allowed cost reductions (Andersen et al, 1997: 90). However, such a possibility was not available for waste producers in publicly-owned rented flats, since their fee was integrated in the cost-based rent (Andersen et al, 1997: 32, 89). Weight-based fees for these waste producers, though in principle possible, since the full waste collection expense could be passed on to the tenant, would have required an amendment to legislation on cost-based rents to make payment for waste disposal separable from the general rent (ibidem).

The volume-based charge further disturbed the adoption of tax avoidance strategies following the incentive it provided to free-riding. Recycling facilities were made available but their voluntary use involved costs, namely time and effort to bring different waste fractions to the right containers. Collection schemes reduced these costs. However, Danish citizens might still not have felt sufficiently compensated for the additional costs of separating waste for recycling when environmental and financial gains from increased recycling benefited all citizens in the area of the waste company. This was the case when volume-based waste collection fees were used, since reduced waste taxes were distributed over the normal fee for waste disposal (Andersen et al, 1997: 90). The use of separation and recycling facilities was significantly higher in municipalities with weight-based fee systems than in those where the charge was done on a fixed amount or according to volume or frequency of collection (Andersen et al, 1997: 8).

Furthermore, the integration of the waste tax in the waste fee and the billing of the latter together with other payments as well as the time gap between waste production and payment lowered tax awareness. For owner-occupied flats the fee would often be integrated into the contribution to the residents' association and for rented houses it was integrated into the rent. For individual households it was often paid together with taxes on real estate to the local council estate (Andersen et al, 1997: 88). Both for owner-occupied flats and rented ones there was a time gap between waste production and payment, since in practice the fee was settled on the basis of waste amounts of the preceding year (Andersen et al, 1997: 89). The fee was updated for tenancy when the cost-based rent was adjusted and for owners when the local council fixed the rate of conversion between weight and volume on the basis of last year's operating result (ibidem).

C. Availability of tax avoidance strategies explained the (un)success of the waste tax

Tax avoidance was instituted in the tax design and was further developed via the application of the tax revenues. The availability of tax avoidance strategies in practice was determinant for the success of the tax in steering behaviours towards recycling among the households (Andersen et al, 1997: 8). Though as a rule recycling schemes were widely available for some waste streams, at least until 1993, when the costs of recycling seem to have increased with consequent increases in taxable waste amounts (Andersen and Dengsøe, 2002: 27), some strategic behaviour by the municipalities was reported. These might have restrained the recycling capacity made available to households in order to fully exploit installed traditional waste treatment capacity.

A refund scheme for recycling companies was introduced in 1990 (Law Proposal 176, 11 February 1986, 4450; Andersen et al, 1997: 26). This allowed the refunding of the tax paid on waste subsequently removed from registered plants, providing an incentive to taxpayers (i.e. registered waste treatment plants) to set separate collection schemes to raise efficiency in recycling (ibidem). By not granting a full refund for the waste tax to recycling companies the incentive to prevent residual waste was maintained, as these companies often generated relatively large amounts of residual waste that had to be landfilled or incinerated (ibidem). However, incinerators also benefited from this measure with regard to residuals subsequently removed usually to landfills (Andersen et al, 1997: 23).

The implementation of recycling schemes was further enhanced with the waste tax revenues. A subsidy scheme for recycling projects was introduced before the tax and financed since the adoption of the latter until 1993, when the scheme was abolished, by the waste tax revenues. This was set to increase the incentive provided by the tax for recycling, since the tax rate of DKK 40/tonne fixed at the introduction of the tax was considered too low to reduce taxable waste amounts (Andersen et al, 1997: 23). After the tax revenues were transferred from the Ministry of Environment to the Ministry of Finance tax revenues were still used for recycling but to a lower amount, since they were also used to support clean technology projects (Betænkning, 16 May 1986, 1779).

It has been discussed whether the municipalities, who were in charge of approving the level of the waste collection fees, did not keep an ambivalent interest regarding the reduction of waste management costs, inter alia by making recycling schemes available. On the one hand, they developed an interest in reducing their waste disposal expenses since these burdened municipal budgets and were experiencing considerable increases (Andersen et al, 1997: 33). But on the other hand, waste companies (mostly owned by (inter)municipalities) calculated fees for their services according to a cost principle and also had a monopoly on local collection. Therefore, it was questioned in economic literature on public service companies whether they had an economic interest in minimising costs (ibidem).

As a rule tax avoidance strategies have been available in practice to waste producers. By the mid 1990s, recycling facilities were well established in Denmark (Andersen et al, 1997: 8). Municipalities had established separate collection schemes for recyclable waste far above the mandatory requirements (Andersen et al, 1997: 75). However, it was reported a lack of efficient collection schemes for organic waste and garden waste in some cities (Andersen et al, 1997: 88-102). This was explained by the desire to exploit available incineration capacity (ibidem).

4.5.3 The waste tax base was a good proxy for polluting emissions

The waste tax was raised on a good proxy for polluting emissions (and consequent environmental damages), but not a good proxy for specific polluting emissions since the composition of the flow was not considered. Environmental damage is caused by the material flow entering the ecosystem. Waste disposal is an example of such flow. Waste weight is positively correlated with polluting emissions and allows measuring the amount of such flow. Therefore, the tax base could only relatively well measure environmental damages. By increasing the price of the tax base it was possible to increase the price of environmental damages, but the correlation between these two variables, though positive, was not constant.

The narrow tax base at the introduction of the tax, which only covered waste brought to municipal waste treatment plants, is not thought to have allowed tax avoidance strategies as polluting as the ones taxed to a significant extent. It was calculated that a waste tax of DKK 40/tonne did not outweigh the extra transport cost incurred when using non registered plants (Andersen et al, 1997: 40). Furthermore, the tax base was extended to cover these plants shortly after the introduction of the tax. Environmentally harmful tax avoidance strategies might have been predominantly related with illegal dumping. Though the drop in waste quantities was matched by increased recycling and statistical data did not show fly-tipping as a serious problem (OECD 1994b: 64, Andersen et al, 1997: 8, Cooper et al 1999: 6, COWI, 2001: 223), illegal disposal has been mentioned especially in municipalities using weight-based collection fees (OECD, 2004: 53).

Consequently, the environmental impact of the successive extensions of the tax base is thought to have been low. Though environmental arguments have been used to support them, extensions were often also supported by fiscal interests. For example, the decision to cover private landfills for inert waste in the 1990 Statutory Order provided considerable revenue (DKK 340 million in 1990) and delivered little environmental improvement (Andersen et al, 1997: 24). According to the Ministry of Environment, its main purpose was to increase recycling of construction and demolition waste that had previously been landfilled at those sites with an estimated waste diversion of around 1 million tonnes, corresponding to a decrease of 15% in total waste amounts (ibidem). However, most progress in this waste had already been experienced since the introduction of the tax (Andersen et al, 1997: 22-23).

4.5.4 Waste tax rates were not always referred to behavioural change

Though the tax rate has been initially referred to pollution abatement costs and also to relative polluting impacts from 1992 onwards, it was not always referred to environmentally correct behavioural change. This failure was rooted in the insufficient levels at which the tax rates were set. Since these did not consider relative *effective waste disposal prices* they were unable to compensate for the costs of recycling with regard to all recyclable waste streams as well as to compensate for lower municipal fees on landfilling as compared to incineration and make this price-competitive with landfill in all the municipalities. Furthermore, the low waste tax levels in absolute terms were unable to raise the position of waste management in the companies' hierarchy of attention. These insufficient levels have not been compensated for by the overlap of tax charges on the same tax base and have raised more concerns during the time following the lack of revision of the waste tax rates after 2001.

At the introduction of the tax, the tax rate aimed at a level sufficiently high to induce recycling taking into account the *effective costs* the latter involved (i.e. considering recycling as the most preferable pollution abatement strategy). Starting in 1992, the waste tax rates have been gradually differentiated according to relative polluting impacts of the waste destinations available, evolving from the higher level to the lower level according to the following order: landfills, incinerators with heat recovery capacity, incinerators with electricity and heat recovery capacity and recycling facilities.

The tax rate applied to waste brought to recycling facilities was in practice 0% since such waste was exempted from taxation. Therefore, recycling was the legal waste management technique most advantaged by the waste tax. And since recycling still entailed positive costs (e.g. the costs of separating, storing and bringing the waste to recycling facilities), waste avoidance was the most advantaged behaviour regarding waste management by relative *effective prices*.

For recycling to be an appealing strategy and induce positive behavioural change it is not enough that it is less taxed than other alternative waste management techniques. It must also entail an *effective waste disposal price* lower than those other techniques. Therefore, following the positive and increasing costs of recycling it was necessary to introduce successive increases in the waste tax rate to make it profitable.

However, in spite of those increases, the waste tax rates performed good behavioural steering towards the environmental hierarchy of waste management mainly in heavy waste streams with low recycling costs. Following these reduced costs, recycling in these waste streams became profitable even at low levels of the waste tax. However, when the costs of disposal for recycling increased, as for example in the case of paper, the waste tax rates proved too low to make recycling a profitable strategy vis-à-vis traditional disposal. The tax rate on waste brought for incineration might have been too low to compensate for the high costs of recycling paper, especially at times when the market prices of waste paper were low.

The failure to refer the tax rates to relative *effective waste disposal prices* with the consequent incapacity of the tax to lead towards the environmental hierarchy of waste management was also experienced in another case. In the tax rate differentiation between waste brought to landfills and that brought to incinerators, neither the 1992 nor the 1997 levels applied to landfilled waste were able to compensate for the relatively lower fees some municipalities charged in landfills compared to those charged in incinerators. Therefore, in some municipalities landfilling continued to be cheaper than incineration.

In other cases, the waste tax rates were too low in absolute terms and their low level hindered the capacity of the tax to induce waste reduction and recycling. For example, the tax level for commercial and industrial waste should have been higher to raise the importance of waste management in the hierarchy of attention in waste-extensive industries, which were those where higher improvement possibilities were available. The price signal provided by the tax in some waste streams was strengthened by other tax instruments. These enhanced the environmental effectiveness of the waste tax, but were still unable to bring the *effective waste disposal prices* up to levels able to compensate for the referred low waste tax levels.

The low levels of the tax were further aggravated in the period 2001 to 2005 following their relative stability. The regulation of the Danish waste management system did not change significantly in the period from 2001 to 2005 (Brix and Bentzen, 2008: 15). Tax rates remained constant at the 2001 level (DKK 330/tonne for waste brought for incineration and DKK 375/tonne for waste brought for landfilling) (ibidem). These amounts were minute compared to the waste disposal expenses. The average costs of handling fees at the treatment facilities (exclusive of the waste tax) in September 2008 were as follows: landfilling (DKK 311/tonne), incineration (DKK 259/tonne), composting of garden waste (DKK 175/tonne) and reuse (DKK 556/tonne) (Brix and Bentzen, 2008: 15).

A. Tax levels were too low to always compensate for the costs of recycling

The level at which the tax was set was especially important for its environmental effectiveness since waste reductions in landfill and incinerators depended to a great extent on the diversion of waste streams from conventional disposal to recycling, which depended on the profitability of recycling. And the latter was directly influenced by the level of the waste tax as a component of the *effective conventional waste disposal price*. Following the successive tax rate increases, during the 1990s recycling was made profitable for heavy waste for which there were alternatives to traditional disposal at low cost, the tax being helpful in supporting the Danish recycling policy. However, it might have never been sufficiently high to make recycling profitable when this involved relatively high costs, like in the case of paper.

Whether the tax was able to shift the balance between recycling and conventional disposal would depend both on the tax level and the price of recyclables. Companies were able to save costs by reducing costs of transportation or increasing waste compression (when charged per volume of waste). Therefore, it was not necessarily true that a tax increase would result in a reduction in waste amounts, though it would often draw attention to waste management (Andersen et al, 1997: 69). A financial incentive provided by a weight-based waste tax was expected to lead to reductions in waste amounts where waste was most concentrated (heaviest) and cheapest to dispose of in an alternative way, increasing profitability of recycling. The results obtained with the Danish tax confirmed this expectation.

The important increases in recycling matched very closely by the reductions in taxable waste amounts during the 1990s indicate that an important part of the behavioural effect of the waste tax came from diverting waste from traditional disposal to recycling. Furthermore, these results confirmed the hypothesis that reductions in waste amounts following the introduction of a weigh-based tax occur mainly where waste is most concentrated (heaviest) and cheapest to dispose of in an alternative way. High reductions were experienced in recyclable heavy waste streams, namely construction and demolition waste (63%), household waste (16%) and other wastes (where slag and sludge were included) (22%) (Andersen et al, 1997: 6).

Marked increases in recycling were noticed in construction and demolition waste, compost and bulky waste (Andersen et al, 1997: 8, Dengsøe and Andersen, 1999: 5; within household waste, two thirds were garden waste and bulky waste, Andersen et al, 1997: 57). The exceptionally good results obtained in the recycling of heavy construction and demolition waste that took place within the construction sector is explained by the fact that companies with very large amounts of this kind of waste achieved large savings by prioritising separation and recycling (Andersen et al, 1997: 111) compared to the relatively high and unshrinkable tax payment per weight of waste brought to landfills they would otherwise have faced.

Symmetrically, following the high costs of recycling paper and the relatively low costs of incinerating waste, it was cheaper to incinerate paper than to recycle it. Compared to the aforementioned, mandatory collection of glass and paper was only able to allow a smaller proportion of total decrease in waste amounts (Andersen et al, 1997: 6), because, despite the several increases in the waste tax rates, their level was still too low to compensate for the high costs involved in recycling these materials.

Successive tax rate increases did not compensate for growing waste management costs

During 1990, waste management in Denmark experienced growing costs. These evolved similarly for traditional disposal and recycling. During the same period, and since its introduction, the tax was unable to cover all the costs of traditional disposal. Therefore, in spite of the successive tax rate increases, it was unable to make recycling profitable for all recyclable materials.

For the collection of waste (domestic waste and some recyclable materials, namely bulky waste, whereas others had to be brought to recycling facilities, Andersen et al, 1997: 100) from households and companies, the local council normally fixed a waste collection fee that would often depend on volume and frequency of collection of waste bins (Andersen et al, 1997: 88). This fee should be fixed according to the non-profit cost-coverage principle, on the basis of total disposal costs, i.e. initial and operating costs of collection, transportation, treatment and taxes (waste tax and VAT) (Andersen et al, 1997: 88,32).The way in which the individual local council or company distributed costs between different users was a matter decided locally (idem, [88-102]).

Competitive recycling requires a tax level sufficiently high to compensate for the additional costs involved in the disposal of recyclable materials. In addition to the tax, total waste disposal costs would normally also comprise cost components such as rent of containers, transportation and fees for landfilling or incineration, as well as internal staff and surface costs (Andersen et al, 1997: 34). Apart from waste collection fees and the waste tax, the same cost components would in principle apply to recycling (ibidem). In addition to these, waste producers incurred costs connected with sorting and separate collection of recyclable materials. Actual waste disposal costs, also for recyclable materials, were determined to a certain degree by the competition in the local waste service market (ibidem).

At its introduction, the tax was estimated at 13-16% of the actual waste collection expenses (Law Proposal 176, 11 February 1986, 4433, estimated annual waste collection expenses per household were at approximately DKK 500, whereas the tax (at that time DKK 40/tonne) was estimated at DKK 35 annually). Following the 1997 change, this percentage was estimated to have increased to 18-28% (Andersen et al, 1997: 32). By the end of the 1990s, and though the waste tax rates had been raised several times, the waste collection fees had not yet reached 100% of the actual waste collection expenses and continued to be collected often according to unit of collection (ibidem). As already mentioned, this differential has increased further during the 2000s following the lack of revision of the waste tax rates.

The disposal costs of recyclables, likewise those of the remaining waste, increased during the 1990s forcing the frequent revision of the tax rate accordingly. In 1985 the tax rate advised to make recycling competitive was at least DKK 30/tonne (Andersen et al, 1997: 22). But in 1989, at the first evaluation of the waste tax it was revealed that the adopted rate of DKK 40/tonne was too low to have a general effect on waste management, except for construction and demolition waste and heavy industrial and commercial waste where an effect was noticed, as waste amounts fell by some 200,000 tonnes from 1987 to 1988 (Andersen et al, 1997: 23). In connection with this evaluation, an assessment by experts indicated that a rate of DKK 100/tonne would be necessary to obtain a significant effect (Andersen et al, 1997: 23-24). The tax rate adopted in the Finance Act of 1990 was DKK 130/tonne and other instruments were also adopted to address the recycling target (ibidem). But again in 1996 the tax rate was considered too low to offset the additional costs connected with sorting and separate collection of recyclable materials (Andersen et al, 1997: 7). Therefore, in 1996 it was necessary to introduce a new increase.

The waste tax level could not compensate for relatively high recycling costs: The case of paper

The availability of containers does not necessarily lead to their use. The costs involved in the effort of bringing the waste to the right container can be reduced following the adoption of collection schemes. However, if recycling is not profitable such schemes will experience a rate of use below their potential. This was the case regarding materials for which the waste tax rates could not compensate for the costs involved in recycling, such as paper, which disposal had to be paid for following the frequent negative market price for waste paper. Following the low economic interest of the municipalities in recycling paper, though forced to implement collection schemes by requirements of the Danish Statutory Order on Waste (in 1996, 99% of the municipalities had implemented separate collection of paper/cardboard and glass following the requirement of the law, Andersen et al, 1997: 75), they might not have induced waste producers to use them (e.g. by using volume-based collection fees that allowed free-ridding or by establishing a low periodicity for collection).

Concerning paper/cardboard, though collection increased between 1986 and 1994, only half the potential was collected (Andersen et al, 1997: 49). Waste paper was not sold to recycling, but disposed of with other waste. These results seem to have been a consequence of the almost constantly negative sales price of waste paper, in contrast with the growing market for used construction materials which made it easier to sell them before they were deemed as waste (Andersen et al, 1997: 70). Limited gains from recycling paper were due to the considerable costs involved, primarily for containers' rent and collection (ibidem). The tax rate was not able to compensate for the costs involved in recycling (investment and operating costs) and make recycling financially attractive. Since unexploited capacity in containers for residual waste as a consequence of recycling entailed extra costs for the company and all waste could always be disposed of as residual waste, a conservative estimation of the capacity to recycle paper was reported in companies, namely universities (ibidem).

B. Tax levels were too low to make landfills competitive with incinerators in all the municipalities

Following the insufficient level at which the tax differential was set, in 1997 the tax rate differentiation between waste brought to landfills and waste brought to incinerators in many municipalities was thought to still have no direct impact on the choice of the management technique applied to waste as far as landfilling versus incineration was concerned. The higher tax rate on landfilling than on incineration was unable to compensate for the lower municipal fees charged in landfills when compared to those billed in incinerators. From 1997 onwards, the behavioural steering effect of the tax rate differentiation ceased anyway given the obligation to incinerate waste suitable for incineration introduced then, following the proposal made in the Government's Plan of Action for Waste and Recycling for 1993-1997 (Andersen et al, 1997: 25-26).

The reasoning behind the 1992 tax rate differentiation was that differences between municipal gate fees for landfilling and incineration led to an economic favouring of landfilling of some DKK 150-200/tonne on average, though with large local variations (Andersen et al, 1997: 25). These might have been influenced by the extent to which municipalities operated incinerators or landfills. Following the referred average difference in the *effective waste disposal prices* of incinerators and landfills, it is evident that the tax differential set (DKK 35/tonne in 1992 and DKK 75/tonne in 1997) could not fully outweigh the differences in municipal fee rates in all the (inter)municipalities (ibidem). Though it might have had some kind of effect on the willingness of the municipalities to host incinerators in the future, lack of data hinders conclusions regarding this aspect.

At a time of loss of yield due to the parallel abolition of the tax on milk cartons, fiscal considerations might have played a role in the decision regarding the waste tax rate. Apparently there was a desire to attract waste from Germany (Andersen et al, 1997: 26). The Act stated that plants receiving only imported waste had to be registered (ibidem). And, though it claimed that an increase in the tax rate was supported by the argument that total disposal costs in Germany (around DKK 7-800/tonne) induced interests in exporting to Denmark, after the 1992 amendment, *effective disposal costs* in Denmark (municipal fees plus waste tax) still remained below the German level (ibidem).

C. Tax levels were too low to induce behavioural change in the industry and commerce

The waste tax rates were not referred to behavioural change in the industry and commerce. They were set with reference to waste-intensive companies. However, they were addressed at waste-extensive ones, since these were where opportunities for improvement lay. Therefore, though industry received the price signal provided by the tax less filtered than households, its behavioural reaction to the tax was in general poor.

For a few very waste-intensive companies (traditional 'chimney-industries') a high tax level could put a significant stamp on financial results (Dengsøe and Andersen, 1999: 72-74). Competitiveness concerns regarding those companies might have hindered increased tax levels (ibidem), the waste tax rates being mainly referred to the impact expected in waste-intensive industries. However, opportunities for improvement (i.e. marginally cheapest waste reductions) were mainly available in waste-extensive industries or within trade and offices that so far had not paid much attention to waste issues displaying relatively low recycling rates (Dengsøe and Andersen, 1999: 72-74). Waste-intensive companies, though with high amounts of waste, were significantly further advanced in their waste management than waste-extensive companies (ibidem).

Therefore, except for selected industries, the tax burden imposed on industry was too low to induce behavioural change. Between 1987 and 1993, industrial and commercial sectors increased their waste production (by 8 per cent) and, except for the construction sector, did not significantly increase recycling (Andersen et al, 1997: 6). Even in 1996 and 1997, in general, the waste tax burden did not exceed 1 per mille of the net turnover or 1% of the operating profit/loss (Dengsøe and Andersen, 1999: 70). This low level was not compensated for by treatment charges for incineration and landfilling (i.e. the price for delivering waste to a treatment facility, excluding waste tax and VAT) since, compared to other European countries, these in Denmark were also low (EEA, 1999). Consequently, waste management has ranged low in the companies' hierarchy of attention (Dengsøe and Andersen, 1999: 70).

4.5.5 The waste tax was raised on polluters able to avoid pollution

Legally the companies liable for the tax payment were registered waste treatment plants, whereas the economic burden following from the tax payment lay ultimately on waste producers. Such economic burden was more accurately transferred to waste polluters not covered by municipal collection schemes, i.e. mainly industry and services, and among these to companies bringing their waste directly to treatment plants. Therefore, though householders were expected to bear the tax via waste collection fees charged by municipalities, the waste tax was aimed primarily at the industry (both waste companies and manufacturing industry), which were the polluters best able to avoid pollution.

The industry has produced the bulk of the waste in Denmark, being responsible for two thirds of the waste production and for the growing total amount of waste produced in the country (Brix, 2010: 1). In the manufacturing industry a great part of the waste increase was due to increasing waste intensities (Brix and Bentzen, 2008: 16), which can be addressed with more efficient management of the production process (e.g. better selection of the material inputs) and technological improvement. A waste tax charged per waste amounts is able to produce both these effects.

Companies could also choose more freely than households the waste management technique to use. When the waste was not covered by municipal waste collection schemes, companies were relatively free to choose between plants and disposal forms until 1993, when the obligation to incinerate waste suitable for incineration was introduced. However they had to comply with assignments in the municipal waste regulation (Andersen et al, 1997: 33). It depended on local conditions whether separate waste fractions were assigned to one or more named plants or whether certain types of plants were assigned (ibidem). Waste able to be used directly by other companies could be delivered directly to them (ibidem). The Guidelines from the DEPA on disposal, planning and registration of waste stated that, according to the Danish Statutory Order on Waste, companies had freedom of choice for recyclable materials (ibidem). Companies could, however, be ordered by regulations to deliver certain waste fractions for recycling (Andersen et al, 1997: 34).

The waste tax was mainly aimed at waste companies by selecting them as taxpayers, by making “more profitable for the waste collection services in each municipality to establish recycling and separation”, since for each tonne of waste delivered for recycling, the waste collection services would save the corresponding tax (Law Proposal 176, 11 February 1986, 4426; Andersen et al, 1997: 33, 90). These were indeed the ones best able to avoid pollution since the entities running most of these companies, namely the municipalities, were the ones able to establish recycling systems, which were paramount for the environmental effectiveness of the tax.

The municipalities controlled the waste service market *de facto* and *de iure*. Local councils played a central role in the waste management system via four variables (Andersen et al, 1997: 7). Firstly, they decided the design of the local disposal structure, i.e. treatment by incineration or landfilling, and waste fractions to be collected separately. Secondly, they decided the relationship between collected and assigned waste in their waste regulation. Thirdly, local councils established a waste company or contracted with such a company, and determined the market for private operators or carriers. Finally, they fixed waste collection fees.

Conclusions

This chapter has analysed the Danish waste tax to assess its degree of inclusion of the design features of environmental taxes and how the latter impacted on the environmental effectiveness of the tax. The conclusion was that the tax performed well and fast in steering behaviours towards pollution prevention, namely through increased recycling, rather than reductions in total waste amounts, in some specific waste streams and among the polluters with the highest tax awareness and possibility of tax avoidance, namely waste companies. Following the low tax level and institutional filters in place, the precise environmental objective of the tax, namely to increase the costs of traditional disposal (i.e. landfill and incineration) in order to make recycling profitable, was only achieved in heavy waste streams for which there were low cost alternatives to traditional disposal and where the financial loss caused by the tax was the highest, namely construction and demolition waste, garden waste and bulky waste. Most improvement in waste management following from the tax depended on municipal voluntary recycling schemes supported by collection and separation schemes set for such waste.

A causal relationship was set between such performance and the level of inclusion of the design features of environmental taxes in the law and institutional practice. Tax awareness and tax avoidance were well established in the tax design, with weight-based charges and tax refunds for waste removed from waste plants for recycling. The tax was raised on a close proxy for polluting emissions. Its tax rate was first referred to pollution abatement costs (i.e. the costs of recycling) and from 1992 onwards also to relative polluting impacts. Though some criticisms can be addressed at the 'Danish waste hierarchy' and the lack of consideration for the composition of the material flow entering the ecosystem, its payment fairly mirrored the evolution of environmental damages and was charged to the polluters best able to avoid pollution by adapting the waste management design mainly by developing recycling schemes. However, the level of the tax rates in some cases was not sufficiently high to induce behavioural change.

The failure to refer the tax rates to environmentally correct *effective waste disposal prices*, both in absolute and relative terms, has reduced the steering capacity of the tax. This capacity was further damaged by the institutional framework where the tax was applied. This has filtered the transference of the referred design features to institutional practice, harming especially the tax awareness, the reward of tax avoidance and the linkage between the tax payment and environmental damages. All these features proved crucial for the environmental effectiveness of the waste tax. Once filtered by the institutional arrangements the tax worked as a fiscal instrument rather than as an instrument of environmental policy except for the referred waste streams.

Following the overlap between the role of the municipalities as indirect taxpayers (via the ownership of most waste companies), *de facto* tax authorities (by deciding how, when and how much the tax payment was transferred to waste producers) and professional players in the waste service market, the design features of environmental taxes were transferred to institutional practice in waste management according to their interests. Municipalities were the ones ultimately deciding on the design of the Danish waste management system and maintained ambiguous interests regarding the reduction of taxable waste amounts.

The *de iure* and *de facto* power of the municipalities, which also explains the subjective incidence of the tax on waste companies, has constrained the capacity of the tax to influence the design of waste management systems on which the environmental tax effectiveness mainly depended. Lack of reference of the tax rate to environmentally correct relative *effective waste disposal prices* allowed municipalities to change the environmental hierarchy of behaviours communicated by the tax. In some municipalities the tax advantage provided for incineration was neutralised by municipal fees. And, following the insufficient level of the tax to compensate for the additional costs of recycling some waste streams, there was a lack of recycling facilities or incentive to use compulsory collection schemes.

The most pernicious influence on the environmental effectiveness of the waste tax came from the billing system and the waste management system as a whole. In industry, institutional filters rooted in the integration of the waste service market hid the price signal provided by the tax. This aggravated further the problem caused by low tax rates, which were set with reference to waste-intensive companies but addressed to waste-extensive ones. The usually low rank waste management occupied in the hierarchy of attention of the companies following the relatively low burden waste disposal costs put on the production costs structure has not been improved following the adoption of the tax except for heavy waste. In households, the charging system used delayed, hid and flattened the price signal provided by the tax, which led to fiscal illusion and reduced availability of tax avoidance strategies, since the linkage between the tax payment and the tax base (and consequently also environmental damages) was cut.

CHAPTER III

THE PORTUGUESE ENERGY TAX

Portuguese legal vocabulary and abbreviations

Diário da República	The Official Journal of Portugal
Portaria	Development legislation
Projecto de Lei	Bill (proposal of new legislation)
Resolução do Conselho de Ministros	Resolution of the Portuguese Council of Ministers

This chapter analyses the energy tax, which was the Portuguese tax where the inclusion of the design features of environmental taxes was expected to be the strongest, and aims at assessing whether indeed this tax included such features and how this has influenced its environmental effectiveness. The conclusion is that, until 2011, there was scarce inclusion of these features in energy tax design and that this kept a causal linkage with the low positive environmental effectiveness of the tax throughout the 1990s.

The interest of this case study also lies in the fact that it allowed assessment of the environmental impact on fossil fuel consumption of an energy tax raised according to the structure set by the Energy Taxation Directive (2003/96/EC). This tax tested the tax design proposed by the Directive especially well, since it followed closely the energy taxation structure set by the Directive for the taxation of fossil fuels and was the only tax raised in Portugal on fossil fuel consumption. No other taxes were charged on fossil fuel consumption, with no taxes raised on specific emissions as was the case in other countries.

The design features of environmental taxes were scarcely included in the Portuguese energy tax on all the parameters considered. Regarding the reference to precise environmental objectives, such objectives were assigned to this tax in 1991, namely to reduce sulphur and lead emissions related to fuel consumption, and again in 2008, to reduce greenhouse gas emissions in industry. A single overarching environmental objective related to the whole economy was absent. This has been understood as primarily having contributed to the failure of the energy tax to lead consumption towards the environmental hierarchy of consumptions with reference to a specific pollutant.

Although the tax law was changed in 2001 to include environmental criteria among those to be followed in the setting of the tax rates, no such criteria seem to have been used in institutional practice. And although, in the law changes that occurred both in 2005 and in 2008, clear reference is made to the need to internalise the CO_2 costs following from energy consumption, it has not been explained in any of the cases how such costs have been calculated in setting the tax rate. Furthermore, in none of the cases referred to have quantitative environmental targets been set and no reference has been made in the tax law to intentions to steer behaviour.

All these features indicate that cost internalisation, rather than pollution prevention, has underpinned the energy tax design since 1990. This understanding seems to be confirmed by the relevance given in the law to the principle of equivalence. The finding that the energy tax failed to communicate the environmental hierarchy of consumption with reference to any precise pollutant is coherent with this rationale, as is the very low institution of tax awareness and tax avoidance in the energy tax design. With regard to its tax base and relative tax rates, the energy tax failed to signal a correct environmental hierarchy of consumption since its payments were unable to accurately mirror the evolution of environmental damage.

The Portuguese energy tax base, though keeping a direct linkage to polluting impacts, was not specific to polluting emissions or a proxy for these, but referred to a measured unit of fuels. Each measured unit of each fuel contains several pollutants causing different environmental impacts. Therefore, the tax was unable to target a negative price signal to any specific pollutant. Furthermore, since the tax base was not the energy content of fuels, the energy tax was also unable to financially burden inefficient energy use. Furthermore, the narrow definition of the energy tax base did not follow environmental criteria.

The energy tax rates made no reference to pollution abatement costs or to the relative polluting impacts of specific pollutants. The structure of the energy tax rates did not take into consideration the environmental impact of different energy products with regard to any single specific pollutant, or to their energy content. In the transport sector, the tax rates signalled simultaneously several hierarchies of consumption, not all of these environmentally correct. These rates signalled at times the lead content of fuels and at other times their sulphur content. Regarding the CO_2 fuel content, the energy tax rates signalled as positive a higher content of this pollutant in motor fuels since diesel was taxed at a lower rate than gasoline. The energy tax rate did not relate to environmentally correct *effective pollution prices* in either relative terms or absolute terms. Since until 2005 the energy tax was been used as an energy price stabiliser and tax rates were not adjusted yearly on the basis of the Consumer Price Index, the tax allowed real energy prices to decrease.

In the tax rate differentiation according to sulphur and lead content in fuels, the insufficient intensity of the regulatory intervention, which meant that the price differential was unable to boost demand for cleaner fuel, also hampered the environmental effectiveness of the energy tax. This problem was aggravated by uncompetitive market structures which proved the relevance of competitive markets for environmental tax effectiveness. These tax differentiations have also shown the importance of relating the tax rate to environmentally correct and effective pollution prices, rather than just considering partial components of the consumer price, such as the tax burden, as well as of transmitting a price signal to those able to avoid pollution.

Regarding subjective tax incidence, the energy tax was raised on the best payers, i.e. households, rather than on those best able to avoid pollution, since the bulk of its burden was laid on motor fuels. Industry has always benefited from more favourable regimes assigned by sector. The tax regime has included several more favourable clauses based on consumption amounts that sheltered major energy consumers from its price signals. Although the law was changed in 2008 to also cover energy-intensive industries, the broad definition of an energy-intensive industry, and the possibility these industries continued to enjoy of receiving energy tax exemptions following energy efficiency agreements assigned by sector, resulted in no substantial change to the subjective incidence of the energy tax.

The energy tax was designed as a common excise tax and has always lain within the responsibility of the Ministry of Finance. Following the failure to include in a relevant way the design features of environmental taxes, the energy tax placed a blunt price on polluting behaviour (i.e. energy consumption), classifying this as 'environmentally-related' rather than 'environmental', as it was sometimes considered (e.g. OECD, 1999; EEA, 2000a: 37). In both the long and the short term, there is no evidence that the tax played an important role in changing consumption patterns and consequently in producing positive environmental change. Even after the rephrasing of the law in 2001, with reference now made to the environment among the criteria to be considered in setting the tax rate structure, this tax kept its tradition of being mainly an important source of tax revenues for the government. In that sense it was aimed at cost internalisation rather than at steering behaviour.

1. THE STRATEGIC IMPORTANCE OF THE ENERGY TAX FOR ENVIRONMENTAL POLICY IN PORTUGAL

By the beginning of the 1990s, data on national energy consumption presented an inefficient and polluting pattern and air emissions were an important source of national environmental problems. Furthermore, air pollution was among the main causes of public concern (MARN, 1995: 166, Ferreira de Almeida, 2000: 17-18). In 2001, the first national air quality targets were approved following the adoption of the first National Climate Change Programme (PNAC 2001). Before this, following the ratification of the Kyoto Protocol and the European Union Burden Sharing Agreement in 1998 and 2002 respectively, Portugal had only agreed in international and EU fora to restrain the increase of its greenhouse gas emissions to a maximum level of 27%, taking as reference 1990 values.

Traditionally, Portuguese energy policy was mainly supply-side managed. Energy was regarded as a basic consumer right for the supply of which the government was responsible and expected to ensure a fair price through regulation and tax instruments. National pricing rules for natural resources and the environment were predominantly based on income redistribution objectives, rather than on economic efficiency (Santos *et al*, 1999: 456-457). Environmental protection was included for the first time in national energy policy in 2003, when it was considered the third strategic goal of this policy, the first being overall competitiveness and the second security of supply (Cabinet Resolution 169/2005, 24 October 2005; 2004 National Climate Change Programme (Resolution of the Portuguese Council of Ministers, 119/2004, 31 July 2004, 2003)).

Throughout the 1990s, no tax was raised on air polluting emissions or industrial energy consumption (OECD, 1999; EEA, 2000a: 70). And, apart from Value Added Tax (VAT), the energy tax was the only tax raised on energy consumption in Portugal until 2011. Furthermore, until the introduction of the EU Emission Trading Scheme in 2005, it was, together with the maximum market prices for energy, centrally decided, the only policy instrument directly influencing price signals provided by the government with regard to national energy consumption.

Despite the reduced number of energy-related taxes, in the early 1990s the high tax component of fuel prices (energy tax plus VAT) explained why environmentally-related taxes were making the highest contribution to total tax revenues (10%) in Portugal compared to all the other EU Member States (EEA, 2000a: 24). According to the OECD/EEA database on environmentally-related taxes[35], Portugal was among the countries presenting the highest percentage of total tax revenues coming from energy-related taxes, whereas pollution taxes unrelated to energy consumption were not significant (OECD, 1999; EEA, 2000a: 70).

Portuguese environmentally-related taxes were mainly targeted at the transport sector and included directly related taxes (on vehicle acquisition and circulation) (3%) and the energy tax (7%) (OECD, 1999; EEA, 2000a: 37). These taxes were 'first generation' pollution taxes, i.e. they were not created for environmental reasons and did not have revenues earmarked for environmental purposes, but were raised on tax bases with a close relationship to environmental damage.

The relatively heavy tax burden imposed on motor fuel consumption was explained by the administrative simplicity and profitability of indirect taxes in a country with low income levels and a highly inefficient tax administration that competed for foreign investment on the basis of low wages and low company taxes. It was publicly discussed whether, under the cloak of environmental concerns, the government was disguising revenue motives (Plenary Session of the Portuguese Parliament, 1999). It was also questioned whether the environmental effectiveness often attributed to these taxes by environmental economics literature was being hindered by an incoherent tax design (OECD, 2001c; Bronchi and Gomes-Santos, 2001: 21-22).

2. THE ENERGY TAX DESIGN

The energy tax, initially named 'tax on oil products' (*Imposto sobre os produtos petrolíferos*) was then renamed 'tax on oil and energy products' (*Imposto sobre os produtos petrolíferos e energéticos*), having in mind the taxation of electricity following the Energy Taxation Directive (2003/96/EC). This was introduced in 1986 (Law 9/86, 30 April 1986; DL 292/87, 30 July 1987).[36] It replaced the previous administratively set regime of price differentials and the tax regime aimed at the national consumption of fossil fuels introduced in 1982, when the process leading to Portugal joining the (then) European Economic Community (EEC) was initiated (Law 40/81, 31 December 1981; DL 133/82, 23 April). Until then fossil fuels were taxed only when imported.

[35] Available at www.oecd.org/env/tax-database.

[36] This regime was changed by DL 261-A/91, 25 July 1991, and again, following Directive 92/91, by DL 52/93, 26 February 1993, which set the general regime of the excise taxes, and again by DL 123/94, 18 May 1994, which regulated the tax base and exemptions, following Directive 92/81, both aspects within the competence of Parliament (and the Ministry of Finance following its authorization), and DL 124/94, 18 May 1994, which regulated the structure of tax rates, following Directive 92/82, which was a matter for the Ministry of Finance and the Ministry of Industry to decide following the authorization of Parliament. DL 123/94 was again changed for the first time in 1995 and the matters it regulated were introduced in the first version of the Portuguese Excise Duties Code (Código dos Impostos Especiais de Consumo, DL 566/99, 22 December), whereas DL 124/94 was changed yearly by the national budget law. Coloration and marking of oil were introduced by Portaria 68/94, 31 January 1994, and Portaria 157/96, 16 May 1996, as required by Directive 95/60, and changed by Portaria 93/97, 7 February 1997.

2.1 References to environmental concerns in the energy tax design were mainly associated with cost internalisation

In 1991, energy tax rates were differentiated according to sulphur content and lead content, aimed at raising the market share of cleaner fuel oil and gasoline. The law did not mention environmental concerns to explain this change. In 2006, the National Climate Change Programme (PNAC) included in the measures to be used to reduce national greenhouse gas emissions an increase in the energy tax on industry (Resolution of the Portuguese Council of Ministers 104/2006, 23 August 2006). In 2008, the law was changed to apply an energy tax to some energy-intensive industry and to start a gradual increase in the maximum tax rates applied to industrial fossil fuels (Art. 64 Law 67-A/2007, 31 December).

In both cases, no quantitative targets were fixed and only the means were specified, namely tax rate differentiation in the first case, and an increase in the maximum tax rates applied to fuels used in industry in the second case. The criteria to be used in the calculation of the new rates were only provided in the second case, namely the costs of the extra CO_2 emissions following from the consumption of coal, oil and fuel oil when compared to natural gas, though no specification was made as to how such costs were to be calculated (Explanatory Note DL 71/2008, 15 April 2008).

The law mentioned the principle of equivalence as the means of addressing environmental concerns in energy taxation, in the sense that the energy tax should transfer to consumers the costs they placed on society by their consumption (Art. 2 Excise Duties Code (Código dos Impostos Especiais de Consumo), hereafter also IEC Code, DL 73/2010, 21 June 2010). No equivalent reference was made in tax law to steering behaviour. The development legislation introducing specific measures, namely the law of 2005 changing the tax rates applied to gasoline and diesel (Explanatory Note Portaria 510/2005, 9 June 2005, and those following) and the law introducing the tax rates applied to coal, oil and fuel oil used in energy-intensive sectors that had previously been exempted from taxation (Explanatory Note Portaria 1530/2008, 29 December 2008), refers mainly to cost internalisation (regarding costs caused by CO_2 emissions). Only in the 2008 change is reference also made to behavioural change, namely the desire to induce the reduction of greenhouse gas emissions by industry (Portaria 1530/2008, 29 December 2008). In the law introducing the rate differentiation for sulphur and lead, no similar references can be found (DL 261-A/91, 25 July 1991).

2.2 The energy tax was raised on measured units of energy

The energy tax was an excise duty raised on fossil fuels and energy products used as motor fuel or heating fuel in a specific amount per weight or volume unit (Arts. 88 to 91 IEC Code). The tax was never raised on electricity, so Portugal has been failing to fulfil its obligations under the Energy Taxation Directive (2003/96/EC) since January 2010 (Art. 18 (7) Energy Taxation Directive). And the new tax base introduced in 2011 did not cover heavily CO_2-loaded peat (Art. 88 (1) (c) IEC Code).

Sometimes exemptions depended on use. Fossil fuels and energy products used in commercial aviation and water navigation were exempted (Art. 89 (1) (b) and (c) IEC Code). Fuels used in public transportation vehicles were also exempted, regardless of their type (Art. 89 (1) (e) IEC Code), as well as coal, oil and fuel oil with a sulphur content not higher than 1% used in the sectors covered by the EU Emission Trading Scheme (hereafter also ETS), including power plants and co-generation units, and those entering energy efficiency agreements (for which refineries were eligible, DL 71/2008, 15 April 2008) (Arts. 88 (6), 89 (1) (d) and (f) and 96 IEC Code).

Traditionally, fuels consumed in energy-intensive sectors have been granted exemptions. Their coverage since 2005, following the changes to accommodate the regime set by the Energy Taxation Directive (Art. 33 Law 55-B/2004, 30 December 2004), led to exemptions being granted directly to particular sectors. This was the case with coal (Art. 88 (1) (c) IEC Code) and natural gas. The latter fuel, the introduction of which to the Portuguese market in 1997 was underpinned by industrial policy objectives, became subject to taxation only in 2007, since its final energy consumption in 2000 was still less than 15% (Art. 33 Law 55-B/2004, 30 December 2004; Art. 92 (4) and previous Art. 70 (1) (c) IEC Code). Traditionally, fossil fuels and energy products used in the same plant where they had been produced were also granted exemptions (Art. 88 (6) and previous Art. 70 (3) IEC Code).

From January 2011 onwards, bio-fuels (including biomass) up to a maximum amount per year produced by small dedicated plants could also be totally or partially exempted, depending on their market price or the market price of their raw materials or the fuels they replaced (Art. 90 IEC Code, according to Law 55-A/2010, 31 December). This condition was aimed at avoiding overcompensation for their production costs. In 2006 bio-fuel added to diesel or gasoline was totally or partially exempted from the energy tax (Decree-Law 66/2006, 22 March 2006). The amount of subsidised bio-fuel was set according to growing market quotas over three periods (2% in 2006, 3% in 2007 and 5.75% between 2008 and 2010). Subsidies were assigned to producers on a bid basis, following an obligatory incorporation regime that forced their purchase by refineries at a premium price.

2.3 The energy tax rate structure related simultaneously to several criteria

Energy tax rates varied according to the minimum tax rates set by EU law (Directive 97/74/EC, OJ L 10/22, 16 January 1998, Directive 92/82/EEC, OJ L 365/46, 31 December 1994; and Energy Taxation Directive (2003/96/EC), OJ L 283/51, 31 October 2003). Except for the tax rate differentiations according to sulphur and lead content in fuels, which were introduced in 1991, energy tax rates have not been based on environmental criteria, and until 2005 were used to stabilise energy market prices. Though consideration for environmental criteria in the setting of tax rates has been mentioned since 2001 as a gradual process already started in 2005, in 2011 the tax rate structure still showed no evidence of the environmental hierarchy of consumption with reference to any specific pollutant.

Reduced rates were usually set according to fuel use. However, some products benefited from more favourable regimes regardless of their use. This was the case of diesel, which was always taxed at a lower rate than gasoline, and, since 2009, of fossil fuel produced from used waste oil or waste. These were taxed at the rate applied to their raw materials, regardless of their compliance with the waste management hierarchy (Art. 92 (5) IEC Code, according to Law 64-A/2008, 31 December).

LPG has been taxed at the minimum tax rate allowed by EU rules, due to its better environmental performance when compared with gasoline (LPG-powered cars emit 30% less emissions than state-of-the-art petrol-powered cars, Speck and Ekins, 2002: 95) (Art. 92 (3) IEC Code). Until 2007, bio-fuel produced or consumed within a research and development project aimed at improving environmental performance of fuels was eligible for an 80% reduction in the energy tax rate (previous Art. 70 (1) (j) IEC Code, revoked by Law 53-A/2006, 29 December 2006). But until 2000 no project had benefited from this possibility (GTAPFA, 2000: 175).

The tax rate structure kept tax rates floating within a previously defined range

Until 2000, most tax rates (general tax rates) were changed yearly by the national budget law (Art. 2 DL 124/94, 18 May 1994). This only provided minimum and maximum values for tax rates, depending on whether the tax was charged for the mainland or the islands, where a lower tax rate applied, with the exact amount of the tax rate being set by joint development legislation (Portaria) of the Minister of Finance and the Minister of Industry (Art. 49 Law 3-B, 4 April 2000). Some special tax rates, such as those applied to natural gas, fossil fuels produced from used waste oil or waste, were not changed yearly, being set by Article 92 (previous 73) of the IEC Code.

From 2000 onwards, the table of minimum and maximum rates was set by the IEC Code (Article 92 previous Art. 73 IEC Code, DL 566/99, 22 December 1999). These rates differentiated between lead content in gasoline, marked and non-marked oil and diesel, which depended on use, and sulphur content. Unleaded gasoline could be taxed at a lower rate than leaded gasoline, but always at a higher rate than diesel and oil. Fuel oil was taxed at a significantly lower rate than gasoline, oil or diesel, with the maximum rate applicable to fuel oil with more than 1% sulphur content being slightly higher than that applicable to fuel oil with less than 1% sulphur content. In January 2011, these limits were the ones referred to in Table 3.1 (Law 55-A/2010, 31 December 2010). Marked oil was used for heating and marked diesel was used mainly in agriculture, for small boats and on the railways, as well as for domestic and business heating (Art. 93 IEC Code). Both these fuels enjoyed reduced tax rates.

Table 3.1 Energy tax minimum and maximum rates in January 2011 (Article 92 IEC Code)

Product (per 1000 litres at 15º degrees Celsius, Art. 91 (1) IEC Code)	Minimum tax rate (Euros)	Maximum tax rate (Euros)
Gasoline with lead content	650	650
Gasoline without lead content	359	650
Oil	302	339.18
Marked oil	0	149.64
Diesel	278	400
Marked diesel	21	199.52
Fuel oil with more than 1% sulphur content	15	34.92
Fuel oil with less than 1% sulphur content	15	29.93

The use of only minimum and maximum rates in the IEC Code is related to the system of maximum market prices applied to 95 octane diesel and unleaded gasoline until 2004 (Art. 77 IEC Code, revoked by Art. 3 of DL 162/2004, 3 July, Portaria 224-A/96, 24 June 1996; Portaria 759-A/96, 26 December 1996). The maximum market price applied to each product followed from the European energy price before taxes (set by Portaria taking into account the considered average cost of the fuel in Member States), plus a correction factor, plus energy tax and VAT. This system aimed at keeping market prices fairly stable following changes in the international market prices for oil. Changes in the latter were compensated for by changes in the energy tax rate. From 2005 onwards, all tax rates were corrected for inflation (Art. 34 Law 55-B/2004, 30 December 2004).

In 2011, and since 2004, an additional levy to the energy tax was raised on the amount that resulted from the application of the fuel tax rate to its tax base with regard to gasoline (Euros 0.5 cents/litre) and road diesel as well as coloured and marked diesel (Euros 0.25 cents/litre) (introduced by Art. 38 (5) Law 107-B/2003, 31 December 2003, and restated every year since by the national budget law). Revenues collected have been earmarked for the Permanent Reforestation Fund (Art. 4 (a) DL 63/2004, 22 March), with a top limit of Euros 30 million per year. In 2008, tax rates applied to gasoline and (Euros 0.086/litre) road diesel were lowered to compensate for the extra tax charged on these fuels (Euros 0.064/litre and Euros 0.086/litre respectively) to finance road works (Portaria 16-C/2008, 9 January, Art. 4 Law 55/2007, 31 August 2007).

Since 2001 environmental considerations have been progressively mentioned in the law

Since 2001, environmental considerations were added to the criteria to be observed in setting the tax rate ("gradually the less polluting fuels should be favoured within the breaks provided") (Art. 40 (6) Law 30-C/2000, 29 December 2000; Art. 92 (1) IEC Code), following a proposal by the Green Party (Grupo Parlamentar 'Os Verdes', 2000). It was then established that tax rates applied to fuels should consider their polluting effect, defining a minimum and a maximum rate for each fuel. However, though both in 2005 and in 2008 tax rate increases referred to the need to account for costs caused by CO_2 emissions, the criteria used to calculate the exact tax rate levels adopted have not been explained (Luís Barata, Portuguese Ministry of Finance, Excise Duty Department, interview, 4 May 2011).

From 2005 onwards, the reference of energy tax rates to environmental criteria was used to explain the gradual narrowing of the difference between the tax burden applied to gasoline and diesel, as well as to road diesel and heating diesel (Art. 34 Law 55-B/2004, 30 December 2004). Tax rates applied to heating diesel[37] and to road diesel[38] were increased in order to bring these gradually to the level of those applied

[37] Until June 2011, Portarias 211/2007, 22 February 2007, 16-C/2008, 9 January 2008, 653/2010, 11 August, and 99/2011, 11 March 2011.
[38] Until June 2011, Portarias 75-A/2006, 18 January 2006, 30-A/2007, 5 June 2007, 16-C/2008, 9 January 2008.

to road diesel and gasoline respectively, aiming to apply the same rate to all fuels by 2014 (Art. 34 Law 55-B/2004, 30 December 2004). However, in March 2011, road diesel (Euros 278.41, Portaria 16-C/2008) and heating diesel (Euros 251.48, Portaria 99/2011) were still taxed at a lower rate than gasoline (unleaded gasoline Euros 518.95, Portaria 16-C/2008; leaded gasoline Euros 620, Portaria 30-A/2007) and road diesel (Euros 278.41, Portaria 16-C/2008) respectively.

In 2008, following the 2006 National Climate Change Programme (Resolution of the Portuguese Council of Ministers 104/2006, 23 August 2006), it was decided to increase the energy tax rates applied to coal, oil and fuel oil with more than 1% sulphur with reference to their extra CO_2 load vis-à-vis natural gas (Art. 64 Law 67-A/2007, 31 December 2007). The maximum energy tax rates applied to the products concerned were due to gradually increase in order to internalise the costs and induce replacement of the previously used fuels by less CO_2 loaded natural gas (Portaria 1530/2008, 29 December 2008).

Starting from March 2009, the tax rates applied to these instances of consumption were first set according to the minimum rates allowed by the IEC Code (keeping the levels introduced in 2005, Portaria 510/2005, 9 June 2005). In 2011, the tax rate applied to coal and oil products was Euro 4.6 and to fuel oil with more than 1% of sulphur it was Euro 29.25 (Portaria 1530/2008, 29 December 2008), whereas natural gas was taxed at Euro 2.78/gigajoule (Art. 92 (4) IEC Code). The higher tax rates were to apply only to energy-intensive sectors not covered by the EU ETS and not party to energy efficiency agreements following the regime of exemptions applied to energy-intensive industries.

Tax rate differentiation according to sulphur content of fuel oil

An economic incentive to promote low sulphur content (hereafter also LSC) fuel oil was created in order to reduce SO_2 emissions. This was implemented as a reduction in the energy tax rate on fuel oil with less than 1% sulphur. The objective was to stimulate an expansion of the market share of the cleaner fuel.

Between 1991 and 1996, high sulphur content (hereafter also HSC) fuel oil was subject to a maximum market price system (hereafter also MMP), whilst LSC fuel oil had a free price regime (Art. 7 (1) DL 261-A/91, 25 July 1991). During this period, the energy tax on HSC fuel oil was calculated indirectly from the fixed MMP and could range within an interval of PTE 1-10/kg (Art. 7 (1) DL 261-A/91, 25 July 1991). The energy tax for LSC fuel oil during the same period had a reduction of PTE 3/kg (Art. 7 (9) DL 261-A/91, 25 July 1991). The tax differential was kept after July 1994 via a fixed tax rate set at PTE 2.5/kg for LSC fuel oil and PTE 5.5/kg for HSC fuel oil (Arts. 6 and 7 Portaria 326-A/94, 27 May 1994). These values were maintained when the MMP was abolished in 1996 (Santos *et al*, 1999: 457-461).

The Energy Taxation Directive allowed Portugal to keep on applying a reduction in the rate of excise duty on heavy fuel oil in order to encourage the use of more environmentally friendly fuels, as long as this reduction was specifically linked to sulphur content and the rate of duty charged on heavy fuel oil corresponded to the minimum rate of duty on heavy fuel oil as provided for in the Directive (Annex I, Point 12 Energy Taxation Directive (2003/96/EC)). In 2011, the minimum tax rate was the same for both fuels (Euros 15), but the maximum rate applied to fuel oil with less than 1% sulphur content was Euros 29.93, whereas for fuel oil with more than 1% sulphur content it was Euros 34.92.

Tax rate differentiation according to lead content of gasoline

Unleaded gasoline was introduced in the Portuguese market in 1989. From January 1991, a rate differentiation was applied in the energy tax according to the lead content of gasoline (Art. 7 (6) DL 261-A/91, 25 July 1991). Between 1989 and 1991 both leaded and unleaded gasoline were under a maximum market price (MMP) regime. Environmental arguments for the reduction of lead emissions were introduced in 1991 to explain the elimination of the MMP regime for unleaded gasoline. The MMP regime continued for leaded gasoline and was used as a reference to set the tax rate differentiation (Art. 7 (1) DL 261-A/91, 25 July 1991). In 1996 unleaded gasoline was reintroduced in the MMP regime (Law 10-B/96, 23 March 1996, Art. 2 Portaria n. 224-A/96, 24 June 1996, revoked by Portaria 1226-A/2001, 24 October 2001).

Unleaded gasoline started to be granted a deduction of PTE 12/liter in the energy tax relative to the value charged for leaded gasoline. The energy tax on leaded gasoline was set indirectly, according to a formula, after the MMP had been calculated and within an interval of PTE 77-109/liter (Art. 7 (4) DL 261-A/91, 25 July 1991). The exact amount of the tax rates was regularly (multiple times per year) updated by legislation (Portaria), following the ranges introduced by decree-laws. For example, in January 1994, the energy tax for leaded gasoline was set in the range of PTE 77-99/liter and in the lower range of PTE 71-93/liter for unleaded gasoline (Art. 2 DL 124/94, 18 May 1994), fixed at PTE 91/liter and PT 84/liter in May (Art. 2 Portaria 326-A/94, 27 May 1994).

The turning point on the tax incentive for unleaded gasoline was reached three years later. In 1997, the fiscal incentive for unleaded gasoline was reduced to PTE 7/liter with the argument that its consumption was approaching the EU average (Santos et al, 1999: 462). In June of the same year the tax differential was reduced further (PTE 6.3/liter) (Arts. 1 and 2 Portaria 224-B/96, 24 July 1996). In December, the energy tax rate for both gasolines was increased in order to recover the tax revenues lost through the tax benefit (VAT and energy tax) given to diesel, and the differential was increased to PTE 6.8/liter (Arts. 1 and 2 Portaria 769-A/96, 30 December 1996).

2.4 The energy tax was not charged on energy-intensive industries

The energy tax was charged to subjects running a plant producing, using or storing fossil fuels or energy products and transferred to final consumers (for example, in the case of motor fuels these were consumers at the pump) (Arts. 4, 7 and 8 IEC Code). Energy-intensive industries have traditionally benefited from more favourable treatment by the energy tax, through exemptions granted to fuels (for example, fossil fuels and energy products used in the same plant where they have been produced, coal until 2005, natural gas until 2008) or to the sectors themselves.

Since 2008, and following the objective to reduce CO_2 emissions from industry set by the 2006 National Climate Change Programme, the exemptions previously granted automatically to sectors with high energy consumption (DL 58/82, 26 February 1982) only remained for sectors covered by the EU ETS or party to energy efficiency agreements (Art. 11 (1) DL 71/2008, 15 April; Art. 89 (1) (f) IEC Code, Art. 64, Law 67-A/2007, 31 December 2007). Energy-intensive sectors, the definition of which was broadened to cover all sectors consuming at least 500 tep/year, including freight transport operators (Art.2 (1), DL 71/2008, 15 April 2008), and not covered by the EU ETS, were eligible for exemptions after joining energy agreements which took into account not only energy efficiency but also CO_2 emissions (Arts. 7 (2) and 11 (1) DL 71/2008, 15 April 2008).

3. Evaluation and Critical Analysis of the Energy Tax

The Portuguese energy tax scarcely included the design features of environmental taxes and this is considered to have led to its low environmental effectiveness. Positive environmental effects following from the energy tax were weak and slow. Throughout the 1990s, despite the high share of tax revenues raised on energy consumption, the tax seems to have played a minor role in bringing national energy consumption towards more sustainable patterns, both in terms of energy efficiency and use of cleaner fuels. For failing to provide environmentally correct price signals, both in relative terms and in absolute terms, to economic agents who still had not explored all their opportunities for improvement, the energy tax did not induce the adoption of efficient prevention and abatement measures with regard to pollution following from energy consumption.

In both the short and the long term, energy tax payments failed to accurately communicate the environmental hierarchy of consumption, leading fuel demand towards CO_2-polluting diesel. The tax differentiations according to environmental criteria, namely regarding sulphur content in fuel oil and lead content in gasoline, also provided a weak incentive for a switch to cleaner fuels. In the short term, when the environmental effectiveness of an energy tax regarding fuel-switch is constrained by technology, the Portuguese tax also failed to provide a price signal strong enough to shift private vehicle use to public transportation. In the long term, when fuel price elasticities are higher, general energy taxes might have some positive environmental impact contributing to a shift in car demand towards smaller and lighter vehicles that consume less fuel. However, it is not evident that the Portuguese tax was able to induce such an effect.

The improvement experienced in the country regarding the use of renewable energy sources was in great part led by industrial policy objectives that overlapped with environmental objectives, since the country is well endowed with energy sources of this type (Resolutions of the Portuguese Council of Ministers 169/2005 (24 October 2005), 104/2006 (23 August 2006), and 80/2008 (17 April 2008)). This policy has been led by central government decisions and public incentives provided since 1998 through subsidies (feed-in-tariffs) to producers of energy based on renewable sources (Reiche et al, 2004: 847). Regarding energy efficiency in industry, the same kind of process was initiated in 1982 (DL 58/82, 26 February 1982, changed by DL 71/2008, 15 April 2008). In the transport sector a similar pattern of development was initiated in 2010 with the adoption of the 'Mobi-e' programme, aimed at raising the number of electric vehicles in the country and thus shifting 10% of fossil fuel consumption in the transport sector to electricity by 2020, which is expected to amount to a 2% reduction in national consumption of fossil fuels (Measure 3 of the 2010 National Programme for Energy, Resolution of the Portuguese Parliament 33/2010, 15 April 2010).

Since empirical studies on evaluation and impact assessment of the Portuguese energy tax are scarce, and mainly focus on the 1990s, conclusions on its impact on the environment have to mainly refer to the tax differentiation according to sulphur and lead content in fuel. Furthermore, the analysis needs to rely to a certain extent on the data available on national energy and environmental performance. However, since the latter are influenced by several variables other than the energy tax, caution is required in the analysis. These data only allow us to argue that the energy tax was unable to cut the trend in national energy consumption experienced throughout the 1990s towards less efficient and more polluting energy uses when compared to other EU Member States.

3.1 Price-elasticity of energy consumption is relatively low and is lower in the transport sector than in industry

Energy consumption demand follows from demand functions that depend primarily on income and prices, especially energy prices ,but also all other prices (Sterner, 2010: 2, on fuels).The effect of energy prices and other factors is usually expressed through elasticities. These show the percentage change in energy consumption that would result from a one percent change in the exogenous variables. An energy tax raises the price of an energy product. Therefore, the potential impact of an energy tax on energy consumption must be assessed with reference to price elasticities.

Own-price elasticities can be inelastic or elastic and the range of each type differs by region and system (Fan and Hyndman, 2011: 3709). For a commodity, the range of inelasticity is usually between the absolute values of 0 and 1, beginning the elastic range with values greater than 1 (ibidem). Thus, price-inelastic energy demand means a less than proportional change in demand for a given change in price, whereas in the elastic range consumer demand responds with a greater than proportional change to a given fuel price change.

Price elasticities of energy consumption vary significantly from one sector to another following technological constraints. Whereas oil demand for transport and coal used as input in the industry sector are assumed to be price-inelastic relative to other demand components (for example, when compared to coal demand in power generation), gas consumption in industry and power generation is assumed to be more price-elastic than household gas consumption (Holismark, 2002: 210).

There are many studies of fuel demand elasticities, particularly for gasoline (see Sterner, 2010: 2, for a literature review). Though different estimates are provided, there is a consensus that the long-term price-elasticity of gasoline goes from (-0.6) to (-0.8), while the corresponding income elasticity is around unity, whereas short-term elasticities tend to be around less than a third of long-term elasticities (between (-0.2) and (-0.3), Sterner, 2007: 3196; Dahl, 1982: 373; Sterner, 2010: 2). Therefore, fuel consumption is relatively inelastic in the short term and has a low elasticity in the long term. However, according to some environmental economics literature (e.g., Dahl and Sterner, 1991: 210; Sterner, 2007: 3194), a prolonged high level of energy and vehicle taxation is expected to produce environmental results in terms of energy efficiency, curtailing energy demand.

Opportunities to improve energy efficiency in industry present a large variation among sectors and in general have been continuously used since the early 1970s, in the sense that few 'low hanging fruit' seem to be still available worldwide (Pieprzyk and Kortlüke, 2010: 3). However, in general industries seem to have more alternatives for changing their pattern of energy consumption than the transport sector. The price elasticity of energy consumption in a specific industry will depend on how much the easiest measures to reduce consumption have already been installed in that sector. For example, long-running price elasticities of electricity demand in industry have been reported at (-0.38) in Australia (NIER, 2007), whereas in some energy-intensive European sectors they have been reported as higher. For example, domestic demand price elasticities in the paper and paperboard sector were reported in the range of (-0.7) to (-0.3), in the ceramic tiles and flags sector between (-3.8) and (-3.1) and in the basic iron and steel and ferro-alloys sector between (-8.2) and (-3.5) (Cambridge Econometrics and Ecorys, 2009: 5). National industries presenting high energy intensity compared to their counterparts operating in other similar countries might show evidence of opportunity for improvement still available.

Therefore, energy taxes are expected to have a stronger impact on industry (especially energy-intensive industries, such as the power sector) than on the transport sector. This impact is likely to be stronger in the long term than in the short term. And the impact of energy taxes on the environment is likely to follow mainly from the consumption shift towards cleaner energy sources. Following the tax charge, relative prices of substitute goods also change, stimulating the substitution away from the taxed energy sources. Given the income elasticity of fuel demand, unchecked or poorly regulated motor fuel prices in economies experiencing growing income levels, like the Portuguese economy throughout the 1990s, cause especial concern with regard to the environmental impact expected from increases in energy consumption.

3.2 Polluting and inefficient energy consumption pattern throughout the 1990s in Portugal

Throughout the period 1990-2005, some of the most relevant environmental problems in Portugal were caused by polluting emissions produced in energy consumption. The national pattern of energy consumption was inefficient and focused on fossil fuel. The transport sector and industry were highly dependant on fossil fuels and the latter showed a high energy intensity following incapacity to decouple economic growth from energy consumption. Also, throughout the same period, energy consumption in the transport sector and electricity demand have steadily increased.

Compared with most European OECD Member countries Portugal had by the beginning of the 1990s low atmospheric emissions in terms of population, but was close to the European average in terms of GDP, and rates of growth in emission levels were generally higher than the average (OECD, 2001d). By the end of the 1990s, Portugal had not yet decoupled its air-pollutant emissions from economic growth, and car traffic and related emissions (namely CO_2 emissions) had increased by rates higher than those of GDP (OECD, 2001d). Throughout the same period, little overall progress had been made in improving energy efficiency (OECD, 2001d).

Throughout the 1990s, greenhouse gas emissions in Portugal were caused in great part by road traffic (OECD, 1993: 88, and EEA, 2002b: 1). By the end of the 1990s, emissions growth placed the country well above its linear Kyoto target paths, requiring a significant effort to meet its targets (COM (2001) 708 final, 30 November 2001, 26). Portugal was also unable to reduce its nitrogen oxide (NO_x) and non-methane volatile organic compound (NMVOC) emissions, the main source of which was the road transport sector. It was the EU member state experiencing the highest increase (19%) of this type of emission during the period 1980-1998 (Eurostat, 2001).

During the 1980s, energy requirements and intensity in Portugal increased by one of the highest rates among not only European countries, but also OECD countries (OECD, 2001d). Between 1985 and 1998, the growth in energy sales in Portugal (+139%) was well above the EU average (+45%) and energy consumption increased dramatically (+85%) (Eurostat, 2001). In the period 1990-2002, primary and final energy consumption increased by 51.1% and 57.1%, which represented 4.3% and 4.8% per year on average (Adene, 2004: 9-10).

Primary energy intensity increased by 18.5% from 1990 to 2002. This ratio between primary energy consumption and GDP allows the evaluation of the economy as a whole, considering the transformation as well as the use of energy (ADENE, 2004: 34). Final energy consumption, measured by the ratio between final energy consumption and GDP, which restricts the analysis to the use of energy, grew by

25.5% along the same period (ADENE, 2004: 34). This shows that increases in energy inefficiency were higher in final energy consumption.

By the end of the 1990s, Portugal was the EU member state with the lowest energy efficiency (Table 3.2). Unlike most other OECD member countries, which had continued to reduce the linkage between growth and energy use, in Portugal economic growth became increasingly dependent on access to energy supplies (OECD, 1993: 88, 129). In 1991, oil-related energy sources represented 71% of total energy consumption (MARN, 1995: 215). This polluting pattern of energy consumption was unchanged by the end of the 1990s, leading to the importing of more than 80% of the energy consumed (Grupo parlamentar do BE, 2000: 21). In 2002, oil consumption amounted to a share of 62% in primary energy consumption and 59% in final energy consumption, as shown in Tables 3.3 and 3.4 (Adene, 2004: 9-10).

Table 3.2 Energy-intensiveness of national economies 1991-2000 (%)

	1991	*1992*	*1993*	*1994*	*1995*	*1996*	*1997*	*1998*	*1999*	*2000*
PT	216.67	228.96	229.2	234.54	237.33	228.61	233.21	239.15	247.44	241.51
EU25	X	x	239.89	231.34	230.39	234.98	227.58	224.16	214.94	208.76
EU15	215.42	211.33	211.85	206.1	205.38	209.35	202.71	201.03	195.69	190.53
DK	156.4	148.65	153.71	151.05	146.94	161.67	146.49	140.67	132.14	125.03
ES	221.43	221.71	215.16	223.3	228.71	220.25	222.71	223.59	227.01	227.03
SK	X	1426.59	1289.74	1195.12	1155.43	1051,81	1055,78	997.62	976.45	955.9

Source: Economic Research and Forecasting Department, Ministry of Finance; Banco de Portugal.

Table 3.3 Primary energy consumption

Primary energy consumption disaggregation in Portugal (1990 & 2002) (%)		
	1990	*2002*
Oil products	66	62
Natural gas (entered the market in 1997)	-	11
Coal	15	13
Hydro	5	3
Biomass	11	7
Others	3	4

Primary energy consumption trends in Portugal (1990-2002) (%)			
	1990/1996	*1996/2002*	*1990/2002*
Oil products	2.0	4.1	3.3
Coal	4.0	0.3	2.2
Hydro	10.0	-7.4	-0.9
Biomass	-0.6	0.1	-0.3
Others	8.6	10.2	12.1
Total	2.6	5.1	4.3

Source: ADENE, 2004: 10.

Table 3.4 Final energy consumption

Final energy consumption disagregation in Portugal (1990 & 2002) (%)		
	1990	*2002*
Oil products	58	59
Natural gas (entered the market in 1997)	-	7
Coal	5	1
Electricity	17	19
Biomass	11	9
Heat	5	5
Others	1	-

Final energy consumption trends in Portugal (1990-2002) (%)			
	1990/1996	*1996/2002*	*1990/2002*
Oil products	4.9	4.6	5.4
Coal	-0.6	-12.0	-6.1
Electricity	4.9	6.1	6.4
Biomass	-1.2	0.1	-0.6
Heat	7.0	1.4	4.5
Others	1.5	-	
Total	3.8	4.7	4.8

Source: ADENE, 2004: 11.

Although by the end of the 2000s Portuguese per capita greenhouse gas emissions (7.7 tCO_2e) were low by European standards (10.2 tCO_2e) (Wiesmann et al, 2011: 2773), greenhouse gas emissions grew by about 3% per annum throughout the period 1990-2004, closely tracking the growth of the national economy (MARN and Ecoprogresso, 2006: 8). Since 2000, the rate of emissions growth has decreased following the slowing of economic growth, the introduction of natural gas in 1997, and the growth in the use of renewable energy sources in primary energy consumption since 2004 (DGEG, 2011).

In 2005, most national primary energy demand was still met by imported oil, coal and gas (dependence on these sources for electricity generation accounted for 65–83% of the electricity generated between 1999 and 2007, DGEG, 2007). Renewable energy sources were the only national energy resources, especially hydro for power generation, biomass used both in cogeneration and in the residential sector, and wind energy (especially since 2004) (DGEG, 2011). Following increased mobility needs, the still-growing transport sector increased its historical CO_2 emissions (kt CO_2e) between 1990 and 2004 by 99.4%, whereas in energy-intensive sectors those emissions increased by 33.5% during the same period (Portuguese Ministry of Environment and Eco-progress, 2006: 19-20).

3.3 There was low inclusion of environmental tax design features in energy tax design

3.3.1 The energy tax design was not systematically led by environmental criteria or precise environmental objectives

Precise environmental objectives were assigned to the Portuguese energy tax in 1991, when the tax rate became differentiated according to sulphur and lead content in order to reduce sulphur and lead air emissions respectively, and again in 2006, when the National Climate Change Programme assigned it a role in the reduction of greenhouse gas emissions by industry, with the tax law changed accordingly in 2008. Together with the first step towards the narrowing of the tax rate gap between gasoline and diesel in 2005, these cases accounted for the main references in the energy tax law to environmental concerns, althoug in the first case no reference is made in the text of the law to such concerns.

All these cases seem to be underpinned by a cost internalisation rationale rather than by behaviour steering concerns. There is no reference in any of them to quantified environmental targets, but only to the means to be used. Specific environmental criteria to be used in setting the tax level were not provided either. Although the reference to the need to internalise costs from CO_2 emissions is common to the 2005 and the 2008 changes, in neither case is the calculation process to be applied to such costs clarified. This understanding seems to be further confirmed by the fact that the only reference to the need to change energy consumption patterns in connection with the law changes is indirect (via reference to reduction in greenhouse gas emissions) and done in the context of an environmental law (DL71/2008, 15 April).

Although since 2001 environmental criteria have been mentioned among those to be observed in setting energy tax rates, in 2011 their structure was still not systematically led by such criteria in the sense that one specific environmental hierarchy of behaviours followed from this. Despite the multiple references to the need to reduce CO_2 emissions, the tax rate structure did not communicate the hierarchy of behaviours necessary to achieve this (for example, diesel was taxed less than gasoline). Tax rates followed simultaneously from non-environmental concerns (mainly fiscal and national competitiveness-related) and from several different environmental considerations.

3.3.2 Tax awareness and tax avoidance were not instituted in the tax design

The energy tax had tax illusion rather than tax awareness instituted in its design. This tax followed closely the traditional design of excise duties, which, in order to reduce taxpayers' resistance to the tax, are hidden in the price of the good consumed (Puviani, 1972: 41-42). The tax design also did not reward tax avoidance strategies with a positive environmental impact. It did not include any possibility of tax abatement or refund measures based on environmental criteria. For instance, the higher tax burden on heavy fuel oil was applied regardless of any proven effort made by the company to abate its sulphur emissions.

The legal technique used in the energy tax allowed the price signal to be hidden. To reduce tax fraud and evasion, as well as to reduce administrative costs, the legal obligation to pay the energy tax was laid on economic agents with a major dimension running plants that produced, used or stored fossil fuels or energy products. The moment when the tax became due (in the sense that the state could demand the fulfilment of the legal obligation born when the fuel was produced or imported) was when fuels passed to consumers (Arts. 7 and 8 IEC Code). In practice final consumers at the pump, who were the bulk of the taxpayers, paid the tax in their motor fuel bill. The tax cost was transferred to the taxpayers, together with the fuel price as a whole. Therefore, the tax paid following each purchase was not evident to the taxpayer.

3.3.3. The energy tax payment did not take a ‘forward looking’ approach at polluting impacts

General energy taxes, i.e. taxes raised on measured units of energy products according to a relatively narrow structure of tax rates, are inadequate to shift consumption towards cleaner fuels (fuel-switch), since following their tax base and tax rates they are not able to allow tax payments to accurately mirror relative polluting impacts. Therefore, such payments will hardly be able to communicate the environmental hierarchy of all consumption in relation to a specific pollutant. Consequently, in the absence of strict market segmentation, positive environmental effects of general energy taxes are likely to follow mainly from reduced energy consumption.

In the Portuguese energy tax, the tax base was not a good proxy for specific polluting emissions and the structure and level of tax rates were unable to lead towards fuel-switch or reduced fuel consumption during the 1990s. Following the energy tax base and the energy tax rate structure, the price signal provided by the energy tax did not correctly communicate the environmental hierarchy of consumption with regard to any specific pollutant. And, following the energy tax level, the price signal provided by the tax did not communicate energy scarcity to an accurate degree.

A. The energy tax base was not a good proxy for specific polluting emissions

The energy tax was not raised on a good proxy for specific polluting emissions, since its tax base was measured units of fossil fuels or energy. Therefore, the tax was unable to communicate a coherent environmental hierarchy of behaviours. Since a measured unit of each fuel can produce different environmental damage depending on the pollutant considered, the price signal provided by the tax would only be able to lead towards the environmental hierarchy of behaviours if the tax rate structure mirrored the relative polluting impacts of the products taxed with reference to a specific pollutant, which would require a specific tax rate for each fuel related to its polluting impacts. The high administrative complexity of such a broad tax rate structure does not recommend it. And this was never the case in Portugal.

Furthermore, its narrow definition, without reference to environmental criteria, sheltered some polluting fuel uses from the price signal provided by the energy tax. Some goods not covered by the tax were sometimes at least as polluting as those covered, for example fossil fuels used in public transportation were exempted, whereas those used in private transportation were taxed.

Due to the exemption of any type of fuel used in public transportation vehicles, the energy tax failed to provide the operators of these systems with an incentive to replace their vehicles with new ones equipped with more efficient and cleaner technology. This might have been the case especially regarding private operators. These were led by a profit-maximising objective and the incentive provided by the vehicle acquisition tax (Decree-Law 40/93, 18 February 1993) for the acquisition of second-hand vehicles imported from other member states might also have acted as a restraint on adopting new technology.[39] Some data show that in Portugal between 1987 and 2006, in order to reduce fuel consumption, fleet operators in the road freight transportation sector overlooked the technological improvements of more efficient cars in favour of operational improvements (Matos e Silva, 2011: 2841).

Furthermore, exemptions provided for electricity, which also benefited from a reduced VAT rate (7%) throughout the 1990s, as well as for primary energy sources used in its production, kept the tax from signalling energy scarcity to an electricity demand which has constantly increased (Wiesmann et al, 2011: 2772). By the end of the 2000s, compared with the European Union average, Portugal still had a relatively low per capita consumption of energy (Eurostat, 2009). Regarding gross inland energy consumption per capita, the average Portuguese citizen consumed 2.45 toe, which was 30% less energy than the average citizen of the European Union. However, the difference with regard to electricity consumption was only 20% (Wiesmann et al, 2011: 2773).

Furthermore, the economic efficiency of centrally-led policies underpinned by subsidies was unclear, despite its environmental effectiveness in terms of the rate of penetration of renewable energy sources,especially in power production (Fouquet and Johansson, 2008: 4079; DGEG, 2010). Although

[39] The polluting tendency associated with road transport was influenced by technological factors (ADENE, 2004: 40). Portugal imposed one of the highest tax burdens on the acquisition of new vehicles, together with Denmark, Finland and Greece (the latter only for luxury vehicles). JE, *Fisco asfixia venda de automóveis*, 3 March 2001, online edition.

there were technological restrictions on fuel-switch from fossil fuels (mainly coal and fuel oil) to electricity production based on renewable energy sources, following the characteristics of the electric grid (Reiche et al, 2004: 846), an increase of the price signal provided by the energy tax reaching the power sector could have reduced the cost of making renewable energy sources price-competitive with fossil fuels.

In a price-capped market, as the Portuguese electricity market was until 2004, the policy of subsidies (feed-in-tariffs) to producers of all renewable energy sources used in power production (except large hydropower and municipal solid waste) in order to enhance their price competitiveness vis-à-vis fossil fuels made an important contribution to the deficit in electricity tariffs, especially after 2001 following very interesting feed-in-tariffs and a relaxed mandatory percentage of self-financing (Haas et al, 2004: 837). This reached in 2008 approximately 640 million Euros, payment of which the government decided to spread over a 15-year period, starting in 2010 (DL 165/2008, 21 August 2008).[40] The problem was likely to increase following the targets set by the national energy strategy for 2020 (60% of electricity generation from renewable energy sources by 2020) (Resolution of the Portuguese Parliament 33/2010, 15 April 2010; and Resolution of the Portuguese Council of Ministers 29/2010, 15 April 2010).

B. There was a failure to relate energy tax rates to environmentally correct behavioural change

The energy tax rates were not related to environmentally correct behavioural change or to environmentally correct *effective pollution prices*, either in relative terms or in absolute terms. The reference of the tax rate to environmentally correct relative *effective pollution prices* follows from the consideration of the relative polluting impacts of fuels in structuring the tax rate, whereas the reference of the tax rate to environmentally correct absolute *effective pollution prices* follows from the level at which tax rates are set. In energy consumption, environmentally correct relative *effective pollution prices* lead towards an environmental hierarchy and environmentally correct absolute *effective pollution prices* induce efficient use.

B.1 The tax rate structure was unable to signal an environmental hierarchy of energy consumption

The energy tax rate structure was not related to environmentally correct behavioural change since it did not refer to abatement costs or relative polluting impacts taking into account a specific pollutant. No specific environmental hierarchy of consumption seemed to follow from the structure of the energy tax rates. Sometimes fuels were ranked according to sulphur content (namely fuel oils), while others were ranked according to lead content (namely gasolines), others according to CO_2 content (namely fuels used in industry after March 2009) and a fourth change inversely related to CO_2 content, with the lower rate applying to fuel with a higher CO_2 content (namely gasoline versus diesel). Furthermore, in the tax rate differentiations according to sulphur and lead content, the tax differential was unable to induce important behavioural change following its insufficient level and the regulator's failure to address institutional filters.

Correct price signals by bloc (i.e. the same tax leading to several hierarchies of consumption, all environmentally correct) will still achieve environmentally positive behavioural change when strict segmentation of the energy market is possible in relation to the most representative consumption. Under this condition the tax can guarantee environmentally correct *effective pollution prices*, in both absolute and relative terms, in each segment of the market and in all of them. For example, if energy products used in industry are clearly different from those used in the transport sector and vice versa, two different hierarchies of consumption can be simultaneously communicated without hampering the environmental effectiveness of the tax, as long as one applies only to industrial consumption and the other applies only to propellant consumption.

Since March 2009, the CO_2-related hierarchy of consumption in industry might have been correct in one small area, namely the energy-intensive sectors not covered by the EU ETS and not party to energy efficiency agreements. This would be the case if, with CO_2 emissions, taking into account the technology and process used, the consumption of coal was as polluting as that of oil since both fuels were taxed at the same rate. However, coal is more CO_2 loaded than oil. Furthermore, for the transport sector in 2011,

[40] Information available at ERSE website, at www.erse.pt, accessed on 21 August 2011.

the CO_2-related hierarchy of consumption communicated was still environmentally wrong, since it indicated that diesel was less polluting than gasoline, which is not the case regardless of the technology used.

B.1.1 The wrong CO2-related environmental hierarchy of behaviours

Relative tax burdens on gasoline and diesel have shown environmental incoherence in the Portuguese energy tax rate structure. Diesel has always been taxed at a lower rate than gasoline, in spite of its higher CO_2 and VOC content and not much greater energy efficiency. The higher efficiency of diesel engines when measured in litres tends to be reduced to a great extent when some aspects are taken into account, namely the fact that diesel cars tend to be heavier than gasoline cars, and following corrections for the higher energy content of diesel and for fleet averages (the diesel and gasoline cars actually chosen on the market as opposed to comparing identical vehicles with different motors) (Schipper et al, 2002: 305; Sterner, 2007: 3199).

Although the gap has been progressively narrowed since 2005, with reference to the need to internalise the costs of CO_2 emissions, in January 2011 the tax rate structure was still not related to environmentally correct relative *effective pollution prices*. Heating diesel was taxed at a lower rate than road diesel and the latter was taxed less than gasoline. A consumption shift towards diesel might have been part of the adaptation to higher fuel prices.

Following the more favourable treatment for diesel, between 1993 and 2003 Portugal experienced one of the highest increases of diesel penetration in EU-15, at 319% (Zervas, 2010a: 5415). The more favourable tax regime for diesel fuels than for gasoline and the high average age of the vehicle fleet were reported as the main inducements to pollution by fine particles in Portugal during the 1990s (OECD, 2003a: 12).

In 2008, it was decided to gradually correct the price signal provided by the energy tax regarding the CO_2-related environmental hierarchy of behaviours. However, this correction applied only to fuels used in industry, namely coal, oil and fuel oil, and the price signal following only applied to a small part of the industry, namely energy-intensive sectors not covered by the EU ETS and not party to energy efficiency agreements.

B.1.2 The insufficient price signal applied to the sulphur- and lead-related hierarchy of behaviours

The energy tax rate differentiations according to sulphur content in fuel oil and to lead content in gasoline presented low levels of environmental effectiveness. Clear fuel substitution effects following from the energy tax rate differentiation were not obvious and both cleaner fuels, namely light fuel oil and unleaded gasoline, were unable to reach relevant market shares until other types of regulatory instruments were adopted. Such low effectiveness seems to have been due to poor inclusion of the design features of environmental taxes in the respective energy tax differentiations. The system of tax benefits for light fuel oil and unleaded gasoline worked as indirect subsidies aimed at lowering the financial burden refineries would need to take on with the technological investment required to produce the cleaner fuel, rather than as price signals to polluters to lead them towards cleaner fuel consumption.

In both cases, the failure to remove institutional filters caused by the uncompetitive market structure that kept the price signal provided by the tax from fully reaching the polluters able to prevent pollution meant that tax rates were unable to guarantee environmentally correct *effective pollution prices* able to induce behavioural change. In both cases this incapacity was caused by a price difference insufficient to induce relevant behavioural change. In the case of fuel oil, *effective pollution prices* were also unable to transmit an environmentally correct hierarchy of consumption. Furthermore, regarding the tax rate differentiation according to sulphur content, the narrow subjective tax incidence hindered the environmental effectiveness of the tax.

The energy tax rate differentiation according to sulphur content in fuel oil

It was not possible to assign clear fuel substitution effects to the energy tax rate differentiation according to sulphur content. This was not able to bring the market share of the light fuel oil up to relevant levels. Some design features of this differentiation might have hindered its environmental effectiveness, namely the failure to transmit the full price signal provided by the tax to polluters able to avoid pollution and especially to major polluters.

Energy tax exemptions assigned to major consumers (43% of the total market) sheltered from the price signal provided by the tax an important part of the polluting sources, namely the power generation sector (Santos *et al*, 1999: 461). This sector was responsible for an important percentage of fuel oil consumption. The major consumer of fuel oil, the Portuguese Electric Company (EDP), consumed mainly HSC fuel oil (3.5% sulphur content) (Santos *et al*, 1999: 461). This company was exempted from the energy tax and did not pay any tax on its sulphur emissions. Therefore, only emissions standards forced it to shift towards cleaner fuel oil, although not LSC fuel oil but fuel oil with about 3% sulphur (Santos *et al*, 1999: 461). The same was true for cogeneration units. Most of these were linked to the national electricity grid and therefore exempted from energy taxation (Santos *et al*, 1999: 461).

Another design feature that might have impacted negatively on the environmental effectiveness of this tax differentiation was the failure to relate the tax rates to environmentally correct relative *effective pollution prices*. This was not due to the absolute or relative level at which tax rates were set, but to the failure to remove the institutional filters that kept the price provided by the tax from fully reaching the polluters able to prevent pollution. The uncompetitive market structure for fuel oil should have been taken into account by the regulator in order to guarantee that the tax rate differentiation would translate into relative *effective prices* for heavy fuel oil and light fuel oil which would make the clean fuel more competitive than the dirty one and consequently induce a shift in consumption from heavy fuel oil to light fuel oil.

Market demand is affected by the *effective fuel price*, which includes the total tax burden on the fuel as well as all the other components of its price to the consumer. In July 1997, the tax share (energy tax plus VAT) in the sales price was about 28.4% for HSC fuel oil and 18% for LSC fuel oil (Santos *et al*, 1999: 458). Therefore, the tax system privileged the cleaner fuel. However, *effective fuel oil prices* were not environmentally correct, since the average sales price was PTE 2 higher for LSC fuel oil than for HSC fuel oil. Under these conditions, a rational consumer would only prefer the cleaner fuel if it had a better operational performance.

Still there was a considerable increase in the market share of LSC fuel oil (267% in 1991-1992, the first year after the introduction of the tax incentive, and 58% in 1992-1993) (Santos *et al*, 1999: 458). Fuel substitution seems to have had some impact on total SO_2 emissions (in 1991 these were reduced by 0.62%, in 1992 by 1.92% and in by 1993 3.6%, with a reduction for the whole period 1980-1998 of 5%, Eurostat, 2001). Despite the rapid growth in the market share of LSC fuel oil, which was 1.3% in 1991, 4% in 1992, 8% in 1993 and 10% in 1994, this product was not able to achieve a relevant market dimension (Santos *et al*, 1999: 458). In 1995, its market share was 11% (ibidem). As in the case of unleaded gasoline, further progress was made in a subsequent phase due to the imposition by EC regulation of maximum sulphur content of 1.5% for fuel oil. HSC fuel oil was only eradicated from the market in 2003 due to EU legislation (Directive (1999/32/EC)).

Both oil companies' commercial policy, following their interest in the market expansion of LSC fuel oil, and technological progress might have induced improvements in SO_2 emissions from fuel oil consumption (Santos *et al*, 1999: 458). How much of the change in consumption patterns for fuel oil was due to the tax differentiation was difficult to identify. However, it has been reported that in 1993 only 8% of its estimated potential had been achieved (Santos *et al*, 1999: 461). Two aspects have been mentioned as having hindered the achievement of its whole potential, namely the narrow subjective tax incidence of the energy tax and the non-transference to the consumer of the tax incentive (ibidem).

The failure to consider the fuel oil market structure led to the disappearance of the tax incentive in the sales price and its pocketing by oil refineries (Santos *et al*, 1999: 461). The fiscal incentive was not completely translated into market prices. Production costs and profit margins explained the fading away of the tax differentiation and the interest of oil companies in promoting LSC fuel oil, which allowed them higher profit margins. They were overcompensated by the production of such fuel oil, since they kept a reward for their extra production and were still allowed to keep LSC fuel oil more expensive than HSC fuel oil.

The energy tax rate differentiation according to lead content in gasoline

The energy tax rate differentiation according to lead content showed low environmental effectiveness. The total market share of unleaded grew from 1.8% in 1990 to 41.4% in 1996, whereas in Sweden the market share following tax differentiation decreased from 100% in 1986 to 40% in 1992 and practically zero in 1993 (Hammar and Löfgren, 2000: 14). However, this might have been due not only to deficient tax design following the non inclusion of the design features of environmental taxes, but also to the pace at which the renovation of the car fleet occurred in Portugal.

The tax rate differentiation according to lead content of gasoline not only was not set at a level sufficiently high to induce strong and fast behavioural change (between 1991 and 1997 its maximum was Euro 0.05), but also, as in the fuel oil case, was not fully translated into market prices (Table 3.5). The strategic behaviour of oil companies following the oligopolistic market structure allowed them to pocket the tax differential by incorporating part of the tax incentive as increased profit margin. The failure of the regulator to remove these institutional filters meant that the tax rates did not guarantee relative *effective pollution prices* able to induce high behavioural change.

Table 3.5 Energy tax applied to unleaded and leaded gasoline (1991-1996, PTE/litre) and unleaded and leaded gasoline sales price (PTE/litre, bold&italic)

	Leaded	*Unleaded*		*Tax differential*
		95	*98*	
1990	***137***	***131***	***-***	6
1991 **Art. 7 (6) DL 261-A/91, 25 July 1991**	93-98 ***147***	81-86 ***137***	-	12
1992 ***Art. 40 (5) Lei 75/93, 20 December 1993***	89-100 ***146***	77-88 ***136***	-	12
1993	86-93 ***151***	74-81 ***141***	74-81 ***151***	12
1994 **Arts. 1 and 2 Portaria 326-A/94, 27 May 1994**	91 ***155***	84 ***152***	84 ***155***	7
1995	***155***	***153***	***156***	2
June 1996 **Arts. 1 and 2 Portaria 224-B/96, 24 July 1996**	97 ***163***	90.7 ***158***	90.7 ***164***	6.3
December 1996 **Arts. 1 and 2 Portaria 769-A/96, 30 December 1996**	98.3	91.5	91.5	6.8
1997 **Arts.1 and 2 Portaria 496-A/97, 17 July 1997**	***168***	***163***	***169***	5

Adapted from: Santos et al, 1999: 462.

The share of taxes (energy tax and VAT) in market prices in July 1997 was circa 72% in leaded, 70.7% in 95 octane unleaded gasoline and 68.7% in 98 octane unleaded gasoline The relevant variable for the consumer's decision-making process, i.e. the *effective fuel price*, did not accurately reflect these tax differentials. The market price differential between unleaded 95 octane gasoline and leaded gasoline was on average PTE 10/liter in the 1991-1993 period and PTE 2/liter in 1995. During the 1991-1997 period this difference was lower than the corresponding tax differential. The reasons for this are similar to those noted in the fuel oil case (Santos *et al*, 1999: 461-464). The tax differential and production costs accounted only partially for the price differential (ibidem). The other part was explained by the commercial strategy of oil companies. From 1991 to 1993, the tax differential accounted for 83.3% of the price differential, but this value fell to 42.9% and 28.6% in 1994 and 1995 respectively, going up again in 1996 (73.5%) and 1997 (73.5%) (ibidem).

The strategy followed by the market operators was also evident in the case of the 98 octane unleaded gasoline, which was introduced in 1993 and was not under a MMP regime. This fuel had the same tax incentive as 95 octane unleaded gasoline. However, its market price was equal to leaded gasoline until 1995 and PTE 1/liter higher after that. Health reasons were given for the differential, since 95 octane unleaded gasoline was considered less harmful to human health than 98 octane unleaded gasoline (Santos *et al*, 1999: 461-464; GT ISP, 1999: 157).

The delay in inducing behavioural change following the introduction of the tax differential might have been due to pressure from the oil refineries lobby, who tried to postpone the investments required to produce unleaded gasoline as long as they could (JE, *Governo quer diminuir peso do chumbo*, 22 November 1997, online edition), as well as to tax revenue concerns. Until 2000, each litre of leaded petrol sold provided higher total tax revenues (energy tax plus VAT) than the same amount of unleaded petrol (Table 3.6) (DGE, 2001). Consequently there might have been a lack of political commitment to eradicating unleaded gasoline from the Portuguese market. It is worth noting that this tax differentiation came later than in other countries. It was introduced in Portugal in 1991, whereas for instance in Sweden gasoline taxes had been differentiated with respect to lead content since 1986 (OECD, 1994b: 58).

Table 3.6 Tax structure of fossil fuel prices between September 2000 and January 2001

	Additivated gasoline	Unleaded gasoline	Gas oil	Fuel oil with less than 1% of sulphur	Fuel oil with more than 1% of sulphur
Energy tax	30.4%	31.7%	37.9%	5.5%	8.3%
VAT	14.5	14.5%	14.6%	10.7%	10.7%

Source: DGE (2001).

The main explanatory variable in the evolution of the unleaded petrol market share used by Hammar and Löfgren (2000) in the Swedish case was also referred to in the Portuguese case (Santos *et al*, 1999: 463). The renovation of the car fleet might explain the pace of the evolution observed in the market share of unleaded gasoline better than the energy tax differential. The slow increase in the number of new cars equipped with cleaner technology might help to explain the slow increase in unleaded market share, which was below that achieved in other EU countries, for example Sweden (Hammar and Löfgren, 2000: 14; see Chapter IV, Point 1(1)(a)).

By the end of the 1990s, data evidenced an old private car fleet in Portugal. Between 1980 and 1999, the average age of passenger cars increased in the EU-15, with a consequent slowdown in the penetration rate of modern cleaner technologies. In 1998, Portugal held the highest value (11 years), far older than the EU-15 average (7 years). By the end of the 1990s, it was estimated that 30% of the vehicles circulating in Portugal were on average 12 years old (JE, *Impostos nas estradas*, 27 November 1999, online edition). Furthermore, in Portugal vehicle abatement accounted for 1% of the total number of vehicles sold, while for instance in neighbouring Spain this was *circa* 20% (2000 data) (Semanário Económico, 18 December 2001).

The understanding that the pace of the renovation of the car fleet affected the environmental effectiveness of the tax differential is also supported by the lack of effect of the 1994 reduction in the tax differential on the evolution of market share (21.9% in 1993, 29.6% in 1994 and 35.3% in 1995). Cars using both leaded and unleaded gasoline were sold in Portugal from the second half of the 1980s and cars with catalytic converters entered the market in 1991 and were compulsory from 1993 onwards. This might explain the continuous evolution of unleaded market share from 1993 to 1995, despite the discontinuous evolution of the tax differential during the same period.

B.2 The energy tax level was unable to lead towards the adoption of energy conservation strategies

The energy tax rates were not set sufficiently high to induce energy conservation strategies, either in the short term, by shifting private vehicle use to public transportation, or in the long term, by shifting vehicle demand towards cleaner and more fuel-efficient vehicles. In times of fuel price increase on the world market, an indexation of fuel prices to consumer price indices or to nominal growth of GDP (as an indicator of income development) is recommended (Schlegelmilch and Markovič-Hribernik, 2002: 78). Instead of such an indexation, the energy tax was used to stabilise energy prices on the Portuguese market until 2004, which led to a decrease in real energy prices (MFP, 2001: 37). The regular updating of energy tax rates in line with inflation started only in 2005 (Art. 34 Law 55-B/2004, 30 December 2004).

Using the energy tax as a price stabiliser helped to boost demand for private-use transportation

In the short to medium term, the reaction of motor fuel consumption to an energy tax mainly depends on the shift from private vehicle use to public transportation. However, following the insufficient level of the Portuguese energy tax, the bulk of the tax on fuels used in private vehicles and exemptions for those used in public transportation do not seem to have been able to lead towards such a shift. In Portugal, during the 1990s, there was a boost in demand for private-use transportation following changes in travel patterns, higher incomes and a fall in private transport prices and energy products in real terms (EEA, 2000e: 73; IEA, 2003).

During the 1990s, the car increased its share of passenger transport in Portugal and occupancy rates decreased (Lacasta and Barata, 1997: point 3). In general, the stock of passenger cars correlates well with GDP per capita (EEA, 2000a: 37). Therefore, between the second half of the 1980s and the first half of the 1990s, growth in the transport sector amounted to 67% due to increasing purchasing power in the 1980s (Lacasta and Barata, 1997: point 3). In the period 1985-1997, road transport increased by 120% in Portugal, which thus experienced, together with Luxembourg, the highest increase in the EU (EEA, 2000e: indicator 1). Between 1970 and 1997, Portugal was among the EU member states with the highest growth in the number of passenger cars (6.9%) (EEA, 2000e: 106).

The decline in real motor fuel prices helped to lower the cost of road transport, which was an important factor in stimulating demand for transport. The trend experienced since 1970 in the Portuguese road transport sector resulted from a general decline in world oil prices, as well as price ceilings on diesel fuel, gasoline and fuel oil (ibidem). In 1998, for example, among the EU Member States unleaded fuel prices were highest in Sweden and lowest in Portugal (EEA, 2000e: indicator 15). Diesel prices followed a similar pattern (EEA, 2000e: indicator 16). External costs produced by road transport were hence decreasingly reflected in prices (OECD, 1993: 97). Energy pricing did not successfully internalise environmental externalities, let alone reflect relative scarcity (Bronchi and Gomes-Santos, 2001: 22).

Energy tax revenues were never dedicated to increasing the price-elasticity of private transportation through investment in better public transportation. According to the 1995 Portuguese Environmental Policy Programme (Resolution of the Portuguese Council of Ministers 38/95, 21 April 1995), revenues from pollution taxes should preferentially be put to the environmental re-qualification of the sector where they were collected, in order to improve public acceptability of the instrument and transparency in its administration. During the 1990s, there was some discussion regarding the use of energy tax revenues, with several unsuccessful attempts from left-wing parties and the Green Party to use fuel and vehicle taxation to improve the quality and environmental performance of public transport.

The energy tax does not seem to have relevantly impacted on long-term vehicle demand

In the long run, when fuel price elasticities are higher, general energy taxes can impact on vehicle demand, shifting it to more energy-efficient and less polluting models. In 2011, such causal connection between the characteristics of the passenger car fleet in Portugal and the energy tax had not yet been addressed by any impact assessment study. However, it was not evident that the Portuguese energy tax had influenced vehicle demand towards smaller and lighter vehicles.

In 2003, Portugal was among the countries with the highest percentage of small cars (Zervas, 2010b: 5440). In the same year the country was also among the EU-15 with lighter cars (1291 Kg), including diesel passenger cars (1038 Kg), with a relatively constant average weight of passenger cars (and even with a small decrease in 1999-2000), in contrast to the EU tendency (Zervas, 2010a: 5416). Consequently it was also among the EU-15 with the least powerful diesel PCs (75 kW) and gasoline PCs (58 kW) (Zervas, 2010a: 5419).

CO_2 emissions depend on several parameters, such as driving profile, annual mileage and, as a consequence, real-world CO_2 emissions are different from the CO_2 emissions obtained according to the official European certification procedure on the New European Driving Cycle (NEDC) (Zervas, 2010a: 5414). For example, CO_2 emissions can decrease due to increased combustion efficiency (e.g., due to the use of lighter vehicles or low fuel consumption fuels), leading to lower fuel consumption and thus to lower CO_2 emissions.

Studies based on the CO_2 exhaust emissions of new passenger cars obtained according to the NEDC show that EU14 (excluding Greece) average diesel emissions decreased from163g/km in 1995 to 134g/km in 2003 (-18%), and average gasoline emissions from 207g/km to 168g/km (-19%) (Zervas, 2010a: 5420). 93% of this decrease in the case of diesel passenger cars and 87% in the case of gasoline cars was achieved only until 2000 (Zervas, 2010a: 5421).

Portugal performed above the EU-14 average regarding the evolution of CO_2 exhaust emissions of new passenger cars. The CO_2 emissions show a very small decrease after this year, which indicates that the progress in CO_2 exhaust control after 2000 is very small (ibidem). This small decrease is partially due to Euro3 emissions limits (such as those on catalysts) (ibidem). Since 1995 until 2003, in Portugal, the average CO_2 emissions decreased by 21% and kept reducing even after 2000 (-4%), whereas for gasoline CO_2 there was not such a steep decrease (Zervas, 2010a: 5422).

The demand shift towards smaller and lighter vehicles, with its consequent positive environmental effect, cannot be attributed to any relevant extent to the vehicle acquisition tax, since a differentiation in the tax rate according to technology-dependant CO_2 emissions was only introduced in 2007 (Law 22-A/2007, 29 June 2007). Such a shift might have followed inter alia from a potential incentive provided by the energy tax as a component of *effective fuel prices*, but it might also to some extent have been a consequence of lower national income levels compared to EU-15. The fact that CO_2 emissions were more affected by changes in the diesel car fleet than in the gasoline car fleet, with its higher energy tax burden, might indicate the minor role played by the energy tax in vehicle demand and consequent environmental improvement.
In any case it must be taken into account that demand for new vehicles in Portugal was just a small part of total vehicle demand, with the remaining demand being for polluting and inefficient second-hand vehicles. Although between 1993 and 2003 Portugal was among the EU-15 with the most significant increase in passenger cars, with an increase of 32%, in 2003. Portugal presented the lowest number of new passenger cars per 1000 in Europe-15, with 16 vehicles per 1000 (Zervas, 2010a: 5414). In 1995, the average normalised specific consumption test values of new cars was 7.2 and 6.4 1/100km, for gasoline and diesel cars respectively, whilst official data reported 10.3 and 9 1/100km for gasoline and diesel cars respectively in the national stock in circulation (ADENE, 2004: 40). In 2000, test values of new gasoline and diesel cars were 6.6 and 6.2 1/100km respectively, while data showed that in Portugal these values were 9.6 and 8.5 1/100km for gasoline and diesel cars respectively (ibidem).

3.3.4 The subjective energy tax incidence was not related to capacity to prevent pollution

The energy tax payments were not imposed on polluters according to their capacity to avoid pollution. This design feature is also likely to have hampered the energy tax effectiveness in cutting environmentally harmful national patterns of energy consumption. The allocation of the energy tax burden between the two major national sectors causing energy-related pollution, namely the transport sector and industry, did not take into account their relative improvement potential.

The 2008 law change brought the subjective incidence of the energy tax more into agreement with the capacity to avoid pollution, at least for a small part of industry, by bringing under the tax coverage energy-intensive sectors not covered by the EU ETS and that were not party to energy efficiency agreements. However, following the still broad exemptions for energy-intensive industries, in 2011 the bulk of energy taxpayers were consumers at the pump displaying relatively low price elasticities even in the long term, (see Section 3.1 of this chapter).

In 2011, it was still early to assess the impact of the 2008 law changes. However, based on the negative impact on the environmental effectiveness of the energy tax rate differentiation according to fuel oil sulphur content of the exemptions provided to industry, and the unchanged pattern of national energy consumption during the 1990s, it might be reasonable to argue that the narrow subjective incidence of the energy tax did not help its environmental effectiveness.

At the beginning of the 1990s, the biggest energy consumers in Portugal were the transport sector and industry, which together accounted for 80% of total energy consumption (Table 3.7) (OECD, 1993: 88). In general, and for the period 1985-1998, demand for petrol and diesel fuels used by road vehicles was the main driving force behind the rise in total energy consumption (EEA, 2002b: 1). Road was the biggest energy consumer sector, accounting for approximately 72 and 92% of transport energy consumption in 1999 and 2002 respectively (Table 3.7) (EEA, 2002b: 1; ADENE, 2004: 31). The approximately 124.5% increase in the national stock of road vehicles during the period 1990-2002 (*circa* 10.4% per year), reaching around 4.2 million on 2002, helps to explain the figures (ADENE, 2004: 18).

Table 3.7 Final energy consumption by mode and by energy

Disaggregation of national transport final energy consumption by mode (1990 & 2002) (%)		
	1990	*2002*
Road	87	92
Rail	2	1
Air	7	6
Water	4	1
Total Mtoe	3.6	6.9

Yearly average trends in transport sector final energy consumption by energy form in Portugal (1990-2002) (%)			
	1990/1996	*1996/2002*	*1990/2002*
Diesel	6.3	12.3	11.6
Gasoline	6.8	0.8	4.0
Jets	4.8	3.8	4.8
Fuel oil	-6.3	-5.0	-4.7
Electricity	8.4	3.1	6.5
Others	-0.8	5.9	2.4

Source: ADENE, 2004: 18-19.

The strong fossil fuel dependence of the transport sector has continued during the period 1990-2010 (DGEG, 2011). During the 1990s, the industry and transport sectors, both traditionally highly dependent on fossil fuels, have not improved their fuel mix significantly, sustaining the negative national energy pattern regarding the intensity and structure of energy consumption (Table 3.8). Benefits from energy conservation efforts, the introduction of natural gas and the success of co-generation projects were more than compensated for by a strong growth in the transport sector, particularly in coastal areas (OECD, 1993: 88). In 2005, transport and industry still bore the lion's share of national final energy demand, with 44% and 26% respectively of total final energy demand (DGEG, 2007; Simões et al, 2008: 3598).

Table 3.8 Final energy consumption by sector

Structure of final energy consumption by sector in Portugal (1990 & 2002) (%)		
	1990	*2002*
Industry	38	34
Transport	30	36
Services	7	11
Households	20	16
Agriculture	5	3
Total	12.2 Mtoe	19.1 Mtoe

Yearly average trends of final energy consumption by sector in Portugal (1990-2002) (%)			
	1990/1996	*1996/2002*	*1990/2002*
Industry	2.7	3.3	3.3
Transport	5.9	6.6	7.4
Services	8.8	12.7	14.1
Households	1.5	2.0	1.9
Agriculture	1.7	-3.3	-1.0

Final energy consumption disaggregation in Portuguese industry (1990 & 2002) (%)		
	1990	2002
Oil	40.6	37.8
Coal	14.1	2.7
Electricity	20.4	20.6
Heat	13.9	15
Natural Gas	-	16
Others	10.9	7.8

Yearly average trends in final energy consumption by sector in Portuguese transport (1990-2002) (%)		
	1990	2002
Diesel	47	60
Gasoline	40	31
Jets	7	6
Fuel oil	3	1
Electricity	1	1
Others	2	1

Source: ADENE, 2004: 11, 13 and 18.

CONCLUSIONS

The analysis given in this chapter of the Portuguese energy tax was aimed at assessing whether its potential high strategic relevance for energy policy was transferred into a tax design with strong inclusion of the features of environmental taxes and how the degree of inclusion affected the environmental effectiveness of this tax. It was concluded that the inclusion was low and that such a low level of inclusion caused a low impact of the tax on the pattern of national energy consumption during the same period (inefficient and highly dependant on fossil fuels) and consequently on pollution following from energy consumption in Portugal during the 1990s. Furthermore, following the strict compliance of the tax with the energy taxation structure proposed by the Energy Taxation Directive (2003/96/EC) and the fact that it was the only tax charged during the period on national energy consumption, this has allowed us to assess the low environmental effectiveness of such a tax design.

Precise environmental objectives were assigned to the energy tax on two occasions: in 1991, in order to reduce energy-consumption-related sulphur and lead emissions through tax rate differentiation, and in 2008 in order to reduce the greenhouse gases in industry through a gradual increase in maximum rates applied to industrial fossil fuels. Environmental concerns were only clearly mentioned in the law in the latter case. These two cases, together with the gradual process initiated in 2005 of moving towards taxation of gasoline and diesel (including heating diesel) at the same rates, account for the main signs of environmental concern in energy tax design since 1990.

These cases express three different environmental concerns, namely sulphur emissions reduction, lead emissions reduction and greenhouse gas emissions (specifically CO_2 emissions) reduction. A single overarching environmental objective for the energy tax only comes in 2008, with the reference to greenhouse gas emissions. It is still not stated for the whole economy, but only for industry. This can explain the failure of the tax to communicate the environmental hierarchy of consumption related to a specific pollutant.

We cannot see the use of environmental criteria in the tax design, despite the reference of the law since 2001 to the use of relative polluting impacts to set the tax rate. In none of the cases referred to, which correspond to those where the linkage between environmental concerns and changes in the energy tax law was the closest, are the potential environmental criteria used to set the tax level clarified. Even when the law mentions the internalisation of CO_2 costs to explain the change, as was the case in 2005 and 2008, the method used to calculate such costs and their transference to the tax rates is not provided.

Following these features, on both occasions when precise environmental objectives were assigned to the energy tax as well as in the gasoline and diesel tax rate convergence process initiated in 2005, the rationale underpinning the law seems to have been cost internalisation rather than steering behaviour. This understanding is in general confirmed by the energy tax design as far as the tax base, tax rates and subjective tax incidence are concerned, as well as the relevance assigned by the energy tax law to the principle of equivalence and its lack of references to behaviour-steering intentions. The prevalence of a cost internalisation, rather than a behavioural steering rationale in the Portuguese energy tax is coherent with its lack of communication of a precise environmental hierarchy of consumption. Likewise, the institution of tax illusion rather than awareness in the tax design and its failure to reward tax avoidance strategies with a positive environmental impact, both usually found in traditional excise duty design, is also coherent with a cost internalisation approach.

The energy tax base was not a good proxy for specific pollution emissions, since it corresponded to measured units of fuels which may cause different amounts of emissions depending on the pollutant considered. Following this type of tax base and the relatively narrow structure of tax rates used, the energy tax payments were unable to mirror relative polluting impacts and consequently to lead towards any specific environmental hierarchy of consumption. This was true with regard to both polluting emissions and energy efficiency, since the tax design made it impossible to target the price signal to a specific pollutant or to energy content.

Moreover, the definition of the tax base did not follow from environmental criteria, which led to the provision of shelter from the price signal consumptions for sectors with an environmental impact at least as negative as those not exempted. Such narrow coverage is likely to have kept the energy tax from providing an incentive to public transportation operators, especially private operators, to adopt new technology. Meanwhile, exemptions for fossil fuels used in the power sector is likely to have helped the users of these fuels to keep ahead of the competition, raising the costs of the measures adopted to increase the use of renewable energy sources. Such exemptions, together with those assigned to electricity, increasing demand for which did not allow a severing of energy demand from GDP in Portugal, hindered the capacity of the energy tax to communicate energy scarcity.

The energy tax rates were not related to behavioural change, either in absolute or in relative terms, since they were unable to guarantee environmentally correct absolute and relative *effective pollution prices* for all the products covered by the tax, which until 2005 were mainly motor fuels. They were not related to pollution abatement costs or to relative polluting impacts with regard to a specific pollutant or to energy content. Consequently, during the 1990s taxation rates were unable to steer energy consumption towards cleaner fuels and energy conservation strategies.

The tax rate structure communicated simultaneously different hierarchies of consumption, which, following the energy market segmentation, would not be a problem if a single and environmentally correct hierarchy was communicated to each market segment, allowing the tax to steer consumers' choices towards less environmentally damaging patterns. But this has not been the case. The hierarchy communicated to industry with reference to CO_2 became close to be correct in 2008. However, the tax rate applied to coal and oil still communicated wrongly their equal polluting impact. Regarding the hierarchies communicated to the transport sector referred to sulphur, lead and CO_2, only the first two were correct, due to the tax benefit to diesel still present in 2011.

Furthermore, in absolute terms, the energy tax rates were weakly able to steer behaviour towards pollution prevention by inducing cleaner and efficient energy consumption. The incentive provided by the tax rate differentiation according to sulphur and lead content in fuel oil and gasoline respectively was insufficient to induce a fast increase in the market share of cleaner fuel. The uncompetitive Portuguese energy market structures aggravated this deficiency in the tax design and proved the relevance of competitive markets for environmental tax effectiveness. This case was also useful to test the relevance for environmental effectiveness of relating the tax rate to environmentally correct *effective pollution prices,* instead of partial components of the price, such as the tax burden, reaching the consumer. Furthermore, it has shown the importance of transmitting the price signal to those able to avoid pollution.

Furthermore, energy tax rates led to decreasing real energy prices during the 1990s. In the short term, this was true not only because of the minimal price inelasticity of energy consumption following mainly from technological constraints, but also due to the tax failure to harvest the potential gains from shifting private vehicle use to public transportation, feasible in the short term. In the long term, when price-elasticity of fuel consumption is higher, the energy tax also did not evidence a positive environmental effectiveness. Following the price signal provided by the tax, private-use transportation increased and it is not reasonable to argue that the energy tax had a relevant role in shifting vehicle demand towards smaller and lighter vehicles.

As far as the subjective incidence of the energy tax is concerned, a cost internalisation rationale is also evident. The tax was mainly raised on the sector showing the lowest price elasticities in energy consumption, namely the transport sector. This was the best payer, but not the polluter enjoying the best capacity to avoid pollution. Industry, and especially energy-intensive industry, though displaying higher price elasticities in energy consumption, has always been sheltered from the price signal provided by the energy tax. This continued to be the case after the 2008 law change, following the enlargement of the definition of energy-intensive industry and of the consequent opportunities to gain exemptions by joining energy efficiency agreements.

Due to the low inclusion of the design features of environmental taxes in the energy tax, it was very similar to an excise duty, despite the several references in the law, especially since 2001, to environmental concerns. Its design and management were retained by the entity traditionally in charge of excise duties, namely the Ministry of Finance. This design made it a blunt instrument for inducing behavioural change towards cleaner and more efficient energy consumption, acting mainly as an instrument of cost internalisation. In Portugal, improvements experienced in energy consumption, which were mainly noticed in primary energy consumption (electricity production) after the mid 2000s, have mainly followed from command-and-control-based policy. Despite the strategic importance of the energy tax for Portuguese environmental policy, its failure to include the relevant design features has hindered its potential role in such policy.

CHAPTER IV

SWEDISH ENERGY TAXATION PACKAGE

Swedish legal vocabulary and abbreviations

Statens Offentliga Utredningar (SOU)	The Swedish Government Official Reports series, Committee Reports
Departementsserien (Ds)	Ministry publications series
Svensk Författningssamling (SFS)	The Swedish Code of Statutes

This chapter uses empirical data from the Swedish energy taxation package to test the environmental relevance of the design features of environmental taxes explained in the Introduction and of using taxes with different degrees of inclusion of the referred features in the same system. The package comprised taxes levied on fuel consumption (the CO_2 tax, the sulphur tax and the energy tax) and the nitrogen oxide (NO_x) charge, which was directly connected to energy production. To simplify the presentation of the comparative analysis between the four taxes (regarding their degree of inclusion of the design features of environmental taxes and consequent environmental effectiveness) done in Section 2.2 of this chapter, Sections 1 below does not provide only a description of the tax design, which was the method used in Chapters II and III, but already provides some critical analysis regarding the aforementioned degree of inclusion.

Empirical evidence showed behavioural responses were stronger and faster the more the tax included the design features of environmental taxes. This involved leadership by environmental criteria and precise environmental goals, as well as tax payments mirroring the evolution of the environmental damage caused, with a 'forward looking' approach at polluting impacts and imposed on those economic agents able to avoid pollution. A decreasing scale of inclusion of design features of environmental taxes and environmental effectiveness was recorded as one progressed from the NO_x charge and the sulphur tax towards the energy tax, passing the CO_2 tax.

The energy tax only scarcely included the design features of environmental taxes. Its design was not led by environmental criteria and missed precise environmental objectives. The same is not true regarding the CO_2 tax. This tax included a precise (though unquantified) environmental objective. And environmental criteria were used to select its tax base and to setup its nominal rates. However, both taxes were very much revenue-oriented. With the exception of the CO_2 tax rates until 1993 and the energy tax differentiation according to environmental classes, their tax rates were unable to lead behaviours towards more sustainable energy consumption. Their subjective tax incidence was not based on the capacity to avoid pollution but on best payment with level-related tax departures.

Despite the close linkage between the tax base and environmental damage in both taxes, only the CO_2 tax base was specific enough ('carbon content') to allow tax payments to mirror the evolution of environmental damage, which explains the stronger (still only visible in the long term) environmental impact of this tax compared to the energy tax. The latter was raised per unit ('weight or volume') of fuel. Each unit of a specific fuel is able to produce different polluting impacts depending on the specific pollutant considered. Therefore, in the absence of a tax rate per fuel, the energy tax payments were unable to mirror the environmental damage following from energy consumption.

The sulphur tax and the NO_x charge included most design features of environmental taxes. This explains the strong and quick environmental impact following their introduction. They were led by environmental criteria, aimed at specific environmental objectives and had tax awareness and tax avoidance instituted in their design. In practice their tax bases were specific polluting emissions, which allowed tax payments to mirror the evolution of environmental disruption. Their tax rates based on abatement costs were able to steer behaviours towards the environmental objectives pursued especially by inducing technological progress. In both taxes taxpayers were polluters able to avoid pollution.

Both the sulphur tax and the NO_x charge evidenced a linkage between the high technical complexity of the tax design and the predominant institutional role played by the Ministry of Environment. In the NO_x charge, the risk of having this entity interested in the tax base was overcome with full revenue recycling back to the industry. This enhanced the inclusion of design features of environmental taxes and the environmental effectiveness of the tax. It raised tax awareness and allowed selective tax incidence and charge levels sufficiently high to steer behaviours.

The failure to refer all the taxes charged on each energy product to environmentally correct *effective pollution prices* hindered the environmental effectiveness of the Swedish energy taxation system and its parts. This failure made the CO_2 tax and the sulphur tax unable to compensate for reductions in the energy tax, which sometimes led to environmentally wrong price signals. In the heating sector, oil and coal had the same *effective pollution price*. In the industry, oil had a lower *effective pollution price* than electricity. And heavy fuel oil was not imposed a higher *effective pollution price* than light fuel oil. Therefore, the level of fuel switch towards biomass has also varied among sectors, since the relative position of this fuel in the environmental hierarchy of consumptions was influenced by the *effective pollution price* of the other fuels and this was not the same in all the sectors.

The Swedish 'energy tax tool box' was highly relevant in the environmental tax reform process (hereafter also ETR) carried out from 1990 onwards (NCM, 1996: 108; OECD, 1994a: 75). In the early 1990s, these taxes were the most directly aimed at pressing national environmental problems, namely air emissions and their environmental consequences, among which acidification played a major role, being environmentally motivated and covered by the OECD/EEA database as environmentally related. Availability of evaluation studies explains the analysis focused on the 1990s with some reference to later developments until July 2011.

The tax shift idea drove Sweden towards the systematic use of pollution taxes

In the early 1990s, the interests of several stakeholders converged opening a window of opportunity for the use of environmentally related taxes. The idea of an 'environmental tax reform' advertised by the OECD and some economists as good for the environment and the public budget sounded attractive regarding the revenue concerns and environmental problems faced by the government. In Sweden, there had already been some background work on the topic in the late 1970s (SOU 1978:43).

The tax shift idea dates back to the late 1980s (though the denomination only came later, Act 1991/92:100, Part I:5, 5, and Part 15, 7). In 1988, the Parliament decided to increase the use of economic instruments within environmental policy and an Environmental Charge Commission was appointed to analyse the scope for a tax shift underpinned by a double dividend argument (NCM, 1991: 33).

The Commission due to analyse the effects of existent pollution taxes and the conditions for and effects of different kinds of revenue neutral tax reforms (Brännlund and Kriström, 1999a: 240) produced several reports (interim reports SOU 1989:21 and SOU 1989:83, and a final report SOU 1990: 59) and proposed the tax shift from labour into pollution and energy. Several of the proposals were accepted and imposed as part of the 1990-91 major tax reform, which shifted approximately 1.9% of total tax revenue and 6% of the gross domestic product (hereafter also GDP) (SMF, 1991: 13; Lundqvist, 1996b: 262; Speck and Ekins, 2002: 96).

In the 1994 debate over the future role of pollution taxes, a more conservative approach was suggested, due to "the danger of unforeseen fiscal drain" (Swedish Commission Directive 1994:11). But in 2001 an even more challenging fiscal programme commenced (30 billion SEK (around 3.3 billion EUR) over a ten year period) (Nordic Council, 2006: 191).

From the start different fiscal expectations were associated with each of the instruments (SOU 1990:59). The government perceived the CO_2 tax as a permanent source of income to enable such shift, whilst other environmentally related taxes were seen as temporary sources of revenue (ibidem). The NO_x charge and the gasoline tax differentiation were not expected to provide any revenue (ibidem).

In contrast to what occurred with the sulphur tax and the NO_x tax, the tax shift idea influenced strongly the design of the energy tax and, though to a less extent, the CO_2 tax, both of which were important elements of the 1990-91 tax shift. Both tax designs were theoretically underpinned by the double dividend argument and biased towards a revenue-oriented rationale linked to tax shift motivations with the agreement of the Ministry of Environment.

1. THE COMPONENTS OF THE SWEDISH ENERGY TAXATION SYSTEM

The Swedish energy taxation system must be considered as a whole. The *effective tax burden* on energy followed from the overlap of several taxes raised on energy consumption, namely the general energy tax, the CO_2 tax and the sulphur tax, and one tax raised on specific polluting emissions following from energy production, namely the NO_x charge, plus VAT. The environmental effects following from energy taxation experienced in Sweden along the 1990s resulted from the *effective pollution price* communicated by the four taxes jointly considered.

In the mid 1990s, the structure of fossil fuel consumption in Sweden was the following: transport sector (40%), industry and the housing and service sector (15% each), heat generation (8%) and electricity (5%) (SGTC, 1997: 91). The general principle was that an energy tax, a CO_2 tax and a sulphur tax were charged on fossil fuels used for heating and as vehicle fuels (ibidem). Additionally, the energy tax was charged on electricity consumption depending on the type of activity (industrial or not), where in the country the consumption took place (ibidem) and the size of the consumer (from 1998 onwards, a higher rate was charged on large consumers, Nordic Council, 2006: 195).

Since the strongest environmental effects following from energy-related taxes are expected in the industry, the next sections focus mainly on industrial energy consumption and production. Therefore, low attention is provided to the taxation of electricity. In 1991, the tax on electricity consumption in industrial facilities was abandoned being reintroduced only in 2004 (by then companies could still opt for a zero tax rate by participating in a voluntary programme to improve energy efficiency and by taking actions to reduce electricity consumption) (Nordic Council, 2006: 195).

The traditional heavy reliance of Swedish industrial exports on large metal-based energy-intensive industries explains why energy management has been associated with industry regulation. The energy policy agenda has been ruled by concerns regarding competitiveness, economic efficiency, flexibility and environmental sustainability (Fischer and Berglund, 1994: 311). A potential conflict between environmental goals and economic growth unsettled by ecological modernisation theories has been evident in the especially favourable treatment assigned to energy-intensive industry. [41] However, the heavy reliance of the Swedish electricity production on hydropower and nuclear power allowed high tax burdens on fossil fuels. Non-transport energy tax rates increased between 50 and 60 percent during the period 2001-2005 (Nordic Council, 2006: 194).

1.1 the general energy tax and the CO2 tax on fuels

1.1.1 The general energy tax on fuels

A. The energy tax design was motivated by different political concerns

The energy tax design was never led by environmental objectives. Its motivation evolved over time and included simultaneously different political concerns (up until the 1970s exclusively revenue concerns, following post-Second World War signs of energy shortage expanded to include energy policy objectives, such as account of external effects and stimulation of energy efficiency). However, its impact was always predominantly fiscal (SGTC, 1997: 183; SOU 2003:38, 41), being an important source of revenue for the Swedish government (Sjölin and Wadeskog, 2000: 23-24).

[41] This concern was publicly addressed by the leader of the Social Democratic Party in 1996 (*Swedish Environmental Quality Objectives - A summary of the Government Bill 1997/98:145*). Dan Lundqvists, Political Adviser for the Social Democratic Party's parliamentary group on tax issues, phone interview, 8 September 2003.

The energy tax has pursued behavioural steering away from traditional (imported) energy sources (oil) allowing the development of new (endogenous) ones (hydro and nuclear) to support industrial growth since 1974, when the special tax on mineral oils and coal was incorporated into it starting a tax rate differentiation (ibidem). In a national economy strongly based on exports from heavy energy-intensive industry this rationale overlapped with the environmental interest in replacing dirty fossil fuels with renewable energy sources. In 1991, environmental concerns were officially introduced in the tax design with tax rate differentiation according to environmental classes. In 1993 the previous excise duty on energy consumption was renamed and became the ‘energy tax’ (Nordic Council, 2006: 194).

B. The energy tax payments were in general environmentally unrelated

The energy tax was levied on all fossil fuels and electricity in a specific amount per weight or volume unit. The formal charging criterion endorsed (and official criterion to assess the instrument’s efficiency, Act 1998/99: 100, Appendix 4), namely proportion to energy content, was not strictly followed (from 2011 onwards this criterion has applied to heating fuels, Government Bills 2008/09: 162 and 2009/10:41, Hammar and Akerfeldt, 2011: 12). The energy tax rates varied according to environmentally unrelated criteria (mainly usage (except for petrol) and sector of use) without consideration for relative polluting impacts or pollution abatement costs. The exceptions were the tax differentiations according to lead content and environmental classes.

Following competitiveness concerns, coal did not have the highest total level of taxation, whilst motor fuels faced an extra high charge. The highest tax rates were on gas oil and heavy fuel oil for heating purposes. Natural gas and LPG were charged with a reduced tax rate. The lower ranking of the latter two fuels had a twofold motivation: they entailed a relatively lower polluting potential and, as emergent markets, needed additional support, summing up to the ones they indirectly enjoyed through the CO_2 tax and sulphur tax.

Apart from heavy metals emissions, the external impact of bio-fuels only differs from the one caused by fossil fuel in terms of CO_2 and sulphur emissions. Hence, the differentiation between bio-fuels and fossil fuels should be restricted to the CO_2 tax and the sulphur tax (SGTC, 1997: 181). However, these fuels were exempted from the energy tax.

The tax rate differentiation according to lead content of gasoline (1986-1991)

The tax differentiation leaded/unleaded gasoline was imposed in 1986 in connection with the decision to impose stricter emissions criteria on new vehicles. It was triggered by the need to make attractive for refineries to produce sufficient amounts of unleaded petrol and avoid mis-fueling of leaded gasoline in cars with catalytic converters thereby securing the performance of the emission reduction technology (positive regulation externality of stricter emission requirements, Hammar and Löfgren, 2000: 2-3).

The tax reduction on unleaded gasoline was not based on estimations of marginal damage or marginal costs of lead (as required to set the prices right) but on estimates of the marginal cost of the supply side (i.e. abatement costs, to compensate refineries for extra production costs) (Hammar and Löfgren, 2000: 9). It was set higher than the extra production cost in order to accelerate the substitution effect (ibidem). The price differential was increased as phase-out approached completion (Act 1986/87: 139; Hammar and Löfgren, 2000: 6-7). Taxpayers showed relative capacity to avoid pollution by changing into vehicles equipped with new technology following the GDP growth experienced in Sweden along the 1980s.

Tax rate differentiation according to environmental classes (1991-)

In January 1991, the energy tax on diesel and heating gas oil, including heavy fuel oil, was differentiated according to three environmental classes to stimulate the introduction of motor fuels with superior environmental properties as a way to improve the quality of oil used as motor fuel, especially in cities (Act 1989/90: 111). The criteria were the content of sulphur and VOC and the boiling temperature of the oil. The calculation of the tax rebate for greener oil products (environmental classes 1 and 2) took into consideration that the energy tax on standard oil (environmental class 3) should remain unchanged (exclusive of CO_2 tax).

Initially the level of the tax rebate was based on the abatement costs necessary to attain the broad environmental objective of stimulating clean motor fuels, by approximating extra refinery costs to produce cleaner gas oil and diesel (SGTC, 1997: 35). In 1992, the rebate was increased and the criteria sharpened following the desire to further promote the introduction of cleaner fuels (SGTC 1997:34-35). Since October 1993, the energy tax differentiation followed an iterative method ('standard pricing approach', Baumol and Oates, 1971: 43-46, 1982: 163-164) aiming at specific behavioural change.

1.1.2 The CO2 tax on fuels

A. The CO2 tax had mixed objectives

The CO_2 tax was introduced in Sweden in 1991 following the recommendations of the 1987 Commission on Environmental Charges, on whose discussions the idea of a special CO_2 tax originated (Per Kågeson, member of the SOU 1989:83, e-mail communication, 10 September 2003). It mixed broad environmental objectives with non-environmental objectives. Though it was the most important pollution tax adopted following the Environmental Tax Commission (SOU 1990:59), both in terms of potential environmental impact and revenue, this tax was from the beginning "very much seen as a means of increasing state revenue", being part of the central budget (SEPA, 1995: 3, 1997a: 42). And a limited impact of the tax on behaviours in the short term was predicted by the Environmental Tax Commission following its rationale and design (SOU 1990:59).

The Swedish CO_2 tax aimed at improving the price competitiveness of less polluting fuels and sustaining the transition to renewable sources, without weakening the competitiveness of the Swedish industry. Since a CO_2 tax creates a price for fossil CO_2 emissions irrespective of the kind of fuel, leading consumers towards less CO_2 intensive energy sources, this tax was considered the primary instrument for Sweden to reduce fossil fuel consumption and thus the CO_2 emissions from sectors outside the EU Emission Trading Scheme (hereafter also ETS) (Hammar and Akerfeldt, 2011: 3). However, the initially discussed target of CO_2 emissions at the 1988 level was soon abolished even before the adoption of the tax.

B. The CO2 tax base was a good proxy for specific polluting emissions

The CO_2 tax was constructed as an input or consumption tax on energy levied on the CO_2 content of measured units of fuel. It was raised on all fossil fuels under the General Energy Tax Act (oil, coal, natural gas, liquefied petroleum gas, petrol and aviation fuel in domestic traffic until January 1997). This tax corresponded well to an emission tax, since CO_2 emissions from combustion are directly related to the carbon content of fuel due to the inexistence of an economically feasible technique for separating off carbon dioxide from flue gases and storing it.

Bio-fuels and waste were exempted to steer fuel choices towards more environmentally positive options, as was peat, despite its heavily CO_2 loaded combustion and non-renewable nature, to protect national industry. Though simplicity in fuel selection advises steering fuel consumption at electricity production, fuel used for electricity production was exempted as it was decided to tax electricity consumption itself.

C. The CO2 tax level was based on compromises between environmental and non-environmental concerns

The CO_2 tax rates were not based on the broad environmental objectives of the CO_2 tax but on compromises between environmental and revenue (and national competitiveness) concerns.[42] The complexity and lack of consensus in estimating external costs regarding a global problem (Sjölin and Wadeskog, 2000: 40) also played a major role in the CO_2 tax rate set up process (Jan Berqvist, chairman of the SOU 1990:59 and member and chairman of the 1997:11, phone interview, 15 September 2003).

[42] All the members of the commissions interviewed, namely, Kriström, Branndlung and Hjalmarsson, supported this statement.

Despite several environmental cost estimates developed by the Swedish Environmental Protection Agency (hereafter also SEPA) (Lindmark, 1998: 141) and the stated incentive intent (Budget Bill 1991/92: 100, Annex I: 1.5, 11), the CO_2 tax level followed from political decisions (Maria Gårding Wärnberg, Secretary of the SOU 1989:83, phone interview, 15 September 2003). The targets used both at the tax inception and in the 1996 tax rise were the obtainment of enough revenue to reduce the labour tax burden in a certain amount (tax shift) (Bengt Kriström, interview, 22 August 2003, Umeå). During the 1990s, this tax was the second largest source of revenue among pollution taxes (the first was the energy tax), with steadily increasing revenue (Sjölin and Wadeskog, 2000: 28).

To secure the value of the tax levels, maintaining the pursued steering effect and level of tax revenues, since 1994 tax rates have been adjusted yearly based on the Consumer Price Index (Hammar and Akerfeldt, 2011: 7). In January 2006 the rates corresponded to approximately SEK 0,92 per kilogram carbon dioxide released, and in January 2011 they had increased to SEK 1,05 (Hammar and Akerfeldt, 2011: 4).

CO_2 tax rates mirrored relative polluting impacts of fuels since they were set proportional to carbon-content of fuels based on their average CO_2 content (Swedish Government Bill 1989/90: 111, 147-154; Hammar and Akerfeldt, 2011: 7). Such rates varied not only between different types of fuel but also between different qualities of fuel and applied regardless of whether it was used for heating purposes or as motor fuel (EEA, 1996: 53). CO_2 tax increases applied uniformly across fuels in the sense that they were neutral between fossil fuels, not discriminating or favouring any fuel (Hammar and Akerfeldt, 2011: 7). However, since CO_2 tax rates failed to compensate for reductions in the energy tax, they were unable to guarantee that relative total tax burdens per fuel (*effective fuel taxes*) mirrored relative polluting impacts of fuels taking into consideration their CO_2 content.

The absolute level at which the CO_2 tax rates were set was neither based on the external costs associated with CO_2 emissions nor usually on the level required to induce the behavioural change (i.e. fuel switch or reduced consumption) necessary to fulfil any specific environmental target, including the international compromise assumed by Sweden in the Kyoto Protocol (Henrik Hammar, academic expert, interview, 21 August 2003, Gothenburg). For example, in 1995, to finance the annual EU membership fee and unconnected to any ecological benchmark (Määttä, 1997: 312), the CO_2 tax was increased (Act 1994/95:203, 2(1), Section 14(3)) (Frederik Lundberg, Green Party, phone interview, 1 September 2003). Those levels were unable to induce behavioural change in most industrial sectors since they did not compensate for reduced energy tax levels.

1.1.3 The effective tax on energy following from the energy tax and the CO2 tax

This section analyses simultaneously the incidence of the energy tax and of the CO_2 tax since these two taxes are seen as two components of a system rather than as two separate taxes. The interaction between the several components of the energy taxation system was the strongest in the case of the energy tax and the CO_2 tax, since the tax bases of these two taxes completely overlapped, whereas the sulphur tax base only partially overlapped with the energy tax base. Following the pressure of the Green Party, in the 1990s, the energy tax was 'transformed' into a CO_2 tax in the sense that increases in the CO_2 tax on fossil fuels coupled with reductions in the energy tax on these fuels both in time and amount of budget revenues (Lennart Hjalmarsson, member of the SOU 1991:90, interview, 21 August 2003, Gothenburg). The CO_2 tax constituted the most significant part of the excise duties levied on energy (more than three-quarters of the total tax on fossil fuel consumption in 2005, Nordic Council, 2006: 192).

From the late 1980s, the tax burden on fuels increased (mainly as a consequence of the VAT expansion to also cover energy consumption and the introduction of the CO_2 tax) and was differentiated according to the environmental impact of energy sources as well as economic sectors. From 1993 onwards, the bulk of this tax burden laid on households and commerce. Following the more favourable tax regimes applied to some sectors, the hierarchy of consumptions communicated by the energy taxation system was not the same in all the sectors.

Following the joint effect of the CO_2 tax and the energy tax, there was a varying net tax increase for all the fuels. The 1991 tax reform did not change the ranking of fuels in terms of tax rates but increased the tax differential (Table 4.1). The highest tax rates kept on being on gas oil and heavy fuel oil for heating purposes, followed by coal, LPG and natural gas. Mineral oils experienced the highest increase. Whilst taxation of coal was heavily increased, the most commonly used fossil fuel (heating gas oil) had its tax burden almost unaffected by the CO_2 tax introduction (Hammar and Akerfeldt, 2011: 7). Natural gas and LPG experienced relevant tax increases due to their previous low taxation (SGTC, 1997: 33).

Traditionally there was a lower CO_2 tax on fuels used for heat production in combined heat and power plants and heating purposes in energy-intensive manufacturing sectors (to improve their competitiveness and simplify the tax system by limiting the use of tax relief and in stationary motors) (Hammar and Akerfeldt, 2011: 4). The general tax level was applied to motor fuels in vehicles (Hammar and Akerfeldt, 2011: 4).

Table 4.1 Effects of the 1990-91 tax reform (except sulphur tax and VAT), 1/100 SEK per kWh

Fuel	1989 energy tax	1991 energy tax	1991 CO_2 tax	1991 Total	Difference
Heavy fuel oil	10.1	5.1	6.7	11.8	+17
Coal	6.1	3.0	8.3	11.3	+85
Natural gas	3.2	1.6	5.0	6.6	+103
LPG, motor fuels	1.4	13.3	6.3	19.6	+40
LPG, others	1.6	0.8	5.9	6.7	+307
Petrol, unleaded	30.3	27.5	6.7	34.2	+13
Petrol, leaded	32.6	30.3	6.7	36.9	+13
Electricity, industry	7.0	5.0	0	5.0	-29
Electricity, households	9.2	7.2	0	7.2	-29

Source: SGTC, 1997: 33.

Following the joint effect of the CO_2 tax and the energy tax, there was only a nominal tax increase for all the sectors (by the beginning of the 2000s, the general tax level was 2.5 times higher than the outset-level, Jonsson *et al*, 1997: 6, 26, 40). Over time most industry became exempted from the general energy tax (in 2005 manufacturing industry did not pay this tax) and was subject to the CO_2 tax to different degrees (Nordic Council, 2006: 195). General CO_2 tax exceptions applied to the industrial consumption of some fuels. On top of those, agriculture, forestry, and fisheries, energy-intensive industries received further tax reductions depending on the CO_2 tax burden as percentage of the value of sales (Nordic Council, 2006: 195; 1.7% in 1993, Bergman, 1996: 332, 1.2% in 1995, Sjölin and Wadeskog, 2000: 28, and 0.8% in 1997, Government Bill 1994: 1776, 9(9)).

More favourable regimes kept for some sectors meant that these hence faced lower than average *effective CO_2 emissions price*s. In 1991, at the inception of the CO_2 tax the *effective price* of CO_2 emissions was substantially increased. However, in 1993 reductions in the energy tax by 50% (except on petrol) partially annulled the CO_2 tax in some sectors (Committee on Industrial Energy Taxation, SOU 1991:90). Firms falling under the provisions introduced then in fact had a zero marginal tax rate and, thus, no incentive to reduce CO_2 emissions and energy consumption (Määttä, 1997: 85). For these companies the CO_2 tax was equivalent to an additional corporate income tax based on the share of the use of energy products (SOU 1991:90, 12). Following the 1993 changes, the *effective tax* on oil for industry was in the mid 1990s lower than before the CO_2 tax was introduced (SEPA, 1997a: 47).

Subsequent revisions increased the CO_2 tax burden on industry by limiting the number of companies benefiting from the ceiling provisions and the fuels covered by tax reductions (from 1995, only natural gas and coal, Energy Tax Act (SFS) 1994:1776, SGTC 1997: 42). In 1997 (Act 1996/97:29, Sections 2-4), the CO_2 tax rate on industry was doubled (changing from 25% to 50% of the general level), though, and despite the annual adjustment for inflation, it was still not as high as back in 1991 (Andersen *et al*, 2001: 56). Energy-intensive industries were still eligible for refunds, but marginal taxation was no longer allowed to be zero. Still, in 2006, manufacturing industry paid only 21% of the general rates of CO_2 tax (Susanne Akerfeldt, Swedish Ministry of Finance, e-mail communication, 3 May 2007).

In 2011, when the process initiated in 2008 towards exempting from the CO_2 tax energy consumption by sectors covered by the EU ETS finalised, lower than average CO_2 tax burdens had been shifted mainly to those sectors. By then the tax design was changed towards a more uniform national price on fossil CO_2 with less exemptions and reductions (Government Bills 2008/09:162 and 2009/10:41) (Hammar and Akerfeldt, 2011: 4). Within the sectors covered by the EU ETS, industry did not pay any CO_2 tax and the CO_2 tax for heat production in CHP plants and other plants amounts to 7% and 94% of the general CO_2 tax, respectively (ibidem). The lower level of the CO_2 tax applied to heating fuels used by industry, agriculture, forestry and piscicultural works as well as for heat production in combined heat and power plants not covered by the EU ETS was raised to 30% of the general level (ibidem).

The allocation of the effective tax burden on energy following from the energy tax and the CO2 tax: The best-payer criterion

the 1990-91 tax reform operated a shift in the energy tax burden from the manufacturing industry towards households and the tertiary sector (see Table 4.2). These took a burden four times larger than the manufacturing industry (Daugbjerg and Pedersen, 2002). Generally, during the first half of the 1990s, following concerns over national industry competitiveness, the total energy tax burden on industrial users was sharply reduced (total energy tax burden on industry was the same in 2000 as in the 1980s), through a decrease in the CO_2 tax and complete abolishment of the general energy tax, whilst on domestic energy users and other sectors it was correspondingly increased (Sterner, 1994a: 22; Svensson, 1997: 40) through a rise in the CO_2 tax (EEA, 1996: 53; OECD, 1996a: 58). Between 2001 and 2005 the actual CO_2 tax rate paid by industry was relatively constant in real terms. To keep the constant CO_2 tax burden on industry the slight increases in nominal tax rates were countered by a lower share to be paid (Nordic Council, 2006: 195).

Households took the main burden of the 1991 increase in the taxation levied on fossil fuel consumption (equivalent to a 40% increase of excise duties on petrol, SGTC, 1997: 33), as the industry was able to deduct VAT and benefited from excemptions or reduced rates regarding the energy tax. The CO_2 tax was paid mainly by households and over-reflected their marginal damage, as the energy-intensive industry benefited from lower rates. Despite the progressive increase in the share of energy tax paid by the industry, the disproportion between tax payments and fuel consumption across sectors stayed along the 1990s (Sjölin and Wadeskog, 2000: 26).

In 1993 and 1995, manufacturing and mining/quarrying consumed over 20% of total energy but paid for 3 and 0% of total energy tax payments, respectively (Sjölin and Wadeskog, 2000: 24-25). Transport companies also paid a disproportionate tax share due to various tax subsidies (ibidem). The share of energy tax paid by private consumers fell from 71% in 1993 to 65% in 1995 (ibidem). But it was still higher than their share of fuel consumption (circa 32% in 1995) (ibidem). The public sector followed the same pattern. The percentage of energy tax paid by other industries was slightly higher than their corresponding share of total energy consumption (Sjölin and Wadeskog, 2000: 27). This disproportion was also observed regarding electricity. In 1995, manufacturing accounted for 37% of total electricity use but was exempted from the energy tax (ibidem). Private households were responsible for 33% of all electricity consumed, while paying 56% of the total energy tax on electricity (ibidem).

In the CO_2 tax, likewise in the energy tax, tax payments were not proportional to the emissions produced per sector. Manufacturing was responsible for 28% of all emissions but paid only 7% of the tax, reflecting a 75% (50% in 1996) tax abatement (Sjölin and Wadeskog, 2000: 29). Transport and communication industries also paid less than they should according to their emissions (ibidem). Private consumers, on the other hand, as major consumers of motor fuels on which the tax was fully charged regardless of use, paid the largest share of the tax (about 47%) but caused only 25% of total emissions (ibidem).

Table 4.2 Actual CO_2 Tax Rates in Sweden (in SEK/t CO_2)

	1991	1993	1996	1997
Industry and horticulture	250	80	92.5	185
Households, commercial, and motor fuels	250	320	370	370

Source: OECD, 1996d: 59.

1.2 The Sulphur Tax on Fuels

1.2.1 The sulphur tax design was led by environmental criteria and quantified environmental objectives

Sulphur emissions have been a national concern in Sweden since the 1970s. In the early 1990s, control of NO_x and sulphur emissions were national environmental policy targets, since the main regional environmental problems, like eutrophication, acidification, reduction of biodiversity, ground ozone layer and emissions of metal (SEPA, 2000: 3), were related to such emissions. In 1990, following some registries of NO_x emissions levels above the national limits (NCM, 1993: 13), it was decided to tax separately NO_x and sulphur emissions, though their sources coincided. In both cases domestic sources contributed only approximately to 10% of total deposit in 1985 (Act 1990/91:90, 23), therefore much less than foreign sources.

Following the adoption of demanding objectives regarding abatement in sulphur emissions and building on an idea put out by the 1978 Commission on the use of economic instruments for the environment (SOU 1978:43), a sulphur tax was introduced in 1991. Though the Tax Agency (Ministry of Finance) kept its administration, the SEPA (Ministry of Environment) was in charge of its design and referred it to the environmental problems addressed, namely acid rain and air quality.

Precise environmental objectives fixed on the basis of the carrying capacity of the environment were assigned to the tax. Sulphur deposits had to be decreased by about 75% of the 1980 level in the south-western part of Sweden to bring deposits below the critical load (ibidem). This was a complex goal following the aforementioned mainly foreign and diffuse nature of the sources of these emissions.

The tax aimed to stimulate reductions in SOx emissions above those achieved through previously existing regulations. The 1991 Environmental Policy Bill required energy and heating production in larger plants to decrease sulphur emissions between 1993 and 1997 to reach emission levels 40% lower than the obligatory ones at the time (Act 1990/91:90, 292) (Lundqvist, 1996b: 310). Major expected effects were the reduction of the sulphur content in light fuel oil below 0.1% and an 80% abatement in sulphur emissions (Brännlund and Kriström, 1999a: 238).

1.2.2. The sulphur tax was raised on a good proxy for specific polluting emissions

The sulphur tax was raised on the sulphur content of measured units of some fuels liable for energy and CO_2 taxes, namely coal and heavy fuel oil, plus peat, which were the fuel categories with the highest sulphur content. It applied to fuel containing 0.05% of weight of sulphur or higher at the time of combustion or sale (EEA, 1996: 52). This was a product tax equivalent to an emission charge on sulphur since sulphur emissions can be approximately measured through the sulphur content in fuel (NCM, 1991: 44).

The charge system graded according to sulphur content was in practice based on actual emissions due to refunds. Deductions were allowed for sulphur removed either by purification or binding into a product or ash (EEA, 1996: 52). Since the tax could be repaid by each unit of sulphur emissions reduced against proof of an effective lower level of emissions than the one resulting from indirect measurement and large plants usually contained measuring equipment, in practice these were taxed on their emissions.

1.2.3 The sulphur tax rate was referred to environmentally correct behavioural change

Since its introduction until 2011, its tax rate remained constant at 30 SEK/kg sulphur for solid fuels and 27 SEK/kg for each thousandth of sulphur content by weight in oils (Nordic Council, 2006: 193; Hammar and Akerfeldt, 2011: 4). The threshold in the sulphur taxation scheme had the result that fuels with a sulphur content not exceeding 0.05 percent in weight were tax exempted (ibidem).

The sulphur tax rate was based on abatement costs and set at a level sufficiently high to induce behavioural change. The Commission on Environmental Charges (SOU 1989:21) argued the tax should be at least equivalent to the sulphur premium (i.e. the price difference between oil with low and high sulphur content) to have the desired effect and calculated the premium at a higher value than the Swedish Petroleum Institute. The government accepted its proposal.

The success of the tax seems to have been associated with the right price signal provided. This approach was expected to allow the attainment of environmental goals at a faster pace and lower cost than the tightening of the standards (Olivastri and Williamson, 2001). Lower average sulphur content was hardly likely to be possible using available crude oil qualities without major investments at Swedish oil refineries (Jonsson *et al*, 1997: 28). The tax incentive was adequate to support such investment (Olivastri and Williamson, 2001). Furthermore, the tax provided a large incentive to adopt tax avoidance strategies corresponding to emissions abatement (ibidem). This incentive was increased following the drastic fall in sulphur dioxide abatement costs during the period 1980-1995, induced by closed pulp production processes, lower sulphur content of fuel (regulated in 1970) and increased use of various scrubbing techniques (Lindmark, 1998: 140, 147), without any revision of the tax rates until 2011 (Speck, 2008: 50; Hammar and Akerfeldt, 2011: 4).

1.2.4 The subjective incidence of the sulphur tax attended to the capacity to abate pollution

In the allocation of the sulphur tax burden tax payments were referred to the amount of emissions produced but not proportional to them. Manufacturing, responsible for 46% of the sulphur dioxide emissions, paid only 26% of the tax, due to the tax relief enjoyed by fuels used for fuel production (e.g. refineries) and industrial processes (Sjölin and Wadeskog, 2000: 31-32). And private consumers, whose tax payment was related to the use of sulphur-rich fuel oil intended for satisfying a basic need, namely heating, were responsible for a higher share of total emissions than tax payments (ibidem).

Industries where a larger substitution effect was expected, namely in electricity, gas and heating plants, paid the largest percentage of sulphur tax compared to the emissions produced (ibidem). In 1995, this was about 62% of the total tax burden, while causing approximately 18% of all emissions of sulphur dioxide (ibidem). It was clearly stated that to promote the use of low sulphur fuel in electricity generation, no exemption was granted to fuels used in power production (SGTC, 1997: 31).

The tax provided incentives for both the use of low-sulphur oils and investment in technology, as emissions were reduced both by more intensive use of flue gas cleaning and fossil fuel substitution (NCM, 1996: 48-49). Possible substitutes were light fuel oil and heavy fuel oil, natural gas, coal, and, the most important, electricity. In electricity, gas and heating plants energy costs represented a large share of total costs. Therefore, they had an incentive to invest in flexibility among energy sources, in particular between oil and electricity (Hammar and Löfgren, 2001: 115).

1.3. The Nitrogen Oxide (NOx) Charge

1.3.1 The NOx charge was underpinned by quantified environmental objectives

A precise environmental objective underpinned the proposal to create a NO_x charge, namely a least cost strategy to ascertain a reduction of NO_x emissions from targeted stationary sources by 30% until 1995 taking as reference 1980 (SOU 1989:83, 155), following the national importance of the environmental problem involved. Concerns with reaching the environmental objectives decided in a cost efficient way were evident in the setup of the tax rate and in the selection of the taxpayers. This might have been related to doubt about the causes of one of the main problems underpinning its adoption, namely eutrophication, whether nitrogen or phosphorus.

The NO_x charge was applied from January 1992 (Act 1990:613) and in 1996 this was still the only emission charge in the Nordic area (NCM, 1996: 50). Until then NO_x emissions tended to not be subject to charges. The incentive to reduce such emissions was provided rather through rebates in the excise duty on motor vehicles with catalytic converters installed (NCM, 1996: 31). The NO_x charge was used partly in parallel with the standards set in 1988 to accelerate the reduction of NO_x emissions (SEPA, 1997a: 31; Höglund, 2000:II-14).

1.3.2 The NOx charge was raised on specific polluting emissions

The NO_x charge was raised on (actual or presumptive) NO_x emission. If the company did not measure its emissions or did not use certified measurement equipment, a standard assessment was used. Tax subjects could also apply for a fixed charge rate. Presumptive emissions levels were substantially higher than actual emissions, therefore measurement was generally preferred (SEPA, 2000: 5). Therefore, the charge was usually based on actual measurements following continuous monitoring of the emissions.

Taxation of NO_x emissions requires direct measurement, since these emissions are produced during any kind of combustion and particularly contingent on technology standards and equipment maintenance. Their amount depends primarily on the temperature at which combustion takes place and the air-fuel mixture, being the nitrogen content of the fuel relevant to a lesser degree (SEPA, 1997a: 31). Therefore, to tackle this environmental problem it is more important to promote fuel efficiency and technological progress (scrubbing) concerning the combustion process than fuel quality.

1.3.3 The NOx charge level was referred to environmentally correct behavioural change

The charge level was based on abatement costs and environmental costs. The level of the tax rate was set taking as reference the abatement costs necessary to reach the environmental objectives decided (the amount below which the tax could not go without loss of behavioural steering impact) and the environmental costs following from NO_x pollution. These were the maximum ceiling for the tax rate level and corresponded to its actual level. The differential between the minimum and the maximum level accounted for the size of the incentive to abate provided by the tax.

The charge level was based on the average damage cost of NO_x emissions and pollution abatement costs following engineering data on expected effectiveness and costs of abatement investments intended for electricity production and district heating (SEPA, 2000: 4-5; Höglund, 2000: II-8). The report SOU 1989:83 contains a detailed technical discussion of the level at which the charge should be set to achieve the desired impact on behaviour (Jonsson *et al*, 1997: 15).

The charge level was set at SEK 40 per kilo of NO_x emitted (constant in nominal terms during the 1990s and raised to SEK 50 in 2007, Act 1990:613, paragraph 5, according to Act 2007: 1372). The government considered it reasonable not to exceed this cost level since it was understood to be the value the Swedish society attributed to the damage caused by such amounts of NO_x. This level was expected to allow the achievement of the environmental targets decided (SEPA, 1997a: 38), providing a strong incentive for tracing and applying cost-effective solutions, since costs of reducing emissions in the covered installations were estimated at an average figure of SEK 10 per kg (EEA, 1996: 57).

1.3.4 The subjective incidence of the NOx charge was based on capacity to abate

The NO_x charge selective tax incidence was based on technological criteria that influenced the polluters' capacity to abate emissions. Big companies were expected to reduce their NO_x emissions more successfully than small ones following their higher technological capacity to adopt continuous monitoring required for the taxation of NO_x emissions. Continuous monitoring was not realistic regarding small units, due to the high costs involved following the capital indivisibility in technological options and the frequent requirement for tailored solutions.

The choice of the size limit of the polluting sources on which to impose the tax was based on the criterion that measurement costs should be in reasonable proportion to the profit that the owner of the furnace could make by restricting emissions and thus reducing the charge payable (NCM, 1991: 36). In January 2010, only installations producing at least 25 MWh useful energy per year were covered (Act 1990:613, paragraph 3) (Per Kågeson, member of the SOU 1989:83, e-mail communication, 10 September 2003). Largest plants (> 50 MWh per year) produced over 90% of overall regulated output (OECD, 2010: 26).
The selection of the taxpayers also aimed at keeping high levels of tax awareness among the payers. Best results in emission intensity levels were expected in companies paying higher than average attention on energy production and its cost effectiveness. Therefore, only final energy producers were charged, with industrial process burning excluded (Act 1990:613, paragraph 2(1)).

The tax coverage was extended several times due to effectiveness in emissions reduction and simultaneous fall in metering costs (paragraph 2 of Act 1990:613 as amended by Acts 1994:1107, 2002: 411, 2007: 1372) (Höglund, 2000: II-7). However, the amount of emissions covered by the tax kept on being very small compared to total domestic emissions (less than 5% of these) and even smaller compared to total NO_x emissions deposits in Sweden (less than 1% of these) (SGTC, 1997: 38-39, Höglund, 2000:Introduction and Summary p. 6). In 2010, it covered circa 40% of NO_x emissions from stationary combustion sources in Sweden (OECD, 2010: 6).

1.3.5 Revenues were used to enhance the environmental effectiveness of the NOx charge

Related to the high technical expertise involved in the tax design, the SEPA (Ministry of Environment) was in charge of the tax design and management. It never kept revenue interests in the charge since revenues have always been recycled back to the group of taxpayers to avoid market distortion following from technology-induced selective tax incidence (Per Kågeson, member of the SOU 1989:83, e-mail communication, 10 September 2003; Maria Gårding Wärnberg, Secretary of the SOU 1989:83, phone interview, 15 September 2003). Otherwise taxpayers would enjoy an incentive to split into smaller companies (ibidem).

Revenues were recycled according to each one final production of useful energy (energy efficiency criterion), which made the measure financially neutral for the group as such, except for abatement and transaction costs (OECD, 2010: 6). This promoted competition among plants for attaining the lowest level of NO_x emissions per amount of useful energy produced within the regulated group (ibidem). Plants performing below the average were net payers and the ones above the average were net receivers.

This design allowed selective tax incidence, which made possible the strict requirement for continuous monitoring of emissions, a charge level high enough to attain significant effects on emissions and avoid strong political resistance among polluters (OECD, 2010: 6). The first two aspects were identified as being responsible for the high success of the Swedish instrument vis-à-vis its French counterpart (OECD, 2010: 10). The SEPA calculated and administered the net refund based on taxpayers information, thus the charge was never actually collected (Höglund, 2000: II-8, III-3).

2. EVALUATION AND CRITICAL ANALYSIS OF THE ENERGY TAXATION PACKAGE

Empirical data from the Swedish energy taxation package show environmental effects were more evident and quicker the more the tax design included the design features of environmental taxes. Those effects were clearer and faster following the NO_x charge and the sulphur tax and more unclear and only evident over a long time period in the energy tax and the CO_2 tax, though clearer in the CO_2 tax than in the energy tax. The performance in environmental terms and in revenue terms followed opposed directions in all the taxes.

The difference in environmental effectiveness might not have been sharper due to the annulment of the increases the CO_2 tax and the sulphur tax introduced in the *effective price* of CO_2 emissions and sulphur emissions by reductions in the energy tax. The annulment effect was even more evident in the case of the CO_2 tax, since following the complete overlap between the tax bases of these two taxes, the energy tax was used to compensate in some industries the *effective pollution price* increase fossil fuels experienced following the introduction of the CO_2 tax.

In the Swedish energy tax system, energy products were simultaneously regulated by instruments providing different price signals following their different rationales, namely a regulatory one (the sulphur tax) and a fiscal one (the energy tax and the CO_2 tax). This made it difficult to use *effective pollution prices* to divert consumption away from fossil fuels. The failure to refer the CO_2 tax and the sulphur tax to environmentally correct relative *effective pollution prices* kept them from providing a correct price signal towards the environmental hierarchy of behaviours compensating the reductions in the energy tax. These had a especially negative impact on the environmental effectiveness of the CO_2 tax and the sulphur tax given the overlap of their tax bases.

2.1 Main environmental effects following each tax

The energy tax design did not include the design features of environmental taxes, except in the tax differentiations according to lead content and environmental classes, being, from 1990 onwards, theoretically grounded in the double dividend argument, supporting the Swedish environmental tax shift. The tax differentiations targeted price signals to specific polluting impacts, namely lead in one case and sulphur and VOC content in the other. In both cases tax rates were referred to abatement costs and set at a level sufficiently high to induce environmentally correct behavioural change and taxpayers enjoyed capacity to avoid pollution.

Accordingly, the regulatory design of these regimes and their positive impact on behaviours (and consequent fast decreasing capacity to raise revenues, MENS, 1994) contrasted with the fiscal design of the remaining energy tax regime and its high capacity to raise revenue. Except for the mentioned differentiations, the energy tax placed a blunt price on polluting behaviour. Therefore, the positive environmental effects following from this tax were mainly in energy efficiency and only evident over a long period of time. The tax differentiations were very effective in leading fast behavioural change towards cleaner fuels.

Though the CO_2 tax had an environmental objective, this was a relatively broad one and it was mixed with non-environmental objectives. Therefore, it failed to provide guidance to design all the features of the CO_2 tax. This tax included only part of the design features of environmental taxes, mainly the ones regarding the use of environmental criteria and the choice of the tax base. The more targeted design of the CO_2 tax compared to the general energy tax, whose payment was unable to mirror any particular polluting impact following from energy consumption, explains its better performance in pollution reduction (by shifting consumption towards less CO_2-loaded fuels). Still its environmental performance was in general moderate and only evident in the long-term. The 1997 SEPA report considered the state's fiscal ambitions with the CO_2 tax had been fulfilled as, in the short and medium term, the tax base could only change slowly because of fossil fuels' low price elasticities (Andersen *et al*, 2001: 59-60).

In the sector where the design features of environmental taxes were more evident in the CO_2 tax design, namely the district heating sector, environmental effects following from the introduction of the tax were high and fast. In this sector tax payments correctly signalling the environmental hierarchy of fuels, in an amount able to lead behaviours and imposed on polluters able to prevent pollution led to strong fuel switch away from coal and towards biomass.

The sulphur tax design included most design features of environmental taxes and induced technological progress towards sulphur abatement both on the demand side and the supply side. Due to its environmental success, this tax followed a downwards path in terms of revenue raising capacity (Sjölin, 2001: 2). In the period 1991-1993, the average result was between SEK 200 and 250 million per year (against the initially estimated SEK 500 million, Act 1989/90:110), as a consequence of the substantial emission reduction prompted by the tax. However, after this first strong impact on behaviour, revenues stabilised fairly quickly (Jonsson *et al*, 1997: 8).

The NO_x charge included most design features of environmental taxes and caused a strong and quick environmentally positive behavioural change (increased efficiency in NO_x emissions abatement and technological progress). NO_x emissions reduced substantially (37% – NCM, 1996: 50; 50% – SEPA, 1997a: 34, 2000: 10) shortly after the introduction of the charge, between 1990 and 1992. The positive environmental effect, noticeable from the introduction of the charge, was biggest in the beginning of the period, but improvements kept on being observed until 1998. Between 1980 and 1995, the reduction in NO_x emissions (approximately 20%) was rather modest when compared to the reduction of sulphur emissions (approximately 80%). However, it was considered significant following the technical complexity involved in the control of this pollutant due to its complex formation process (OECD, 2010: 36). The NO_x charge has never produced revenues since these have always been fully recycled back to the paying sector.

The general energy tax

Taking as reference the 1990s, the environmental effects following the energy tax were not as evident as those associated with other taxes part of the Swedish energy tax system. Results obtained in terms of switching to cleaner fuels, which were mainly caused by the CO_2 tax and the sulphur tax, were stronger than the gains experienced in energy efficiency, which were the only ones due to the energy tax. CO_2 and SO_x emissions reductions due to lower energy consumption induced by the energy tax followed a longer time-series than the same kind of progresses due to fuel substitution induced by more targeted instruments such as the CO_2 tax and the SO_x tax (SGTC, 1997:90).

The composition of Swedish energy supply has changed considerably since 1970, but the energy supply has been stable since then except for a modest effect on petrol consumption (Hanisch, 1998: 540 A-544 A) and a slight decrease in total industrial energy use. The steady decrease in the Swedish manufacturing industry's oil intensity in terms of production, which might be attributable to the energy tax, is only evident over a long period of time (SGTC, 1997: 90). However, there was a strong decrease in energy content per unit produced and energy consumption per worker (ibidem).

a. Tax differentiation leaded/unleaded gasoline

Apparently the tax differential had an important role in the phase-out of lead gasoline. However, the rapid growth in the unleaded petrol market share since 1986 resulted from the combined effect of both command-and-control and economic instruments. The specific policies addressing lead together with the high registration rates of new cars equipped with catalytic converters driven by GDP growth contributed to explain the reduction of lead emissions from the Swedish transport sector (Hammar and Löfgren, 2000: 17).

The substitution effect was faster than planned either due to cost miscalculation following information asymmetry between the government and industry or the adoption of a low reference point (i.e. the unleaded fuel market share in 1986). After the rapid unleaded petrol market share increase between 1986 and 1989, the market stabilized due to the large share of passenger cars without catalytic converters and therefore unable to use unleaded petrol. In 1992, spurred by the tax differentiation that made it profitable to develop unleaded petrol, there was technological innovation allowing its consumption to expand and eliminating leaded petrol from the market (decreased from 100% in 1986 via 40% in 1992 to practically zero in 1993, and was legally banned in March 1995) (Hammar and Löfgren, 2000: 14). Between 1988 and 1993, the total amount of lead emissions dropped by about 80%. (EEA, 1996: 55).

b. Tax differentiation according to environmental classes

The positive environmental impact following from the energy tax differentiation according to environmental classes contrasted with its poor fiscal performance and was the cause of the latter. In 1990, before its introduction, less than 1% of the diesel sold in Sweden was of the 'clean' type. In 1994, 60% of all diesel fuels complied with class 2 and 15% with class 1 requirements (EEA, 1996: 55). In 1996, almost all diesel fuel used in the transport sector was of the cleanest type. The use of cleaner fuels has reduced sulphur emissions from diesel vehicles by 75% on average (OECD, 1997c: 26), and even by 95% in cities (EEA, 1996: 55).

The unpredicted fast evolution in clean fuel use represented forgone tax revenue (in 1992, SEK 125 million and, in 1995, SEK 506 million) (SGTC, 1997: 36). This effect was expected to increase further with the rising share of class 1 diesel. The rapid shift towards clean fuel was due to technical innovation spurred by the tax rebate, as it compensated oil companies more than enough for their extra production costs, which were lower than expected (EEA, 1996: 55-56). The tax differentiation made it more profitable to sell heating oil belonging to classes 1 and 2 than to class 3 (ibidem). Apparently, rebates included, production costs of classes 1 and 2 were lower than production costs of standard fuel (ibidem).

The CO_2 tax

It is difficult to demonstrate any significant impact of the CO_2 tax on absolute CO_2 emissions (Fischer and Berglund, 1994: 316). Such an impact on the emissions from the Swedish industry was rather small in the short run. However, over a longer time span those effects are noticeable. Based on empirical evidence drawn from the Swedish case, the 1997 SGTC concluded that, from an environmental point of view, taxes such as the CO_2 tax do have the intended effect on domestic emissions (SGTC, 1997: 110).

According to the 1997 SEPA assessment, there was a direct effect of the CO_2 tax on market prices and probably also an indirect effect, as the tax increased awareness about environmental problems associated with burning fossil fuels (Andersen *et al*, 2001: 59-60). Domestic CO_2 emissions were estimated to be by the end of the 1990s approximately 3-4% lower because of the tax than they would have been in a business-as-usual scenario but in absolute quantities there was a minor rise (Andersen *et al*, 2001: 93).

There was considerable substitution from fossil fuels towards electricity (Brännlund and Kriström, 1999b: 12), as the use of fossil fuels decreased during the period 1974-1993 despite a significant increase in production. Between 1990 and 2007, the CO_2 equivalent emissions were reduced by 9% while at the same time Sweden experienced an economic growth of 51%. (Hammar and Akerfeldt, 2011: 11). In 2007, the Swedish 35% of gross inland fossil fuel consumption of energy in all sectors of society (in the EU-27 this average was 77%, European Commission, 2010) placed the country among the lowest CO_2 emitters (per capita) within the EU, with 6.7 tonnes per inhabitant compared with the EU-27 average of 9.1 tonnes (Hammar and Akerfeldt, 2011: 3).

The sulphur tax

The effect pursued by the sulphur tax was attained (OECD, 1997a: 48-49). And *ex post* analysis identified the tax as responsible for a 80% reduction of sulphur emissions by the end of the 1990s compared to 1980 levels and for a 30% of the total reduction in these emissions from 1989 to 1995 (OECD, 2001g: 106).

Table 4.3 Emissions of sulphur from domestic sources in Sweden (1980, 1990, 1994), 1.000 tonnes

Source of emissions	1980	1990	1994
Road traffic	11	8	2
Other communications	32	29	22
Combustion of oil and gas	318	34	22
Combustion of coal and coke	6	11	3
Combustion of other solid fuels	4	9	11
Industrial processes	137	43	36
Total	508	134	96

Source: SGTC, 1997: 31.

There was an almost 10% reduction of the sulphur content of fuel oils per year between 1990 and 1992 (Table 4.3) (Ds 1994:33). The sulphur content of light oils was in 1998 below 0.1% on average (well bellow the legal limit of 0.2%) (Table 4.4). Sulphur dioxide concentrations were generally well below the planning goals laid down by the SEPA (0.076%, which was less than half the legal limit of 0.2%), though still in excess of the critical load due to emissions from foreign sources (Brännlund and Kriström, 1999a: 234-235).

Table 4.4 Average sulphur content of fuel oil

Year	Light Fuel Oil (% sulphur by weight)	Heavy Fuel Oil (% sulphur by weight)
1976-1988	0.2	0.8
1989	0.2	0.7
1990	0.2	0.7
1991	0.08	0.5
1992	0.076	0.45
1993	0.056	0.45
1994	0.058	0.45
1995	0.076	0.35

Source: SGTC, 1997: 31.

The main effect of the sulphur tax might have been to reduce concentrations of sulphur in oil, but it has also affected the more efficient removal of sulphur during combustion (Jonsson *et al*, 1997: 6). This tax has primarily worked via three channels: 1) by inducing technological progress on the demand side, 2) by enhancing technological progress on the supply side, i.e. the reduction of sulphur content of oil in the refinery industry (Hammar and Löfgren, 2001: 119), and 3) by leading to substitution between heavy and light fuel oil, this being the least applicable of the three (Hammar and Löfgren, 2001: 107).

The NO_x charge

Ex post analysis identified the NO_x charge as responsible for approximately two thirds of the total reduction in NO_x emissions, which makes it a very successful instrument of environmental policy (EEA, 1996: 10; OECD, 1997a: 45-46; Barde, 1997: 243), the remaining reduction is likely to have followed from the standards set in the plant's operating permit condition under the Environmental Code that applied to many of the plants charged (SEPA, 1997a: 35). Emissions abatement was above the limits of quantitative standards for most plants (OECD, 2010: 22). In the absence of the charge, emissions from charged boilers in 1995 were due to have been much higher (80% higher – Jonsson *et al*, 1997: 6; 25% higher – Hammar *et al*, 2001: 55-56) than the actual amount.

Emissions in the charged plants have been decoupled from increases in energy production. Emissions have remained fairly constant in regulated plants, while between 1992 and 2007, energy output in regulated plants has increased by 77% (OECD, 2010: 6). There was a 20% emissions reduction per unit of useful energy produced between 1992-1999 (SEPA, 2000: 11).

The degree of success in the efforts taken to reduce emissions as well as the financial impact of the charge was different among industries, which shows the price effect of the charge. Sectors subject to stronger financial pressure following the charge were also more responsive to its price signal. Firms cleaner than average, which made a net profit, tended to worsen or only slightly improve their environmental performance; while the others, which made a net payment, showed a more positive evolution in their environmental impact (Table 4.5) (Hammar *et al*, 2001: 56).

The paper and pulp industry was a net payer while the power industry (heat and cogeneration) was a net receiver (Hammar *et al*, 2001: 56). The largest net payer was the waste incineration sector, which managed to bring about an emission reduction of 40%. The largest net receiver was the metal industry, whose emissions slightly increased (EEA, 1996: 57). All sectors, except the metal industry, have substantially reduced their emissions since the charge was introduced (SEPA, 2000: 11).

Between 1990 and 1995, overall, NO_x emissions recorded as emanating from combustion for energy generation decreased by approximately 20% and emissions from industrial processes, which were not included in the charge system, increased by about 17% (SEPA, 1997a: 36-37). Among the boilers in the charge system and for the same five year period, emissions per energy unit fell by about 60%, whereas total emissions fell by approximately 50%. The differential was explained by the fact that total energy production from these boilers increased (SEPA, 1997a: 35).

Table 4.5 NO_x **emissions: Analysis by sector**

	Number of installations	***Net payments per GWh of energy produced***	***NO_x emissions production 1992-1993 (%)***
Waste incineration	5	9763	42
Energy generation	53	(878)	23
Chemical industry	23	(94)	17
Pulp and paper industry	39	1304	13
Metal industry	2	(9168)	- 2
Total	122	176	20

Source: EEA, 1996: 57.

Inefficiencies in abatement were highlighted following the introduction of the charge, which put a positive price on pollution. Abatement activities were taken following the payoff for investment in emissions control it introduced. The strict requirement for continuous monitoring just brought another parameter into the optimization formula (OECD, 2010: 8). The fact that abatement measures taken represented different types of trimming activities that plants did continuously anyway allowed that about a third of emissions reductions had taken place at a zero or very low cost when comparing cumulative abatement costs for each plant with the attained emission reduction (OECD, 2010: 8). This supports the hypothesis that 'low-hanging fruit' was abundant in abatement activities (Höglund, 2000: II-36).

Technological progress (adoption and development) followed the introduction of the charge. The tax design based on competition for the lowest NO_x emissions per energy output produced and the requirement for continuous monitoring affected demand for several different technologies (OECD, 2010: 17). Innovation in mitigation technology made it possible to reach lower emission intensity levels for the same output level (OECD, 2010: 24). Between 1992 and 2007, most plants report adoption of some kind of mitigation technology. The rate of adoption was particularly high following the introduction of the charge in 1992 and its extension in 1996 and 1997 (OECD, 2010: 36). This might indicate the impact of the charge in the spread of technology or lobbying from innovating firms to get the government approving high charge levels (OECD, 2010: 34).

2.2 The inclusion of the environmental tax design features in the tax design

2.2.1 The use of environmental criteria and precise environmental objectives

The general energy tax never took into account environmental criteria. It was ruled by the economic logic of avoiding distortions while collecting revenue, only aiming at environmental objectives in the tax differentiations according to lead content and environmental classes, which accounted for a small part of this tax regime. Likewise, the CO_2 tax was never ruled by precise environmental objectives, though it made reference to environmental criteria in the selection of the CO_2 content in fuels as tax base and the setup of the nominal tax rates according to such content. Both taxes included a fiscal rationale based on the double dividend argument.

Unlike in the energy tax and the CO_2 tax, where the objectives pursued followed from political compromises based on revenue and national competitiveness concerns, the precise and quantified environmental objective pursued by the sulphur tax reflected a sustainability criterion whilst being fixed on the basis of the carrying capacity of the environment. The NO_x charge was also aimed at precise and quantified environmental objectives. However, following scientific uncertainty about the causal process underpinning one of the main problems it addressed (eutrophication), such objectives have not been related to the carrying capacity of the environment.

Also unlike the energy tax and the CO_2 tax, science was entangled in the sulphur tax and NO_x charge process leading to technically complex tax designs in the way the stimulus to pollution abatement was regulated, namely regarding the delimitation of the tax base with reference to actual emissions, which allowed tax payments to accurately mirror the environmental damage caused, the setup of the tax rate and the choice of the taxpayers.

2.2.2 The institution of tax awareness and tax avoidance in the tax design

The energy tax and the CO_2 tax did not have tax awareness instituted in their design. The financial loss their payment involved was passed to the polluters together with the energy price. Furthermore, the design of these taxes was unable to reward tax avoidance strategies. The sulphur tax, in contrast with the CO_2 tax and the energy tax, had tax awareness and avoidance instituted in its design by allowing and inducing tax payments according to actual emissions. Reimbursements following proved abatements created an incentive to install measuring equipment and emissions measurement raised tax awareness. This tax design, which allowed tax payments to accurately mirror the evolution of the environmental damage, stimulated the adoption of tax avoidance strategies providing an important incentive for the reduction of sulphur emissions by the manufacturing industry (Hammar and Löfgren, 2001: 107). By the mid 1990s, about a quarter of the tax subjects had taken measures to clean flue gases and were receiving tax refunds. The reduction of sulphur emissions was 70% on average (EEA, 1996: 53).

The payment of the NO_x charge usually according to continuously measured emissions (following the incentive provided by high levels of presumptive emissions to adopt measurement) not only rewarded tax avoidance strategies, but also raised awareness. The latter was further instituted in the NO_x charge design through the selection of the taxpayers. It covered only companies where, following the high attention dedicated to energy production and its cost effectiveness as well as the high financial loss involved in the tax payment, the charge was due to enjoy high visibility. Tax awareness was also instituted in the NO_x charge design by making clearer the linkage between the tax base and the tax goal for the taxpayer via recycling of the tax revenues back to the industry according to a criterion directly relevant for environmental damage production, namely efficiency in energy production.

In plants belonging to the energy sector, where tax awareness was due to be high, over-compliance with the NO_x charge was registered in 1996. These plants reduced their emission intensities by more than their cost-minimizing level ('forecast reduction', SEPA, 1997a: 36) following their public ownership (compliance with environmental objectives was added to the profit-maximising objective) and higher than average attention to energy-related issues since this was their core business. Reduction in marginal abatement costs, due to the revelation of low-hanging fruit following the introduction of the tax (Höglund, 2000: II-36), as well as some technological progress (indirect determination of NO_x emissions) and consequent reduction in monitoring costs (SEPA, 1997a: 36-37) are thought to have contributed to over-compliance.

2.2.3 The 'forward looking' approach at polluting impacts in the tax payments

The energy tax payment did not accurately mirror any kind of environmental damage caused by the taxed consumption. Though the energy tax base kept a direct linkage with environmental damage, each unit of its tax base was able to cause many different kinds and amounts of environmental damage depending on the specific pollutant considered. This tax was charged on a unit of measure (volume or weight of fuel), which is not a good proxy for any specific polluting emissions. Furthermore, energy tax rates were not set according to any relevant environmental criteria. Therefore, the tax was unable to signal any specific environmental hierarchy of consumptions. The exception was the energy tax differentiation according to lead content and environmental classes, which targeted a price signal to specific polluting emissions and was based on the costs of abating them as well as set at a level sufficiently high to steer consumption towards cleaner fuels.

The CO_2 tax was raised on the CO_2 content of measured units of fuel. Therefore, it was a product tax equivalent to a tax on CO_2 emissions, i.e. its tax base was a good proxy for CO_2 emissions, since reduction of CO_2 emissions can only follow from fuel-switch or reduced fuel consumption. However, the CO_2 tax payments were unable to always provide environmentally correct behavioural steering. Its rates, though based on relative polluting impacts since they were set proportional to the carbon-content of fuels, in some sectors (especially in the industry) failed to refer to environmentally correct relative *effective pollution prices*. Therefore, the CO_2 tax was unable to compensate for reductions in the energy tax and accurately signal the environmental hierarchy of consumptions.

The sulphur tax and the NO_x charge payments mirrored accurately the evolution of the environmental damage and were set at a level sufficiently high to induce behavioural change. The sulphur tax was a product tax equivalent to a sulphur emission charge, since it was raised on the sulphur content of measured units of fuel and took into account abatement efforts. Therefore, its tax base was a good proxy for sulphur emissions. The tax base of the NO_x was NO_x emissions, usually actual emissions but also presumptive emissions. The sulphur tax, likewise the NO_x charge, had its rates based on abatement costs and provided an adequate incentive towards technological improvement, on which the environmental effectiveness of both taxes strongly depended. As far as the sulphur tax was concerned, on the supply side, tax rebates compensated the cost of producing cleaner oils, inducing reductions in the concentrations of sulphur in oil; whereas on the demand side, the tax stimulated the use of cleaner oils and the more efficient removal of sulphur during combustion by cleaning flue gases in fuels with high sulphur content. The level of the NO_x charge provided a strong incentive for tracing and applying cost-effective solutions.

2.2.4 The subjective tax incidence based on the capacity to avoid pollution

The subjective incidence of the general energy tax was never related to the capacity to avoid pollution or pollution amounts. The polluters best able to control pollution (the industry) and the biggest polluters (the energy-intensive industry) were sheltered from the tax payment, which laid mainly on households and services. This tax has always contained exemptions for some industrial heavy energy consumers, regardless of their environmental impact or capacity to improve in terms of energy consumption. The more favourable treatment assigned in this tax has impacted negatively on the *effective pollution price* following from the energy taxation system disturbing its communication of the correct environmental hierarchy of fuels in several sectors (especially in the industry).

The sheltering from the price signal provided by the CO_2 tax of polluters able to adapt (following ceiling provisions for the industry and the bulk of the tax burden lying on households), with the exception of the district heating sector, and more favourable tax regimes based on pollution amounts (following reductions and exemptions for energy-intensive industry) are due to have impacted negatively on the environmental effectiveness of this tax. Polluters able to adapt but subject to wrong price signals following the more favourable regimes in the energy tax and/or the CO_2 tax kept from shifting to cleaner energy sources.

The sulphur tax burden was allocated according to the capacity to abate pollution with the major tax burden lying on industries where large substitution effects were expected, namely electricity, gas and heating plants. By not sheltering big polluters from the tax, the regulator was able to provide them with the appropriate price stimulus to induce the required investment in desulphurisation technologies. Tax avoidance strategies by big polluters seem to have been relevant in the sulphur tax effectiveness. In 1997, approximately one quarter of the 240 large-scale consumers/taxpayers had implemented emissions control measures through desulphurisation technologies, thereby reducing the tax assessed to them by 70% (Cansier and Krumm, 1997: 60).

The NO_x charge had a selective subjective incidence based on technological criteria that affect the control the polluters hold over pollution. It targeted big emitters, since NO_x abatement technology is characterised by indivisibility and high costs for the most effective types and often involves tailored solutions only available for big companies (OECD, 2010: 6). This was evident following the large spread between best performers. Plants with very high emission intensity levels have improved performance the most, but many were not able to reach levels as low as the ones attained by companies starting from low emission intensity levels (OECD, 2010: 30).

Targeted subjective incidence in the NO_x charge made continuous monitoring of emissions feasible. This together with the high levels charged, both made possible thanks to revenue recycling which also induced tax awareness, were especially relevant for the environmental success of the charge. Behavioural response was highest in net payer plants having the regulated outcome as core business, following their higher tax awareness, and enjoying the financial capacity to invest in technology. The good results obtained in large combustion plants, which were able to exploit economies of scale, were obscured by the performance of the smaller units brought into the system since 1996 (Act 1994:1107, paragraph 3; Hammar *et al*, 2001: 54-55).

2.2.5 Allocation of tax powers based on technical expertise and lack of interest in the tax base

In contrast to the highly technical regulatory sulphur tax design, the design of the energy tax and the CO_2 tax was very similar to the one of standard excise duties on which the Ministry of Finance has traditionally large experience. This entity has designed and administered with a very revenue oriented interest the energy tax and the CO_2 tax, whereas the sulphur tax design was carried out by the SEPA (Ministry of Environment), though its tax administration was led by the Tax Agency (Ministry of Finance). The entity designing the sulphur tax kept no interest in the revenues, since these were never earmarked. This lack of interest was evident in the strong institution of tax avoidance in the tax design, which is part of its strong regulatory facet and explains to a great extent its environmental effectiveness.

The collaboration of the regulated industry is very important to achieve environmental effectiveness in NO_x emissions reduction following the technical difficulties in controlling such emissions due to their complex formation process. Abatement often requires tailored solutions. Consequently, the technical expertise of the entity designing and managing a charge on NO_x emissions is relevant for the environmental effectiveness of such an instrument. Following its technical specificity the NO_x charge was clearly different from an excise duty. It was designed and run by the SEPA (Ministry of Environment), who never kept any interest in its tax base since revenues were fully recycled. The 'charge' denomination followed from these regulatory features.

2.3 Systemic environmental effects of energy taxation were both negative and positive

The energy tax, the CO_2 tax, the sulphur tax and the NO_x charge (especially regarding oil) added to build the *effective pollution price* in the Swedish energy market. The clear price effect on consumption experienced in this market, with the consequent impact on the evolution of CO_2 emissions, sulphur emissions and NO_x emissions, was associated with the *effective pollution price* rather than to partial price signals provided by individual taxes.

The failure to relate the CO_2 tax and the sulphur tax to environmentally correct relative *effective pollution prices* kept them from compensating reductions in the energy tax. Instead, these reductions were used as a way-maker to increase them. This not only had costs in terms of efficiency (Jonsson *et al*, 1997: 6; Andersen *et al*, 2001: 59-60), but also sometimes led to environmentally wrong prices impacting negatively especially on the environmental effectiveness of the CO_2 tax and the sulphur tax following the overlap of tax bases. Uncompensated changes in the energy tax annulled the incentive provided by the CO_2 tax to switch from coal to oil in the heating sector, since both fuels had the same price, and to replace heavy CO_2 loaded oil with electricity (with a strong renewable component) in the industry, since oil was cheaper. This affected the incentive some sectors experienced to switch towards biomass.

The relevance of *effective pollution prices*, rather than partial price signals, for environmental effects was evident following the negative impact of the 1993 energy tax change on CO_2 and sulphur emissions. In January 1993 the industrial energy tax was abolished, although the CO_2 tax and the sulphur tax remained in operation. This tax change led to oil becoming relatively cheaper than electricity (Hammar and Löfgren, 2001: 122). This made interesting the switch from electric boilers to oil-fired boilers. Following this change, emissions increased until 1995, breaking the downward trend, especially in the industry where uncompensated reductions in energy prices following the tax change have been the highest (SEPA, 1997a: 46). The 1993 sharp energy prices fall (approximately 30% for industry) led to an increase in industrial CO_2 emissions of *circa* 21%, 13% of which was attributed to the reform itself (SEPA, 1997: 48, Speck and Ekins, 2002: 92).

The different degrees of substitution of fossil fuels by biomass experienced in the industry also evidenced the relevance of *effective pollution prices*, rather than partial price signals, on energy demand. The positive environmental impact of the CO_2 tax was mainly expressed in increased use of biomass fuels for heating purposes in all sectors but industry (Jonsson *et al*, 1997: 6, 26).

The 1991 tax reform led to coal prices that were more than double the previous level and coal was replaced by biomass, which was both free from taxation and showed price reductions due to technological development and market pressure (Johansson *et al*, 2002: 8). But after the very high level in 1991-1992, the CO_2 tax burden was significantly reduced in 1993 following reductions in the general energy tax that partially annulled it in some sectors.

Following the different *effective pollution prices* experienced in each sector, tax induced price effects on energy consumption were widely different across sectors. District heating experienced the biggest environmental impact, whereas the lack of energy and CO_2 taxes on fossil fuels used for electricity production (where only the sulphur tax was charged) discouraged biomass-based electricity production (Johansson *et al*, 2002: 6).

District heating companies displayed a good capacity to adapt based on technical criteria and management aspects, since their public ownership allowed them a quick reaction to the coherent and sustained price signal they were provided. Following their exposure to fuel prices that accurately mirrored the relative polluting impact of energy products, since they paid fully both the CO_2 tax and the energy tax, these companies adopted a strong fuel switch away from coal and towards biomass.

Following the exemption of solid bio-fuels from the CO_2 tax and the energy tax in district heating, while fossil fuels were charged with both taxes, large differences in the taxation of fuels used for heating purposes made it profitable for plant owners to burn all sorts of fuels except fossil fuels (NCM, 1996: 47, 108-109). In district heating systems the cost of biomass-based heat production was about 50% lower than fossil-fuel-based heat, whilst in industry the opposite happened.

Increased interest to convert existing plants mainly using coal and oil allowed continued growth in the use of wood fuels in district heating along the 1990s, with a fourfold increase between 1990 and 1999, when more than 50% of total district heating supply was represented by biomass (approximately a third of total energy input; whilst the use of oil was less than 10%, ENER-IURE, 2001b: 4). In industry the increase for the same period was 20%. With a 75% increase on CO_2 tax on fuels for district heating and no change in taxes on fuels used in industry, the trend subsisted after 1999 (Johansson *et al*, 2002: 4, 8, 10, 23, 29).

In the heating sector the lack of coordination between the CO_2 tax and the energy tax led to an environmentally wrong price signal (NUTEK 1995; SGTC, 1997). The lower CO_2 tax on oil was neutralised by the lower energy tax on coal, which was only 60% of the energy tax on oil. As the total excise tax burden was the same on both fuels, even though combustion of oil contributed less to CO_2 emissions, the effect of the CO_2 tax largely disappeared in the heating sector. The impact still noticed in this sector might mean high opportunities for substitution.

The environmentally perverse effect of wrong *effective pollution prices* following the 1993 tax reform was also evident in the 30% increase in fuel consumption the energy-intensive paper and pulp industry experienced in the year following the reform compared with about 20% for industry as a whole (EEA, 1996: 53). This evolution was due to the quick change between biomass fuel and oil paper and pulp industry could take and the fact that the CO_2 tax differential induced it to sell its by-products to heating plants and burn low-taxed oil instead (SEPA, 1997a: 52).

Along the first half of the 1990s, the replacement of fossil fuels by biomass (especially in district heating) induced by the CO_2 tax and the general energy tax (Jonsson *et al*, 1997: 6, 26-7) had a positive impact on sulphur emissions, whereas the negative impact of the systemic effect on these emissions was evident in the case of light and heavy fuel oil. The sulphur tax differentiation in favour of light fuel oil, though narrowing the gap between the prices, was not able to compensate for the reduced level of energy tax on heavy fuel oil, which remained cheaper than light fuel oil. Following the environmentally wrong relative *effective pollution prices*, until 1995 the substitution between light and heavy fuel oil only accounted for a small part of the reduction of sulphur emissions following from the sulphur tax compared to other causes (Hammar and Löfgren, 2001: 121).

A systemic effect was also noticed in NO_x emissions. Together with the introduction of a number of new boilers employing modern technology, the fuel switch explains the improvement in NO_x emissions in energy generation. This switch was mainly from coal to biomass fuels and from oil to natural gas following also the price signal introduced by the CO_2 tax and the sulphur tax. Both replacement fuels produce lower NO_x emissions (SEPA, 1997a: 36), being plants using them able to reduce emissions more for a given cost than plants using other fuels (Höglund, 2000: II-36-37,39). Since oil burning produces particularly large quantities of NO_x emissions subject to the NO_x charge, this has also led to fuel switch away from oil. From fuels used by boilers subject to the charge in 1999, 44% were bio-fuels (SEPA, 2000: 6). Tax relief to light heating oil and diesel is likely to have provided an incentive to increase sulphur and NO_x emissions following the income effect it produced (SEPA, 2007: 217).

The 2011 step towards building up the energy tax into a pure energy tax, based on the energy content, plus a CO_2 tax and sulphur tax component with a clear steering intention (as had already been suggested by the SGTC 1997:183, and was proposed in 2011 by the European Commission for the revision of the Energy Taxation Directive (COM(2011) 169 Final, 13 April 2011)), was still unable to guarantee a correct price signal towards cleaner fuel consumption, since economically rational behaviour is led by the sum of the several components of the *effective pollution price*. If the energy tax component disturbs the correct signal provided by the regulatory elements of the energy tax system, as has been the case in the past, the environmental behavioural hierarchy will not be accurately communicated to energy consumers.

CONCLUSION

This chapter analyses four energy related Swedish taxes (the Swedish ‘energy taxation tool box’), namely the energy tax, the CO_2 tax, the sulphur tax and the NO_x charge, with major emphasis on the 1990s and updates until 2011. These taxes were highly representative in terms of revenues and addressed national environmental priorities, namely air emissions and their environmental consequences, such as acidification. They experienced different levels of environmental effectiveness depending on the level of inclusion of the design features of environmental taxes following the causal relationship between inclusion and effectiveness. Stronger and faster environmental effects followed from higher levels of inclusion and *mutatis mutandis*.

Those features include tax designs led by environmental criteria and aimed at precise environmental objectives. This did not verify in the energy tax or the CO_2 tax, which were both used in connection with the early 1990s environmental tax reform and consequent tax shift. The NO_x charge and the sulphur tax complied with such requirement. Furthermore, the sulphur tax had its objectives set with reference to the carrying capacity of the environment.

The reference to environmental criteria with the consequent need to entangle science in the tax process added technical complexity to the tax design with the Ministry of Environment keeping the tax design and management of the NO_x charge and the tax design of the sulphur tax. In any of the cases this entity kept an interest in the tax base following revenue recycling back to the industry in the first case and lack of revenue earmarking in the second. In the energy tax and the CO_2 tax the Ministry of Finance was in charge of a tax design similar to the one of an excise duty and the consequent strong and stable tax revenue stream.

The institution of tax awareness and tax avoidance in the tax design is another feature of the environmental tax design. This was only present in the sulphur tax and NO_x charge. Their usual payment according to actual emissions, following the incentive provided in the tax design for emissions measurement, led to the adoption of tax avoidance strategies that corresponded to pollution prevention and abatement. Tax awareness was also instituted in the NO_x charge design in the selection of the taxpayers and the use of the revenues (full recycling back to the industry).

Another design feature of environmental taxes are tax payments mirroring the evolution of polluting impacts, which requires specific polluting emissions or a good proxy for these as a tax base. The CO_2 tax and the sulphur tax were product taxes equivalent to tax on emissions, whereas the NO_x charge was raised on (actual or presumptive) emissions. However, the energy tax was charged on a unit of measure (volume or weight of fuel) to which different polluting impacts could be associated depending on the pollutant considered. Therefore, unlike the others the energy tax had a tax base (and consequently tax payment) unable to mirror the evolution of environmental disruption. Consequently its price signal was not targeted at a specific polluting impact as required to lead consumption towards the environmental hierarchy of behaviours.

Tax payments must also take a ‘forward look’ at polluting impacts, which requires tax rates based on abatement costs or relative polluting impacts taking into account a specific pollutant and leading to environmentally correct relative *effective pollution prices*. The CO_2 tax rate was based on relative polluting impacts, whereas the sulphur tax and NO_x charge rate was based on abatement costs and was able to stimulate technological progress. The energy tax had rates unrelated to any of those criteria, except in the differentiation according to lead content and environmental classes from which good environmental results followed, and have hence sometimes disturbed the price signal provided by the other taxes towards the environmental hierarchy of consumptions, since energy consumption is determined by *effective prices* rather than partial price signals following from individual taxes.

This has harmed the environmental effectiveness especially of the sulphur tax and the CO_2 tax (since the tax base of the energy tax overlapped with the one of these taxes) in shifting consumption to cleaner sources especially in the industry. The failure of both these taxes to have their tax rates based on environmentally correct relative *effective pollution prices* and compensate for reductions in the energy tax led to more CO_2 and sulphur emissions than would have been the case otherwise.

The subjective tax incidence needs to focus on polluters able to avoid pollution in order to be environmentally effective. This was not the case in the energy tax (except for the tax differentiations according to lead content and environmental classes) and the CO_2 tax where level-related tax departures led to the allocation of the tax burden to best-payers, with the exception of the district heating sector where ability to adapt and endurance of the full tax burden led to good environmental results. Opportunity to improve was attended in the allocation of the sulphur tax burden, with a disproportionately higher tax burden regarding the amount of emissions on the sectors best able to adapt, as well as in the NO_x charge, where taxpayers were selected according to their technical capacity to adapt.

CONCLUSIONS

This dissertation aimed at assessing what an environmental tax is by identifying the design features of such regulatory instrument. The policy guidelines following from their understanding as instruments aimed at implementing environmental policy goals have been tested against and confirmed by empirical data drawn from three country case studies, namely the Danish waste tax, the Portuguese energy tax and the Swedish energy taxation system. These cases have confirmed that the environmental effectiveness of a tax depends on its compliance with precise guidelines, which allow the distinction between environmental taxes and environmnetally-related taxes.

The intensity and quickness with which environmental taxes, as policy instruments aimed at behavioural steering towards precise environmental goals, will be able to deliver such objectives depends on how environmental criteria and the referred objective lead their design, how tax awareness and tax avoidance are instituted in the tax design, how tax payments take a 'forward look' at polluting impacts and taxpayers have the capacity to avoid pollution as well as how much the institutional design has kept tax powers from being used to serve non-environmental concerns. This demanding tax design raises the environmental effectiveness of taxes but also reduces the number of cases where it is possible to use them as instruments of environmental policy.

As regulatory instruments of environmental policy ruled mainly by environmental law and economic law, environmental taxes should aim primarily at pollution prevention or abatement rather than damage restoration or damage compensation. The latter require financial resources that can only steadily follow from tax instruments underpinned by a fiscal rationale that does not reward tax avoidance strategies. Environmental taxes pursue instead behavioural change towards less polluting patterns and clean technological progress (adoption and development). Since they do it through price signals, they are limited by the price system in the sense that their (environmental) effectiveness depends on the existence of competitive markets and they are only able to guarantee the cost and not the level of pollution abatement. However, they are not limited by cost-benefit analysis since they do not aim at internalising external costs, but at using prices to steer behaviours towards the precise (preferably quantitative) environmental objectives previously decided by policy makers.

Both regulatory and fiscal taxes can raise awareness of pollution through price signals that highlight inefficiencies in pollution abatement and opportunities for technological progress. They both also perform shared responsibility by raising taxes on polluters. However, these taxes are substantially different. Their different objectives are served by different normative tax designs and consequently their capacity to deliver environmental effects is far from being the same. Only regulatory taxes directly aim at the fulfilment of environmental goals and consequently stronger and quicker environmental effects follow from their use when compared to fiscal taxes, which may embed environmental concerns as one of the many other possible goals to be attained whilst pursuing their ultimate goal, namely raising revenues. Fiscal taxes tend to include some kind of environmental concern when raised on tax bases holding a close relationship with environmental damages, such as energy taxes. Such taxes can be environmnetally-related, being also sometimes called pollution taxes when raised on polluting substances, but they are not regulatory instruments primarily delivering pollution prevention or abatement.

Though the failure to clearly set the distinction between environmental taxes and environmnetally-related ones threatens environmental tax policy effectiveness and credibility, this has been the case both in the literature and in institutional practice, which is mainly explained by the evolution process these instruments went through from economic ideas to institutional practice as analysed in Chapter I. Some literature has highlighted the importance of choosing the reference elements of pollution taxes, namely the tax base, the tax rate and the subjective tax incidence, in order to allow them to achieve the desired environmental objective (Surrey, 1973: 160, Johl, 1997: 716). It has also been pointed out that when the tax design flows from the decision on the level of environmental quality desired, taxes tend to be the most efficient instruments of environmental policy (Surrey, 1973: 157, Rehbinder, 1993: 66). Moreover, some design features threatening the environmental tax effectiveness have been highlighted. These include, for example, taxable characteristics loosely linked to environmental impacts, exemptions which cannot be motivated on environmental grounds or by particular administrative conditions, level-related tax departures, tax designs anchored to the revenue need rather than environmental goals and differentiated tax rates for non-environmental reasons (Määttä, 1997: 340). However, a systematic highlight of the design features of environmental taxes has been missing.

The design features of environmental taxes

Empirical data analysed in Chapters II to IV confirm that environmental effectiveness of tax instruments causally follows from the design features presented in the theoretical proposal underpinning this dissertation and explained in further detail in Section 3 of the Introduction Chapter. Furthermore, evidence has also been gathered on the little use of such design features in institutional practice, both at EU level and at national level, which might be explained by the demanding tax design involved, as well as the cut with the tradition in the allocation of tax powers it requires. The Ministry of Finance, who traditionally is the authority holding tax powers, tends to miss the expertise required to address the high technical complexity involved in applying environmental criteria to tax instruments. Furthermore, it keeps a direct interest in the non-reduction of the tax base. When holding tax powers on polluting bases, this ministry tends to design traditional excise duties, which are able to perform cost internalisation but are too blunt to induce environmentally correct behavioural change.

The design features on which the environmental effectiveness of taxes depends include a tax design led by environmental criteria, primarily or exclusively aimed at precise environmental objectives and referred to behavioural change by considering the improvement potential rather than mere pollution amounts. A 'forward looking' approach at polluting impacts requires the institution of tax awareness and tax avoidance in the tax design and tax payments mirroring the evolution of environmental damages and imposed on polluters able to avoid pollution at levels sufficiently high to induce behavioural change. Tax payments can mirror the evolution of environmental damages when the tax is raised on specific polluting emissions or a good proxy for them at rates referred to pollution abatement costs or relative polluting impacts taking into account the specific pollutant considered in the tax base. The probability of having the law design and institutional practice following these guidelines increases when the entity with tax powers holds environmental expertise and does not keep an interest in the stability of the tax base.

Among the taxes analysed, the ones with the best environmental performance were those with the highest inclusion of the design features of environmental taxes, namely the Swedish sulphur tax and NO_x charge. Both these taxes scored highly in the inclusion of all those design features. The Swedish CO_2 tax and energy tax and the Portuguese energy tax presented low inclusion of such features and also low environmental effectiveness. In the middle of the rank was the Danish waste tax, which environmental effectiveness was strong and fast but focused on some specific waste streams. A more detailed overview of which features each tax design included is provided in the next sections. This correspondence is listed below in the analysis of the design requirements referred to the institutional design (Section 8) and presented in Table 6.1. below.

Table 6.1. Inclusion of the design features of environmental taxes in the selected taxes

Taxes	Design features of environmental taxes							
	Environmental criteria	Precise objectives	Tax awareness	Tax avoidance	Tax base	Tax rate	Subjective tax incidence	Institutional design
Danish waste tax	M	M	L	L	H	L	H	L
Portuguese energy tax	L	L	L	L	M	M	L	M
Swedish energy tax	L	L	L	L	M	L	L	M
Swedish CO_2 tax	M	L	L	L	H	M	L	M
Swedish sulphur tax	H	H	H	H	H	H	H	H
Swedish NO_x **charge**	H	H	H	H	H	H	H	H

The level of inclusion of the design features of environmental taxes were ranked according to the following scale:
(H) High; (M) Medium; (L) Low

1. Environmental criteria should guide the setup of the tax objectives and the reference of the design features to the latter

Since environmental taxes are instruments of environmental policy there must be a direct causal linkage between the behavioural change they induce and the fulfilment of an environmental objective. For the objective to be environmental and for the behavioural change induced by the tax to be adequate to achieve it, both the decision on the tax objectives and the shaping of the design features of the tax with reference to such objectives must be led by environmental criteria. This requires the use of sustainability criteria to decide on the level of pollution abatement to be pursued, using the carrying capacity of the environment as reference. The design of each tax feature should be adequate to reach the objective set, being adequately judged according to scientific knowledge. If the latter does not provide enough guidance, the precautionary principle should be applied to adopt a conservative approach to environmental risk.

In the taxes analysed, the hierarchy of reference of the tax design (including the setup of the tax objectives) to environmental criteria overlapped with the one communicating the level of environmental effectiveness attained with the tax. In contrast with all the other taxes analysed, the Swedish sulphur tax and NO_x charge had science entangled in their tax design, which led to technically complex but environmentally effective solutions in the way the stimulus to pollution abatement was regulated. This was particularly evident in the delimitation of the tax base in order to cover actual emissions, which allowed tax payments to accurately mirror the environmental damage caused by taxpayers, as well as in the setup of the tax rate with reference to precise abatement costs and in the choice of the taxpayers according to their capacity to abate pollution. In the Swedish sulphur tax the objectives pursued were fixed on the basis of the carrying capacity of the environment, whereas in the NO_x charge such a criterion was not used to set the level of pollution abatement pursued probably due to scientific uncertainty.

Environmental criteria were only partly used in the Danish waste tax. The hierarchy of waste management it pursued slightly deviated from sustainable waste management by ignoring the prioritisation of waste reduction and identifying energy recovery with incineration. Regarding its tax design, it accounted for one of the criteria relevant to assess the material flow entering the ecosystem from which environmental damages follow, namely the amount of the flow, by accounting for waste weight, but did not discriminate according to the composition of the flow. In the Portuguese energy tax and the Swedish energy tax and CO_2 tax, environmental criteria have had no influence on the general tax objectives, which mainly followed from political compromises. The first two taxes adopted tax rate differentiation according to sulphur, lead and VOC (the latter just in the Swedish tax) content in fuels and the CO_2 tax had its rates referred to CO_2

content in fuels. However, due to the relatively small part such regimes represented in the whole tax regimes of the first two taxes and the several deviations from the referred rule in the CO_2 tax, environmental criteria were considered as having had a relatively low and medium importance in the design of these taxes, respectively.

2. Precise and quantified environmental objectives are required for environmental effectiveness

The environmental effectiveness of a tax depends on its targeted design at precise environmental objectives. The tax can aim at one or several objectives as long as they are all served by the same environmental hierarchy of behaviours, since to steer behaviours it is necessary to communicate one precise hierarchy of behaviours. All the design features have to be fine-tuned with reference to that same precise objective (or convergent set of objectives) to guarantee the coherence of the whole from which maximum effectiveness follows.

Likewise in the use of environmental criteria, the hierarchy of use of precise environmental objectives corresponded well to the ranking of the taxes according to environmental effectiveness, which has been understood as proving the need for clear and coherent environmental leadership to steadily guide all features of the tax design towards environmentally correct behavioural change. Taxes with quantitative environmental objectives (such as the Swedish sulphur tax and NO_x charge) tended to perform better than those with qualitative environmental objectives (such as the Danish waste tax until 1990), and both tended to be more environmentally effective than taxes without overall precise environmental objectives (such as the Portuguese energy tax and the Swedish CO_2 tax and energy tax). This may be explained by the fact that the clearer the objectives are the better purpose-oriented the tax design can be.

Precise environmental objectives seem to be associated with increased environmental tax effectiveness even when they are only partial objectives of the tax regime. The Swedish energy tax, though without precise environmental objectives applied to the whole tax regime, had precise qualitative environmental objectives aimed at behavioural change (fuel-switch) included in part of its design, namely in the tax rate differentiation according to lead, sulphur and VOC content in fuels. The environmental effectiveness of these sections of the tax regime was higher than that of the tax as a whole.

The Portuguese energy tax had a similar partial inclusion of precise qualitative environmental objectives, though only referred to sulphur and lead emissions. However, its environmental effectiveness was lower than that achieved with the similar Swedish regime. This might be explained by a lack of compliance with the design requirements applied to the tax rate and the subjective tax incidence, as discussed below (Sections 6 and 7, respectively).

In the Portuguese tax, a precise qualitative environmental objective was also introduced in 2008 (though following phased application, in 2011 it had not yet been fully implemented) with regard to greenhouse gas emissions in the taxation of industrial energy consumption. This partial regime was not expected to deliver relevant environmental effects, due to its multi-purpose and lack of compliance with the design requirements applied to subjective tax incidence, as explained below (Section 7).

Accumulation of environmental objectives or concerns with other kinds of objectives seems to amount to weak environmental leadership in the tax design setup, since low levels of inclusion of the design features of environmental taxes were observed in the presence of such accumulation. This occurred in the Portuguese energy tax and the Swedish CO_2 tax and energy tax, which were also the taxes with the lowest evidence of environmental effectiveness. Environmental concerns have been included in the Portuguese energy tax since 2000, when the law was changed to include internalisation of environmental costs as a guiding principle of excise taxes. These concerns were added to the pre-existent ones related to revenue collection and national competitiveness. These objectives were also embedded in the Swedish CO_2 tax together with relatively vague environmental objectives referred to the improvement of the price competitiveness of less polluting fuels and sustaining of the transition to renewable sources. In the Swedish energy tax the same non-environmental objectives were combined with the already referred to partial objectives regarding lead, sulphur and VOC emissions, as well as overall concerns with energy efficiency.

3. TAX AWARENESS MUST BE INSTITUTED IN THE TAX DESIGN

The price signal provided by the tax works as an incentive to change behaviours when polluters are aware that they get a financial loss if they pollute and avoid it if they prevent or abate pollution. They must be aware of the tax payment (including the size of such payment) and its rationale (why they pay and how they cannot pay). Polluters should understand the linkage between the tax payment and the environmental damage and actually support the financial loss rather than transfer the tax burden to others. Designs that lead to fiscal illusion by hiding the tax payment in another price, spreading it over time or delaying it impact negatively on the environmental effectiveness of the tax.

Among the taxes analysed, the importance of tax awareness was especially evident in the Danish waste tax and the Swedish NO_x charge. Whereas in the Danish case this feature explains the relative environmental effectiveness of the tax, in the Swedish case it has contributed to the high environmental effectiveness attained. Except for the Swedish sulphur tax, where high presumptive levels of emissions induced the adoption of emissions measurement that led to high tax awareness, in all the other cases this was low due to tax payments as part of the energy prices, lack of pollution awareness and no significant financial loss following the tax payment, since the biggest energy consumers were exempted in the Portuguese energy tax and Swedish CO_2 tax and energy tax.

In the Danish waste tax, the waste companies as well as the few waste producers bringing their waste directly to treatment plants paid the tax according to waste weight and as a stand-alone price, namely as tax payment and gate fee, respectively, without any relevant time gap between the waste delivery at the plant and the payment. However, waste companies tended to pass the financial burden of the tax to waste producers. Companies using professional carriers took the financial loss of the tax usually as part of the total waste disposal price. All the other waste producers covered by collection schemes took such loss as part of another price (sometimes unrelated to waste disposal, such as rents, real estate taxes, and contribution to residents' association) with a relevant time gap between waste disposal and payment. Furthermore, this payment was usually flattened into a volume-based waste collection fee where the linkage between the payment and waste disposal was cut.

For all the payers, the amount of the payment was set at a level too low to raise cost-awareness, except regarding heavy weight waste streams following the tax targeting of the price at weight. For waste producers this was due to the absolute low level of the *effective waste disposal price*. This was especially evident in industry. Waste-extensive companies, where most opportunities for improvement lay, were not economically stimulated to reduce and recycle waste following the tax levels set with reference to waste-intensive companies. For waste companies, the level of the tax might have been insufficient to compensate for concurrent economic interests in exploiting available traditional disposal capacity for which they set cost-based prices under monopoly. Such a level was all the more insufficient as a great part of the tax burden was transferred to waste producers through waste management fees.

Therefore, tax awareness was the highest in waste treatment companies and the few waste producer companies delivering their waste directly to treatment plants. Tax illusion was especially high among householders following the municipal charging system used. In the industry still enjoying opportunities for improvement, which was the main part of it, waste disposal cost-awareness was low, except with regard to heavy waste. The behavioural reaction to the tax was accordingly highest among the waste companies (and the municipalities who own most of them) who set and run collection schemes for recycling crucial for the waste diversion from traditional disposal to recycling. There was not a significant increase in recycling when this was not mainly managed by the waste companies through collection schemes. Also, the biggest increases in recycling were experienced in heavy weight waste streams.

Tax awareness was instituted in the Swedish NO_x charge by its targeting a selected group of polluters where high attention was dedicated to the regulated source of environmental damage, namely energy production and its effectiveness, and where a high financial impact was expected to follow from the tax as a consequence of its focus on the core of the business. The linkage between the tax payment and environmental damages was enhanced to taxpayers by setting payments based on continuously measured emissions (following the incentive to adopt measurement provided by high levels of presumptive emissions). This linkage was reinforced by the recycling of the revenues back to the industry according to a criterion relevant to the production of the environmental damage, namely efficiency in energy production. A substantial (even over-compliance in 1996, SEPA, 1997a: 36) and fast reduction in NO_x emissions, control of which is technically difficult following the complex formation process (OECD, 2010: 36), was experienced in the regulated industry as a consequence of the use of low cost abatement opportunities available that the cost of opportunity introduced by the tax revealed as well as of technological improvement (Höglund, 2000: II-36; SEPA, 2000: 10).

4. THE TAX DESIGN SHOULD REWARD THE ADOPTION OF TAX AVOIDANCE STRATEGIES THAT SERVE THE TAX PURPOSE

Polluters taxed according to their capacity to avoid pollution, aware of the tax burden and understanding its rationale will pursue tax avoidance strategies linked to pollution prevention or abatement if they follow a cost-minimising approach. The tax design may induce further the adoption of such strategies by (1) allowing them (e.g. narrow tax bases focused on the most polluting goods and for which there are non-taxed cleaner alternatives should be preferred over extensive tax bases), (2) raising the position of the tax payment in the hierarchy of attention of the taxpayers (cost-awareness may increase following the rise of the financial impact of the tax) and (3) signalling and rewarding tax avoidance strategies linked to pollution prevention or abatement (e.g. by using tax rate differentiation or tax refunds). More favourable regimes should be based either on the adoption of such kind of strategies or on reduced capacity to adopt them.

The reward of tax avoidance strategies was especially important for the environmental effectiveness of the Swedish sulphur tax and NO_x charge. The negative impact the absence of such a reward had on the capacity of the tax to steer behaviours was especially evident in the Danish waste tax. In all the taxes where the inclusion of the design features of environmental taxes was considered low, namely the Portuguese energy tax and the Swedish energy tax and CO_2 tax, tax avoidance strategies were not rewarded. In the Portuguese case, following the free price regime applied to light fuel oil, the consumption of this fuel was even more costly than the use of heavy fuel oil, which was sold under a maximum market price regime. Consequently, the market share of the clean fuel continued to be irrelevant after the adoption of the tax rate differentiation according to sulphur content in fuel oil.

Both the sulphur tax and the NO_x charge had tax avoidance instituted in their design by allowing tax payments directly proportional to actual emissions. In the sulphur tax reimbursements following proved abatements provided an incentive for the adoption of measuring equipment. In sulphur emissions, where abatement was easier than in NO_x emissions, tax avoidance strategies have been common practice among the taxpayers, a quarter of which reacted to the tax in the short to medium term by taking measures to clean flue gas leading to a 70% reduction in sulphur emissions on average (EEA, 1996: 53). The reward of tax avoidance strategies, together with compliance with the design requirements referred to subjective tax incidence, might provide the main explanation for the higher environmental effectiveness of the sulphur tax vis-à-vis the CO_2 tax.

In the Danish waste tax, the transformation of the weight-based tax into a volume-based or frequency-based waste collection fee eliminated the reward for any reduction in the amounts of waste delivered for traditional disposal when such reduction could not be expressed in reduced volume capacity or frequency of collection, which was especially the case in publicly-owned rented flats and single-family houses. A reduced use of recycling facilities was observed in municipalities using such a kind of charging, since waste tax savings were distributed over the normal fee for waste disposal, creating a free-ridding problem (Andersen et al, 1997: 90). Tax avoidance was instituted in this tax design mainly by tax refunds provided for waste next removed for recycling, which induced strategies matching pollution prevention or abatement and provided an incentive to waste companies to create waste separation schemes.

5. THE TAX BASE SHOULD BE ABLE TO MEASURE THE ENVIRONMENTAL DAMAGE

The tax base should be able to measure the environmental damage caused by the taxpayer. This is necessary because decisions on pollution abatement follow from the comparison between the private benefits derived from pollution added to the environmental tax payment on the one hand, and the costs of preventing or abating pollution on the other. Therefore, the price signal provided by the tax needs to be proportional to such benefits to be environmentally effective, and private benefits tend to be proportional to the environmental damages.

Furthermore, by increasing the price of the tax base, the tax aims at manipulating its size, namely reducing it, as a means to change the amount of environmental damage and also reduce it, in the sense that the tax takes the tax base as a proxy for the environmental damage. By increasing the price of the tax base it aims at increasing the price of the environmental damage. For the tax to be able to deliver the expected environmental improvement it is hence necessary that the tax base is positively correlated with the environmental damage. This occurs when it is the specific polluting emissions that the tax pursues to prevent or abate or a good proxy for them.

The taxes with the highest environmental effectiveness corresponded well with those whose tax base allowed their payment to accurately mirror the evolution of environmental damages. The NO_x charge was raised on specific polluting emissions, namely NO_x emissions, either presumptive or measured. The Swedish sulphur tax and CO_2 tax were raised on the sulphur and CO_2 content of measured units of fuel, which are a good proxy for sulphur and CO_2 emissions, respectively. Therefore, they were product taxes equivalent to sulphur and CO_2 emission taxes, respectively. The Danish waste tax was raised on weighted waste disposed at landfills and incinerators, which is a fairly good proxy for polluting emissions (and consequently the environmental damages they caused) following from such behaviour, since damages are caused by the amounts of material flow entering the ecosystem, though also by the composition of such flow for which the tax was unable to account. The Portuguese and Swedish energy taxes were general energy taxes raised on a measured unit of fuel or energy product. Therefore, they were not good proxies for specific polluting emissions, as explained below in this section.

Moreover, the tax base should be comprehensive enough so as not to allow tax avoidance strategies that do not correspond to less environmental damage. The failure to follow this requirement seems to have led to low environmental effectiveness in the Portuguese energy tax. The narrow coverage of this tax, namely the exemption of fossil fuels, is thought to have kept it from providing an incentive to public transportation operators, especially private ones, to adopt new technology, and to power plants to shift towards renewable energy sources. The tax exemption provided to electricity kept the tax from pricing energy scarcity and is thought to have contributed to the rising pattern of electricity consumption in the country, the demand of which was not de-linked from GDP growth.

General energy taxes tend to perform poorly in environmental behavioural steering

General energy taxes raised on measured units of a fuel or energy product tend not to apply one rate per product due to the increased administrative costs such an option involves. They usually apply the same rate to several products, as evidenced by the Energy Taxation Directive (2003/96/EC) and the energy tax designs adopted in Portugal and in Sweden. However, taxes raised on measured units of a fuel or energy product will hardly be able to accurately communicate the environmental hierarchy of behaviours with reference to a specific pollutant unless they use one rate per product. This is due to the fact that usually one same measured unit of different products is able to generate different amounts of environmental damage depending on the composition of the product.

If the same rate is applied to two or more products that have different polluting contents, the tax is wrongly communicating that they have the same position in the environmental hierarchy referred to a specific pollutant. For example, in Portugal, since 2008, the energy tax raised on industrial fuels, which aimed at reducing greenhouse gas emissions in general, signalled to energy consumers a similar impact of coal and oil in terms of those emissions, since the same rate applied to coal and oil. However, these two fuels have different CO_2 contents. The same occurred in the Energy Taxation Directive and in the Swedish energy tax.

Depending on whether the price signal provided by the general energy tax is not changed by the other components of the *effective pollution price*, which is the price ultimately leading consumption behaviours, such kinds of taxes might also misguide consumption. Therefore, it is relevant for environmental effectiveness whether general energy taxes are simultaneously used with taxes raised on specific polluting contents of measured units of a fuel or energy product (the systemic effect is analysed next in Section 6). The same reasoning applies if following market segmentation the general energy tax addresses simultaneously several kinds of polluting emissions, being the part of the tax applied to each market segment analysed as if it was a separate tax.

Furthermore, the tax base should be narrow enough so as to allow the opportunity to adopt tax avoidance strategies corresponding well to pollution prevention or abatement. However, the comprehensiveness of the tax base of general energy taxes eliminates or significantly reduces the space for fuel-switch. Therefore, any improvement on some specific emissions must follow from reduced consumption, which is difficult to attain due to the usually low price elasticity of energy consumption even in the long run. Depending on positive price elasticity, these instruments can induce energy efficiency by raising the price of energy, especially when they are raised according to energy content. However, following their bluntness they perform poorly in reducing specific emissions.

Therefore, lower environmental effectiveness is expected in general energy taxes than in taxes raised on the specific polluting contents of measured units of a fuel or energy product. This hypothesis has been confirmed in the cases analysed. Among the taxes raised on a measured unit of fuel or energy product, the tax with the highest environmental effectiveness, namely the Swedish sulphur tax, was raised on the sulphur content of the measured units and those with lowest evidence of environmental effectiveness, namely the Portuguese and the Swedish energy tax, were raised on each measured unit without further specification.

The low environmental effectiveness of the Swedish CO_2 tax, which was also raised on the specific polluting content of each measured unit, seems to have been due to a great extent to the negative systemic effect of the Swedish energy tax, which tended to neutralised the signals provided by the CO_2 tax, spreading its bluntness to the latter. Though this systemic effect was also experienced in the sulphur tax, the strongest inclusion of other design features of environmental taxes, especially the ones regarding tax avoidance and subjective tax incidence, might explain the higher environmental effectiveness of the sulphur tax vis-à-vis the CO_2 tax. Under the sulphur tax the polluters best able to avoid pollution were taxed and could avoid the tax burden by adopting abatement technology, whereas under the CO_2 tax the only option available to prevent pollution was increased energy efficiency and the biggest energy consumers tended to be exempted.

Environmental effectiveness is hindered by loose linkage between tax base and environmental damages

A loose linkage between the tax base and the environmental damage reduces the probability of influencing the amount of pollution with the tax. For example, if environmental damages following from consumption depend more on the process and technology used than the product consumed, a tax raised on the amount of product consumed is expected to have a low impact on the amount of environmental damages. The same effect occurs following institutional filters that dissociate tax payments from changes in the tax base. In such cases it is difficult to reward environmentally correct tax avoidance strategies. Therefore, it is necessary to not only carefully choose the tax base but also address any institutional filters that disturb or cut the linkage between the tax payment and environmental damages, in the sense that changes in environmental damages are not reflected in the tax payment because the latter fails to mirror changes in the tax base.

The failure to address these kinds of filters in the Danish waste tax as far as waste producers covered by volume-based waste collection fees were concerned, as discussed above whilst analysing the design requirement for reward of tax avoidance (Section 4), made their payments for waste disposal the same regardless of the amount of waste disposed unless volume capacity or frequency of collection were changed accordingly. The failure of the tax to reduce absolute waste amounts and the lower levels of recycling registered in municipalities using volume-based charging seems to indicate the loose linkage between the tax payment, which did not reflect changes in the tax base, and the environmental damage caused. This loose linkage reduced the influence of the tax over pollution.

Consideration for abatement efforts when the tax base is a proxy for polluting emissions raises environmental effectiveness

To improve the accurateness of the linkage between the tax base and environmental damages when the tax base is a proxy for polluting emissions, the efforts taken to prevent or abate such emissions through the use of technology should be taken into account. For example, if the regulator addressed the amount of sulphur emissions by taking as tax base the sulphur content of energy products, the adoption of abatement technology should be taken into account in the tax payment even if the tax base did not change. Allowing the tax payment to take into consideration the adoption of abatement technology provides an incentive to polluters not only to adopt it, with the consequent environmental improvement, but also to measure emissions. The latter is stimulated by setting presumptive emissions at a level substantially higher than actual emissions. Self-measurement of emissions may lead the polluter to adopt pollution control measures following raised tax awareness and pollution awareness. In the Swedish sulphur tax, the reward of this kind of avoidance strategy seems to have had the effect described with the consequent positive impact on the environmental effectiveness of the tax, as explained above, whilst addressing the design requirements referred to tax avoidance (Section 4).

6. THE TAX RATE SHOULD LEAD ENVIRONMENTALLY CORRECT BEHAVIOURAL CHANGE

The tax rate measures the intensity of the regulatory intervention and should be levelled according to the precise environmental objective pursued taking into account the price-elasticity of the tax base and relative *effective pollution prices*. The intensity of the price signal provided by the tax follows from the application of the tax rate to the tax base. This operation allows the price signal to be set at the amount required to induce the desired behavioural change, rather than cost internalisation, which is just a means of achieving pollution prevention or abatement and usually is neither a feasible (due to the high levels of taxation required) nor a sufficient means of pursuing pollution prevention. Behavioural change is led by both absolute prices and relative prices. Therefore, all prices would need to be correct for behavioural change to follow from the cost internalisation tax. Cost internalisation should be pursued at the amount necessary to induce the desired behavioural change rather than as an end in itself that ultimately aims at full cost internalisation.

To induce behavioural steering the tax rate should be referred to pollution abatement costs or relative polluting impacts taking into account a specific pollutant leading to environmentally correct relative *effective pollution prices*. In competitive markets, the level of the tax rate should correspond to the one allowing the tax payment to match the level of abatement costs necessary to attain the objective pursued, since a profit-maximising firm will prevent or abate pollution until the marginal cost of doing it equals the tax payment. Therefore, if abatement costs drop following technological progress and the tax rate is not revised accordingly, the tax will tend to deliver higher environmental results than initially expected.

As regulatory instruments rather than punitive ones, environmental taxes should account for the distinction between subsistence emissions and luxury emissions and efficient and inefficient production attending to the alternative technologies and processes available. Therefore, the level of the tax rate should highlight low-cost opportunities in pollution prevention and abatement as well as make it financially attractive for polluters to shift towards the behaviour pursued.

When abatement is not possible or sufficient to achieve the objective pursued, the tax should communicate the relative polluting impacts of the products taxed from which the environmental hierarchy of behaviours follows. In such a case environmental effectiveness depends on a substitute effect. For this to occur the tax differential between the green good and the polluting good must reach a threshold level in which more environmentally-friendly products have the same price as (or a lower price than) alternative regular ones. And such a effect will be the highest, depending on how wide the price difference between substitute goods is.

Relevant for polluters' decision is not the price signal provided by the tax but the *effective pollution price* ultimately reaching them and of which the tax is part together with all the other components of the price system. *Effective pollution prices* in relative terms explain consumption-shift, whereas in absolute terms they influence the total level of consumption. Since the relevant for pollution decisions is the aggregate price, the tax level should compensate for wrong price signals provided by other elements of the price system. Any institutional filters that prevent the tax from communicating environmentally correct *effective pollution prices*, both in relative and absolute terms, should be addressed.

In the taxes analysed, the Danish waste tax and the Swedish sulphur tax and NO_x charge were the only ones referred to abatement costs, in the sense that tax rates were set at a level aimed at making pollution prevention or abatement economically profitable. The Danish waste tax intended to make recycling profitable. However, the level at which the tax was set only led to environmentally correct relative *effective waste disposal prices,* compensating for the costs of recycling, in heavy weight waste streams and consequently only regarding those the tax reached high levels of environmental effectiveness. Since 1992, the tax was also referred to relative polluting impacts as far as the relative tax rate differentiation was concerned with the intent of favouring some waste management techniques over others. However, the absolute level at which each rate was set was not always able to compensate for low municipal fees charged in landfills. Therefore, in some municipalities the tax advantage to incineration did not materialise into lower *effective waste disposal prices* at incinerators than in landfills. Regarding its absolute level, the setup of the tax rates in levels that would not threaten the competitiveness of energy-intensive industries led to tax levels too low to induce behavioural change in the industry where more opportunities for improvement were available, namely waste-extensive industries and services. Therefore, the tax stayed short of its full potential in behavioural steering (Andersen et al, 1997: 8).

In the Swedish sulphur tax and NO_x charge the setup of the tax rate above abatement costs led to technological improvement, on which the environmental effectiveness of both taxes strongly depended. In the sulphur tax, tax rebates compensated the cost of producing cleaner oils, inducing reductions in the concentrations of sulphur in oil and use of cleaner oils, as well as the costs of adopting clean technology for more efficient removal of sulphur during combustion. The strong and fast environmental effects following from the NO_x charge were to a great extent due to the revelation of low-hanging fruit following the introduction of the tax (Höglund, 2000: II-36) evidencing inefficient production. The drop in abatement costs without revision of the rate of the charge led to over-compliance in plants belonging to the energy sector in 1996 (SEPA, 1997a: 36-7).

The lack of evidence of environmental tax effectiveness both regarding energy efficiency and emissions reduction is coherent with the Portuguese energy tax design as far as tax rates are concerned. Designed as a traditional excise duty on fuel consumption, with low inclusion of the design features of environmental taxes, this tax provided a blunt price signal to consumers. Accordingly, fuel-switch experienced in Portugal during the 1990s was low. Therefore, its effect was expected to be limited to the adoption of fuel conservation strategies, which depend more on the absolute level of *effective pollution prices* than on their relative levels.

However, following the use of the tax as market fuel price stabiliser until 2005, the absolute level at which tax rates were set led to decreasing real energy prices during the 1990s, which, together with the low reward for tax avoidance strategies entailing reduced pollution and the narrow subjective tax incidence, is thought to have led to its low behavioural steering capacity. In the tax rate differentiation according to lead content of gasoline, the level of the tax differential was too low to significantly induce behavioural change. Even after 2008, when it was decided to raise maximum energy tax rates applied to industrial fuels, the rates used continued to correspond to the minimum levels set in the past.

Furthermore, the relative prices provided by the Portuguese energy tax were not always environmentally correct. Following the failure to address institutional filters, namely uncompetitive market structures, the tax differentiation according to sulphur content was not able to lead to environmentally correct relative *effective pollution prices*, the market price of heavy fuel oil being lower than that of light fuel oil. The tax also led to environmentally wrong relative prices following the always lower tax rate applied to diesel than to gasoline and the same tax rate applied since 2008 to coal and oil.

Lessons on the structure of energy taxation systems: Systemic effects affect environmental effectiveness of individual taxes

The structure of national energy taxation systems, including those proposed by the Energy Taxation Directive (2003/96/EC) and the 2011 European Commission's proposal for the revision of the referred directive[43], could be significantly improved following the acknowledgement that systemic effects influence the environmental effectiveness of individual taxes. Energy consumption-related pollution would be addressed in the most environmentally effective way if taxes raised on energy consumption were referred to specific polluting impacts of the fuels or energy products rather than only or also to measured emissions of the latter or their energy content.

Relevant for decisions on energy consumption are not the partial price signals provided by individual taxes but the whole tax burden (both in absolute and relative terms) on each energy product (*effective tax*), and ultimately by the sum of all the price components reaching the final consumer where the *effective tax* is included (*effective price*). The amount of energy consumed is influenced by absolute prices and the kind of energy consumed is influenced by relative prices.

Since pollution decisions follow from the comparison between *effective pollution prices*, the simultaneous use of several regulatory taxes overlapping the same tax base but pursuing different environmental objectives disturbs the environmental effectiveness of each other when such effectiveness follows from different environmental hierarchies of behaviour, in which case a choice between objectives is necessary, and enhances it when the same hierarchy serves all the objectives. This systemic effect was experienced in the Swedish case.

The overlap of the Swedish CO_2 tax and energy tax over the same tax base led, since 1993, to *effective pollution prices* too low to induce behavioural change. This incapacity mainly followed from a systemic effect that led to environmentally incorrect relative *effective pollution prices*. This was due to the neutralizing effect the lower energy tax rates had on the price signals provided by the CO_2 tax and the failure of the latter to refer to environmentally correct relative *effective pollution prices*, as required to compensate for the wrong price signal provided by the energy tax.

The negative environmental impact of systemic effects was evident in the different level of switch from fossil fuels to biomass between 1990 and 1999 in district heating (50%), which paid fully both the CO_2 tax and the energy tax, and in industry (20%), where the CO_2 tax was compensated by reductions and exemptions in the energy tax (Johansson *et al*, 2002: 6). Environmentally wrong relative *effective pollution prices* also failed to provide an incentive to switch from coal to oil in the heating sector, since both fuels had the same price, and to replace heavy CO_2 loaded oil with electricity (with a strong renewable component) in the industry, since oil was cheaper.

Systemic effects have also affected the environmental effectiveness of the sulphur tax and the NO_x charge. In the case of the sulphur tax, together with a negative impact, especially regarding the neutralisation of its incentive to switch to light fuel oil, was experienced a positive impact, since the switch to biomass in district heating led to reduced sulphur emissions. The NO_x charge led to positive fuel switch away from oil, since burning oil produces particularly large quantities of NO_x emissions, and has hence influenced the environmental effectiveness of the CO_2 tax and the energy tax. Similarly, the environmental effectiveness of the NO_x charge was positively affected by the fuel switch induced by the CO_2 tax and the energy tax.

Following the comparison between the Portuguese case and the Swedish one, the analysis showed the importance of using taxes raised on specific polluting emissions or a good proxy for them as much as possible to address energy consumption-related pollution. Ideally only this a kind of taxes should be used due to their higher environmental effectiveness when compared to general energy taxes and the negative systemic effect the latter play on the environmental effectiveness of the first. For the same reasons, complementing the general energy tax with taxes raised on specific polluting emissions or a good proxy for them might be preferable to the exclusive use of general energy taxes.

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[43] European Commission (2011), Proposal for a Council Directive amending Directive 2003/96/EC restructuring the Community framework for the taxation of energy products and electricity, European Commission, COM(2011) 169 Final, 13.4.2011.

The higher environmental effectiveness of the Swedish energy taxation system, which was dominated by the CO2 tax, when compared with the Portuguese energy tax, is thought to be related to the level of the tax burden and the use of taxes raised on specific polluting emissions or a proxy for them. The higher *effective price* of pollution following from energy consumption in Sweden, where several taxes were simultaneously raised in connection with energy consumption, than in Portugal, where industrial energy consumption has been mainly sheltered from energy taxes, might help to explain the better results obtained by Sweden in improvement of energy efficiency. Furthermore, adding taxes raised on specific polluting emissions or a good proxy for them to the general energy tax in Sweden, but not in Portugal (where the only other tax raised on energy consumption was VAT, which did not change the price signal provided by the energy tax), might help to explain the better results Sweden achieved in terms of shifting consumption towards cleaner fuels.

The use of taxes raised on specific polluting emissions or a good proxy for them is all the more important as, following the low price elasticities of energy demand in some sectors, the impact of energy taxes on the environment is expected to follow more from the consumption shift towards cleaner energy sources than from raised energy efficiency. Since the 1973 oil crisis, the price elasticity of oil demand has decreased continually in the past decades, as the structure of demand has changed and the easiest measures to reduce consumption have been installed (Pieprzyk and Kortlüke, 2010: 3). These measures have mainly concerned the reduction of crude oil in the heating sector, as exemplified by the Swedish case. One of the most polluting and traditionally fossil fuel-dependant sectors, i.e. the transport sector, has been allocated the highest bulk of energy taxation in most countries since the 1970s without showing much improvement in its energy consumption pattern (ibidem). This has been confirmed by data from the application of the Portuguese energy tax. This tax was unable to change the rising pattern experienced in national private transportation since the 1970s.

The comparative analysis of environmental effects following from energy taxation in Portugal, whose regime resembled the one set by the Energy Taxation Directive (2003/96/EC), and in Sweden, whose energy taxation system was more similar to the 2011 European Commission's proposal for the revision of the referred directive (COM(2011) 169 Final), indicates that, though some improvement is expected in the Directive if the proposal for its revision gets approved, the directive will continue not to lead the structure of the national energy taxation systems towards their full potential in terms of environmental effects.

The simultaneous use of general energy taxes and taxes raised on specific polluting emissions following from energy consumption or proxies of these, though an improvement vis-à-vis the use of general energy taxes only, was already allowed by the Energy Taxation Directive (2003/96/EC). This approach faces the criticism that the price signal provided by taxes raised on specific polluting emissions following from energy consumption or a good proxy for them tends to get disturbed by the price signal provided by general energy taxes, whose payments cannot sign a single environmental hierarchy of consumption referred to a specific pollutant following the tax base used (i.e. measured units of fuel) unless one rate per fuel is used. The split of the energy taxation system between an energy component and a CO_2 component, as proposed by the Commission, reduces the intensity of the price signal provided by the energy component but does not neutralise it.

From an environmental perspective, an important improvement would follow from a structure of the energy taxation system that included only taxes on specific polluting emissions or proxies for these, as long as the pollutants involved produced the same environmental hierarchy of consumption. Under these conditions, energy taxation would lead to an environmentally correct *effective tax* in relative terms. The add-up of the several taxes on each fuel could provide a total tax burden (*effective tax*) sufficiently high to induce energy efficiency. This system could also include electricity depending on the use of certificates of origin where the primary energy sources used for its production were mentioned.

7. TAX SUBJECTS SHOULD CORRESPOND TO POLLUTERS ABLE TO AVOID OR ABATE POLLUTION

Environmental taxes are regulatory instruments aimed at behavioural steering. Therefore, they must be targeted at those who control the cause *sine qua non* of pollution and still have not explored all their opportunities for improvement. High environmental effectiveness is expected to follow from selective tax incidence focused on industries where large substitution effects are expected. Different tax treatment for equal polluters will only be coherent with the regulatory rationale when based on the control polluters hold over pollution. Technological aspects influencing this control may justify selective tax incidence as well as more favourable treatment. In contrast, the amount of pollution cannot justify more favourable treatment, though it can support selective tax incidence if high amounts of pollution coincide with opportunities for improvement. Tax exemptions based on the amount of pollution hinder the environmental effectiveness of the tax and provide a wrong price signal that tends to perpetuate the polluting patterns into the future. Furthermore, they are inefficient, unfair and might challenge the feasibility of adopting the tax.

The use of revenues in connection with selective tax incidence exemplifies how the application of the polluter pays principle does not necessarily lead to the most environmental effective tax design, though it might lead to economic efficiency in a broader sense. Recycling the revenues back to the taxpayer according to an environmental criterion allows limited tax incidence focused on the polluters best able to avoid pollution by addressing potential market distortion following thereof, as well as tax levels sufficiently high to induce behavioural change, by reducing political opposition to the tax. Moreover, it increases tax awareness by enhancing the linkage between pollution and financial loss. Following the refund polluters will stay short of paying regarding the emission charges refunded. However, this is expected to not negatively impact on the environmental effectiveness of the tax if three conditions apply. First, there is a single output upon which the refunding can be based. Second, such output does not correspond to the tax base. And, third, each plant's output is small enough relative to the total output by regulated plants to form a competitive situation.

The strongest behavioural effect induced by the Danish waste tax was experienced following actions taken by the taxpayers, namely the setup of collection and separation schemes by waste treatment plants. The kind of pollution abatement pursued by the tax depended mainly on the taxpayers and only to a lower extent on those ultimately taking the tax burden, namely the waste producers. Waste treatment plants at the time of the tax introduction were mostly owned by the municipalities. These controlled the waste management system. The tax relied heavily on changes to this system to achieve its goal, namely to reduce traditional waste disposal through increased recycling and, since 1992, to shift traditional waste disposal from landfills to incinerators (and, since 1997, especially incinerators with energy recovery). Therefore, the municipalities (through waste treatment plants) were the best able to prevent pollution in the way envisaged by the tax, i.e. by setting recycling facilities, collection schemes for recyclable waste streams and by investing in incinerators with energy recovery.

In the Portuguese energy tax, the exemptions provided to those best able to prevent or abate pollution explains to a great extent the low evidence of positive behavioural change following the application of the tax. This tax was mainly raised on consumers of motor fuels, who enjoyed very low alternatives both in the short and in the long run, whereas the industry, especially the energy-intensive one where increased opportunities for improvement were expected, stayed sheltered from the price signal provided by the tax. Following the law change in 2008, the concept of energy-intensive companies was broadened. Therefore, the simultaneous application of the tax to energy consumption by these companies when they were not covered by the ETS or energy efficiency agreements was expected to have a low behavioural impact. Exemptions to energy-intensive companies were especially harmful for the environmental effectiveness of the tax differentiation according to sulphur content in fuel oil.

The comparative analysis between the almost null behavioural impact of the tax rate differentiation on the Portuguese energy sector with the significant reductions achieved with the sulphur tax in Swedish power plants also pointed towards the negative impact on the environmental effectiveness of taxes of exemptions provided to big polluters. The Swedish sulphur tax was raised according to the capacity to abate pollution. Its tax burden was mainly allocated to the sectors where large substitution effects were expected, namely electricity, gas and heating plants, which in Portugal were exempted from the energy tax. Oil companies in both countries were overcompensated by the production of clean fuel. In addiction, in Sweden there was an incentive towards technological development in the supply side. In Portugal, such an incentive was lower following the low demand for the clean fuel which was only based on better operational performance of light fuel oil, since the market price of heavy fuel oil was often lower than the one of light fuel oil and polluters did not pay for sulphur emissions. The accurate targeting of the Swedish tax together with the reward of tax avoidance strategies also explain to a great extent its success in inducing the adoption of investment in desulphurisation technologies (demand-side technological developments). In 1997, approximately one quarter of the 240 large-scale polluters/taxpayers had implemented such technologies, thereby reducing the tax assessed to them by 70% (Cansier and Krumm, 1997: 60).

In its subjective incidence the NO_x charge was as well targeted as the sulphur tax. The targeted subjective incidence and the high level of the NO_x charge, both possible thanks to revenue recycling, were the main drive of the success of this tax in reducing NO_x emissions. Reduction of NO_x emissions is technically difficult and depends to a great extent on continuous monitoring. The most effective abatement technology entailed high costs. Polluters covered needed to have the capacity to adopt such technology. Therefore, the Swedish NO_x charge was raised on plants belonging to the energy sector, which were also those who were expected to suffer the strongest financial impact following the application of the tax and where tax awareness was expected to be the highest, as explained above when analysing the design requirements related to tax awareness (Section 3). Following abatement costs and the desire to extend the regulatory intervention to other sectors, small units were brought to the system since 1996. The low performance achieved by these units has obscured the good results obtained in large combustion plants (Hammar *et al*, 2001: 54-55).

The Swedish CO_2 tax and energy tax, like the Portuguese energy tax, tended to exempt energy-intensive companies. Therefore, the behavioural reaction experienced following the application of these taxes was also low where more opportunities for improvement were available. The negative environmental effect following from such exemptions was evident following the 1993 law change and the higher fuel switch from fossil fuels to biomass experienced by district heating compared to industry, as explained above when discussing the design requirements applied to the tax rate (Section 6). Energy-intensive companies experienced the high tax burden following from the accumulation of the CO_2 tax and the energy tax between 1991 and 1993. The emissions increase until 1995, following the exemptions provided to these companies in 1993, brooked the downward trend initiated in 1991. The tax-induced 1993 sharp energy prices fall (approximately 30% for industry) led to an increase in industrial CO_2 emissions of *circa* 21%, 13% of which was attributed to the reform itself (SEPA, 1997: 48, Speck and Ekins, 2002: 92).

8. TAX POWERS SHOULD BE ALLOCATED ACCORDING TO ENVIRONMENTAL EXPERTISE AND LACK OF INTEREST IN THE TAX BASE

Compliance with the demanding tax design required to deliver environmental effects recommends tax authorities with environmental expertise to address the technical complexity involved. This requirement tends not to advise having the Ministry of Finance in charge of the tax design and management of environmental taxes. Such allocation of tax powers is also negative for the environmental effectiveness of these taxes due to the interest the Ministry of Finance holds in keeping the tax base stable to sustain the revenue stream. However, the allocation of tax powers to the Ministry of Environment does not guarantee *de per se* that such a shortcoming is avoided. Designs that neutralise such interest in the tax base, such as the allocation of tax revenues to a third entity or its recycling back to the taxpayers according to precise conditions, proved helpful in delivering environmental effectiveness. However, other kinds of interests, rather than revenue-related ones, might also keep the entity in charge of the tax from introducing in the law or in institutional practice the design features of environmental taxes. If those remain unaddressed the environmental effectiveness of the tax is due to be negatively affected.

The taxes experiencing the lowest inclusion of the design features of environmental taxes, namely the Portuguese energy tax and the Swedish CO_2 tax and energy tax, were designed and managed by the Ministry of Finance. These taxes were very similar to traditional excise taxes. They were raised on measured units of fuels or energy products, missed precise overall environmental objectives and used environmental criteria poorly. Their design was not aimed at behavioural change but at raising revenues, with low tax awareness and scarce reward of tax avoidance strategies entailing pollution prevention or abatement, tax rates unable to induce significant behavioural change (neither fuel switch nor energy efficiency) and tax charging on the best-payers. This design, which seems to be associated with low environmental effectiveness, might have followed from lack of technical expertise by these entities or their interest in keeping stable revenue sources.

The taxes showing the highest inclusion of the design features of environmental taxes, and consequently the most technically complex design, which were also those displaying the highest environmental effectiveness, namely the Swedish sulphur tax and NO_x charge, were designed by the Ministry of Environment. In the sulphur tax, the Ministry of Finance managed the tax, whereas in the NO_x charge this was also the responsibility of the Ministry of Environment. In the first case the split between the entity designing the tax and the entity keeping its revenues together with the environmental expertise and attributions of the Ministry of Environment were expected to lead to a strongly regulatory tax design. This was evident in the strong inclusion of environmental criteria, both in the setup of quantified environmental objectives and in the ruling of the tax design. Such regulatory design was further confirmed by the institution of tax awareness and the reward of tax avoidance strategies corresponding to pollution prevention and abatement, as well as tax payments that not only mirrored accurately the evolution of environmental damages and induced strong behavioural change (following the use of a tax base that mimicked polluting emissions and tax rates referred to pollution abatement costs) but also were imposed on the polluters best able to take such change.

The NO_x charge displayed a degree of inclusion of the design features of environmental taxes similar to that of the sulphur tax, with just a slight difference, namely it was raised directly on polluting emissions. Its administration by the Ministry of Finances never raised an issue since this entity did not keep an interest in the tax base. The recycling of almost all the revenues back to the industry has been embedded in the tax design since the beginning. This aspect might help to understand the difference in the degree of inclusion of the referred design features, and consequent level of environmental effectiveness, experienced in the Swedish NO_x charge and in the Danish waste tax, despite both taxes being designed and administered by the Ministry of Environment (the waste tax just until 1993).

In the Danish tax, though holding environmental competences, as evidenced by the relatively good inclusion of environmental criteria in the tax design, namely in the setup of the objectives, choice of the tax base and definition of the subjective tax incidence, the regulator fell short of inducing behavioural change. This seems to have been due to the interest the regulator kept on the tax base, namely revenue interests (the Ministry of Environment) and the desire to fully exploit traditional waste disposal capacity available (the municipalities). Such interest may explain the main design features accounting for the low environmental effectiveness of the tax, namely the failure of the regulator to address institutional filters that neutralised tax awareness and eliminated the reward of tax avoidance strategies entailing reduced pollution, both at the law level (by the Ministry of Environment) and at the level of the institutional practice in waste management (by the municipalities), as well as to set tax rates sufficiently high to induce behavioural change.

CONCLUSIONS

This dissertation aimed at identifying the design features of environmental taxes, from which the environmental effectiveness of taxes follow and which allow their distinction from environmnetally-related taxes. These features have been tested against and confirmed by empirical data from three country case studies, namely the Danish waste tax, the Portuguese energy tax and the Swedish energy taxation system, the latter of which included the energy tax, the CO_2 tax, the sulphur tax and the NO_x charge. The level of inclusion of such features was positively correlated with the level of environmental effectiveness, this being assessed by the strength and speed of the production of environmental effects following the adoption of the tax. Both were highest in the Swedish sulphur tax and NO_x charge, which were clearly environmental taxes, and lowest in the Portuguese energy tax and in the Swedish CO_2 tax and energy tax, which were only environmnetally-related taxes. The Danish waste tax was in the middle of the ranking regarding both parameters. This tax was in the borderline of being considered already an environmental tax.

Environmental taxes have been identified as regulatory instruments of environmental policy aimed at steering behaviours towards pollution prevention or abatement rather than damage restoration or damage compensation. These taxes use price signals to lead towards less polluting patterns and clean technological progress. As such they are limited by the price system. Their environmental effectiveness depends on competitive markets and they are only able to guarantee the cost and not the level of pollution abatement. However, they are not limited by cost-benefit analysis since they do not pursue cost internalisation.

Like environmnetally-related taxes or pollution taxes, environmental taxes are raised on tax bases closely connected with the production of environmental damage and use price signals to highlight inefficiencies in abatement and opportunities for technological progress; whilst doing it they raise awareness of pollution costs and share responsibility for environmental damage prevention and abatement. However, these two kinds of taxes are substantially different in their objectives, consequently also in their normative design and therefore in their capacity to prevent and abate pollution. Environmental taxes aim directly and mainly or exclusively at fulfilling environmental objectives and following their targeted design, delivering such effects in a stronger and faster way than environmnetally-related taxes. The latter may include environmental concerns together with other kinds of objectives whilst ultimately aiming at raising revenues. The more demanding tax design of environmental taxes makes them more environmentally effective but also less common in institutional practice than environmnetally-related taxes.

The design features of environmental taxes include the leadership of the tax design and the objectives setup by environmental criteria, the adoption of precise (preferably quantified) environmental objectives, the institution of tax awareness, the reward of tax avoidance strategies corresponding to pollution prevention and abatement, tax payments with a 'forward looking' approach to polluting impacts and taxpayers with control over the cause *sine qua non* of pollution who still enjoy opportunities for improvement. Tax payments fulfil the referred condition when they mirror the evolution of environmental damage, following tax bases that correspond to specific polluting emissions or a good proxy for them, and induce behavioural change, by using tax rates referred to pollution abatement costs or relative polluting impacts and set at a level sufficiently high to steer behaviours towards the environmental hierarchy. Furthermore, this demanding tax design requires tax authorities with environmental expertise and without an interest in the stability of the tax base, which tends to advise the allocation of tax powers to the Ministry of Environment with the inclusion of design features that control for the interest in the tax base.

Empirical data showed the leadership of the tax design by precise environmental objectives served by one specific environmental hierarchy of behaviours and according to environmental criteria rather than political compromises is determinant for the environmental effectiveness of the tax. The institution of tax awareness and the reward of tax avoidance strategies corresponding to pollution prevention or abatement together with the level at which tax rates are set seem to make the difference between the ability of the tax to deliver relevant and quick environmental improvement or just revenues. The importance of referring tax rates to environmentally correct *effective pollution prices*, both in absolute and relative terms, rather than at partial price signals, was also confirmed. The comparative analysis showed the structure of national energy taxation that is best able to deliver environmental effects does not include general energy taxes, as required by the Energy Taxation Directive (2003/96/EC) and the proposal for its revision made by the European Commission in 2011 and as it is generalised in national institutional practice. Furthermore, it also evidenced the relevance for environmental effectiveness of not assigning more favourable treatments based on pollution amounts in contrast to common institutional practice.

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