Part B Evaluating the Efficiency and Cost-Effectiveness of Implementation

One of the working hypotheses introduced in chapter 2 is that imperfect implementation can be efficient when the policy implemented involves inefficiencies. More precisely, imperfect implementation can be efficient when it leads to a polluter's response, which is efficient and hence different from the inefficient policy requirements. Part B of the thesis analyses the question of whether implementation, and potentially a gap in implementation, was efficient in the case of the European Directive on atmospheric emissions from municipal waste incinerators (89/429/EEC). In line with the working hypothesis, this requires studying the efficiency both of the Directive and of the implementation processes that took place in France, Germany, the Netherlands and the United Kingdom.

In a European context, the feature from which inefficiencies in the policy can arise is often that of the heterogeneity of national situations, which are difficult to account for by a European policy. There are two approaches to studying the efficiency of a policy. The first is related to the subsidiarity principle, which is one of the guiding principles of European policy making, dealing with the efficient allocation of decision-making tasks between the European and the national levels. Taking this principle as a basis, the question is whether the policy objectives were defined at the appropriate policy level. For the MWI Directive, this issue is investigated in chapter 4. The second approach to studying the efficiency of a policy is to analyse the efficiency of the policy's objectives. The question here is whether the policy objectives were sufficiently differentiated to take into account the heterogeneity of contexts found on a national level. This is the issue of chapter 5. Note that the two approaches to studying the efficiency of the policy are related: the efficiency properties of the level on which the policy decision was taken are the cause of the efficiency properties of the policy's contents.

The finding that the MWI Directive is subject to inefficiencies constitutes the basis for investigating whether, in practice, Member States adapted the Directive's requirements during the implementation processes to local circumstances and by this maybe enhanced the policy's efficiency. This requires a detailed study of the specific implementation paths and of the rationales or driving forces behind the decisions the countries made. Chapter 6 focuses on the implementation outcomes, both in terms of goal attainment of the policy objectives and of the cost-effectiveness of implementation relative to the Directive's cost-effectiveness. Chapter 7 finally studies the driving factors that led to the identified policy outcomes. With this it sheds some light on the environmental impact and performance of the policy studied, i.e. how much the Directive actually contributed to the achievement of its objectives. Its primary objective, however, is to assess whether cost-effectiveness considerations drove national implementation decisions, i.e. whether the countries intentionally aimed at reducing the Directive's inefficiency during implementation.

Chapter 4 An Economic Assessment of Directive 89/429/EEC – Was the Policy Decision Taken at the Appropriate Level?

1 Introduction

In 1989 the European Council adopted two directives on atmospheric emissions from municipal waste incineration (MWI), one dealing with 'new', the other dealing with 'existing' plants. In chapter 2 it was suggested that imperfect implementation, mirrored in an implementation gap, could sometimes restore the efficiency of an inefficient policy. The necessary prerequisite for the possible efficiency of an implementation gap is that the policy is inefficient. Therefore, before analysing the implementation process of the latter Directive and the outcomes of this implementation process for four European Member States (France, Germany, the Netherlands and the United Kingdom) it is necessary to assess whether the MWI Directive itself was efficient. The present chapter studies this policy's efficiency with respect to the question of whether the policy decision was taken at an appropriate policy level and hence whether the Directive resulted in an efficient allocation of tasks between the European level and the Member States. This allocation of tasks is the issue of the subsidiarity principle, one of the guiding principles of European policy-making. To this end, we use economic criteria from the environmental federalism literature and the literature on strategic environmental policy-making, which are capable of rendering the subsidiarity principle operational.

The chapter is structured as follows. Section 2 provides context information on the EU Directive's background, as well as information on the environmental issues it addresses and the specific requirements it defines. Section 3 aims at tracing the major issues of discussion in the negotiation process of the directive. This has two objectives: it firstly aims at investigating whether the political scientists' claim that EU institutions, and in particular the European Parliament, tend to increase the ambition of regulations during the negotiation process, applies to the MWI case. Secondly, one may assume that differences in the opinions of the actors involved may indicate differences in national contexts, which might advocate for differentiated solutions. Section 4 makes the transition to the core of this chapter's analysis by identifying criteria for the optimal level of policy intervention in two streams of economic literature. In section 5 these criteria are applied to a discussion of whether, from an economic point of view, centralised European policy-making was justified and efficient in the case of the MWI Directive. Section 6 concludes.

2 The Directive's Background, Environmental Issues Addressed and Specific Requirements

2.1 The legal context

In 1989 the European Commission adopted two directives on the incineration of municipal waste, Council Directive 89/429/EEC applying to 'existing' plant and Council Directive 89/369/EEC applying to 'new' plant.[1] They present daughter directives to the so-called Waste Framework Directive[2] and belong to the group of regulations specifically dealing with certain waste disposal processes and facilities. Both aim at a reduction of atmospheric emissions caused by such facilities. They were supplemented by the 1994

[1] OJ L 203/50-54, 15.7.1989 and OJ L 163/32-36, 14.6.1989 respectively.

[2] The general framework for the overall European Union strategy for non-hazardous waste management was set by the 1975 Directive on waste (75/442/EEC), amended by Council Directive 91/156/EEC. On a general level, it establishes that waste must be disposed off without endangering human health and without harming the environment. The EU waste management strategy is complemented by the Hazardous Waste Directive (78/319/EEC, amended by directive 91/689/EEC and 94/31/EC) and by the Regulation on Waste Shipments (84/631/EEC, amended by 93/259/EEC and 97/120/EC). This framework is elaborated through two types of daughter directives, those dealing with requirements for waste disposal processes and facilities, and those dealing with specific waste streams.

Directive on the Incineration of Hazardous Waste (94/67/EEC)[3]. These three directives have more recently been amended under, and were replaced by, Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste[4]. While the discussion sometimes requires to relate also to Directive 89/369/EEC applying to 'new' plant, the focus in the following is set on Directive 89/429/EEC directed at atmospheric emissions from 'existing' municipal waste incinerators.

2.2 The rationale of the Directive: environment and health hazards related to waste incineration

In the absence of appropriate treatment of off gases, incineration of waste leads to emissions which are harmful to human health and the environment. Substances of concern include acid gases (hydrogen chloride, sulphur dioxide), oxides of nitrogen, fine particulate matter and trace elements, such as heavy metals (European Commission, 1997: iii)[5]. The most notorious substances, however, are dioxins[6] which are suspected to cause cancer and birth defects (European Commission, 1997: iii). Given that demanding control technology can significantly reduce the emissions from waste incinerators, various European countries introduced regulations directed at atmospheric pollution from such plant. However, standards varied markedly between countries, with countries such as Germany, the Netherlands and Austria implementing the most demanding emission limits. All this was the background for introducing minimum standards for air emissions at an EU level in 1989 (European Commission, 1997: iii; Hannequart, 1993: 237). The European policy makers further justified the establishment of the European policy by the fact that municipal waste incineration gives rise to emissions of some substances that may have transboundary features. This issue is discussed in more detail in sections 4 and 5.

2.3 Environmental issues addressed by the Directives

The 1989 European Directives constitute the first European approaches towards pollution to the atmosphere from municipal waste incineration plants. They established emission standards for the pollutants which are listed below (cf. Table 4.1) together with their sources and their suspected effects on human health and the environment.

As presented in the table, the pollutants regulated are notorious for health effects (carcinogenic and cardio-vascular effects), for acid pollution and for affecting the tropospheric ozone layer. The 1989 Directives were not able to specify limits for dioxins, the pollutant that had first raised the alarm. They did however address dioxin pollution indirectly by defining combustion requirements that help reducing the likely formation of dioxin compounds. Furthermore, the abatement technologies capable of meeting the emission limits which were defined for the pollutants listed in Table 4.1, as a side effect, capture dioxins to some extent (Pernin, 1997; Milhau and Pernin, 1994; http://www.environnement.gouv.fr/actua/cominfos/dosdir/DIRPPR/ dioxine/infodiox.htm, 24.11.2003).

[3] OJ L 365, 31.12.1994.

[4] OJ L 332/91-111, 28.12.2000.

[5] See also: http://www.parliament.the-stationery-office.co.uk/pa/ld199899/ldselect/ldeucom/71/7102.htm (4 December 2001).

[6] In the following, 'dioxin' is often used as short descriptor of the family of polychlorinated dibenzo-p-dioxins and the related furans.

Table 4. 1: Pollutants regulated by Directives 89/429/EEC and 89/369/EEC

Pollutant	Noxious effects	Origin
Particulate matter (dust)	Carcinogenic	Originate from combustion
Lead + chromium + copper + manganese	Toxic and carcinogenic (Cr: lung cancers, ecological effects)	Originate from the waste
Nickel + arsenic	Toxic and carcinogenic (respiratory tract cancers, ecological effects)	Originate from the waste
Cadmium + mercury	Toxic (both) and carcinogenic (only Cd: respiratory tract cancers, ecological effects)	Originate from the waste
Hydrogen chloride	Acidification of ecosystems, erosion and corrosion of building materials	Originate from the compounds in the waste
Hydrogen fluoride	Acidification of ecosystems, erosion and corrosion of building materials	Originate from the compounds in the waste
Sulphur dioxide	Acidification of ecosystems, erosion and corrosion of building materials; inflammation of bronchial system	Originate from the compounds in the waste
Carbon monoxide	Respiratory and cardio-vascular effects	Originate from (incomplete) combustion
Organic compounds	Toxic, greenhouse effect (tropospheric ozone)	Originate from compounds in the waste

Source: Milhau and Pernin, 1994; European Commission, 1997; CITEPA (http://www.citepa.org/pollution/effets.htm)

2.4 Scope and requirements of Directive 89/429/EEC

Directed at air pollution from municipal waste incineration plants, Directive 89/369/EEC applies to 'new' plant and Directive 89/429/EEC to 'existing' plant. 'Existing' plant are defined as '…those for which the first authorisation to operate is granted before 1 December 1990' (art. 1, 89/429/EEC). Consequently, 'new' plant are those for which authorisation to operate is granted from 1 December 1990 onwards (art. 1 and 12(1), 89/369/EEC). The directives apply to 'domestic refuse, as well as commercial or trade refuse and other waste which, because of its nature or composition, is similar to domestic refuse'. The waste has to be treated by 'technical equipment used for the treatment of municipal waste by incineration, with or without recovery of the combustion heat generated. Plants used specifically for the incineration of sewage sludge, chemical, toxic and dangerous waste, medical waste from hospitals or other types of special waste, on land or at sea, are excluded from this definition, even if these plants may burn municipal waste as well' (art. 1).

The two Directives define ***emission limit values*** (expressed in concentrations) which apply to individual plants (cf. Table 4.2). Emission limits differ with plant size, where emission limits are stricter for larger incinerators. By defining differentiated emission standards according to plant size, the Directives take account of economies of scale prevalent in pollution abatement of municipal waste incinerators.[7] Furthermore, as far as existing plant are concerned, upgrading deadlines for various sizes of plant were defined, with shorter time frames for larger incinerators. In the following, the focus is mainly on 'existing' plant requirements which constitute the basis for the empirical investigation of the implementation processes in four Member States in subsequent chapters.

The Directives distinguish between nominal capacity categories of plant. Capacity refers to the sum of incineration capacities of furnaces of which the plant is composed. The requirements for 'new' plant are presented in annex 4.A (Table 4.A.1). Requirements similar to those for 'new' incinerators were eventually to be met by 'existing' incinerators (art. 2). However, for plants of a capacity below 6 t/h the Directive established additional interim requirements. The 6th and 7th columns in Table 4.2 define these transitional arrangements smaller incinerators had to meet by 1 December 1995. The 2nd to 5th columns define the emission limits plants of a capacity greater than 6 t/h had to comply

[7] This issue will be further discussed when dealing with cost effectiveness in chapter 5.

with by 1 December 1996, and the final emission limits incinerators of a capacity below 6 t/h had to comply with by 1 December 2000.

Table 4. 2: EU emission limits in mg/m3 for existing incinerators

Compliance deadline	**1 Dec. 2000**			**1 Dec. 1996**	**Transitional arrangement from 1 Dec. 1995 to 1 Dec. 2000**	
Overall capacity group	**< 6 t/h**			**> 6 t/h**	**< 6t/h**	
Sub-capacity class	**< 1 t/h**	**1 t/h- < 3 t/h**	**≥ 3 t/h**	**> 6t/h**	**<1t/h**	**1- < 6t/h**
Dust	200	100	30	30	600	100
Pb+Cr+Cu+Mn	-	5	5	5	-	-
Ni+As	-	1	1	1	-	-
Cd+Hg	-	0.2	0.2	0.2	-	-
HCl	250	100	50	50	-	-
HF	-	4	2	2	-	-
SO_2	-	300	300	300	-	-
CO	100	100	100	100	100	100
Organic Compounds	20	20	20	20	-	-
Residence time of combustion gases	The gases resulting from the combustion of the waste is raised, after the last injection of combustion air, and even under the most unfavourable conditions to a temperature of at least 850 °C, for at least two seconds in the presence of at least 6% oxygen				The gases resulting from the combustion of the waste is raised, after the last injection of combustion air, and even under the most unfavourable conditions, to a temperature of at least 850 °C, in the presence of at least 6% oxygen, for a sufficient period to be determined by the competent authorities	

Emission limits in mg/m3 for existing incinerators. Standard conditions: 273 degrees K, 101,3 kPa, 11% Oxygen or 9% CO_2 and dry gas. - Source: Art. 2, 3 and 4 89/429/EEC

In line with the European Treaty's article 130t (new numbering article 176), the Directives established the possibility that Member States maintain or introduce more stringent protective measures. National competent authorities may lay down emission limit values for additional pollutants when they consider this appropriate because of the composition of the waste to be incinerated and of the characteristics of the incineration plant. This may also refer to dioxins and furans (Art. 3).

Appropriate measurements and verifications are a pre-requisite for ensuring compliance with emission limit values. The Directives specify ***requirements for measurement of pollutants*** (i.e. monitoring requirements) and define which measurement results are considered as compliance. These are presented for 'new' and 'existing' plant in annex 4.A (cf. Table 4.A.2 to Table 4.A.5). Not only emission limit values, but also monitoring requirements are stricter for larger size plants. And as far as 'existing' plant are concerned, again, weaker interim requirements were specified for plants of a capacity below 6 t/h, while final monitoring requirements for existing plants are identical to those for new plants.

As pointed out in chapter 2, European Directives are not directly binding in the Member States, but need to be transposed into national law. The Directives consequently specified ***transposition requirements***, demanding Member States to bring into force the laws, regulations and administrative provisions necessary to comply with the 1989 Municipal Waste Incineration Directives by 1 December 1990. Member States were finally required to communicate to the Commission the texts of the provisions of national law, which they adopted in the field governed by the two Directives (Art. 10, 89/429/EEC and art. 12, 89/369/EEC).

Summing up, it is obvious that the 1989 Directive constitutes a classical piece of 'command-and-control' regulation with a limited differentiation of emission limit values according to capacity classes. This aspect is important for an evaluation of the efficiency of this policy's contents and of the cost-effectiveness of the implementation outcomes across the Member States and will be further discussed when introducing country heterogeneity in the following chapter.

3 The negotiation of the Directive: cost considerations seem to have played a major role

The couple of 1989 Directives was issued under article 130s (new numbering article 175) of the Treaty. This bases the Directives on the article 130r (new numbering 174), defining a legal basis for European environmental policy, which was introduced under the Single European Act (SEA) in 1986. Furthermore, article 130s (new numbering article 175) establishes that the Directives' negotiations fell under the co-operation procedure (article 189c, new numbering article 252). The development of environmental Directives, generally, involves a first proposal, which is submitted by the European Commission to the Council of Ministers. Further institutions involved in the negotiation process are the European Parliament, the Economic and Social Committee and the Committee of the Regions.

Tracing back the major issues of discussion in the negotiation process of the Directive might shed light on two issues. Firstly, it allows to investigate whether the claim of some political scientists' that EU institutions, and in particular the European Parliament, tend to increase the ambition of regulations during the negotiation process, applies to the MWI case. Secondly, differences in the opinions of major actors involved may shed light on differences in national contexts whose accommodation under the policy might require differentiated objectives or discretion on a local level and, if not taken into account, might entail implementation problems. Ideally, the 1989 municipal waste incineration Directives' development would have been traced from the first Commission proposal to the text finally adopted, with an emphasis on the interests of, possible conflicts between, and coalitions of the different Member States. However, no official documentation is available on the negotiation processes that involved the Member States, and the process of the evolution of the Directives can hence not be traced back in detail. Documents available are the Commission's initial proposal (COM/1988/71/Final a and b), the comments made by the Economic and Social Committee (88/C318/02), the European Parliament's Opinion[8] and the Directive finally adopted. What is said about the Member State's interests below is based on interviews undertaken in France[9].

The focus in the following presentation is set on the evolution of those requirements during the negotiation process of the Directives that are the most decisive for the strictness of environmental requirements and for implementation costs. These are the differentiation of emission standards according to plant capacity, the differentiation of compliance deadlines, requirements relating to the application of 'best available technology' (BAT) versus 'best available technology not entailing excessive cost' (BATNEEC), and the definition of dioxin abatement requirements. (For a more concise survey of the evolution of the Directives 89/429/EEC and 89/369/EEC cf. Table 4.B.1 for 'new' plant requirements and Table 4.B.2 for 'existing' plant requirements in annex 4.B).

3.1 Differentiation of compliance deadlines

The initial proposal of the Directive had foreseen to leave existing plant a transitional period of 5 years to reach intermediate emission limits and a time span of 10 years to upgrade plants to meet the same emission limit values as new plant. Both the Economic and Social Committee (ESC) and the European Parliament criticised these deadlines foreseen to upgrade existing plant as too long, and Parliament suggested to reduce the transitional period to 3 years, and the deadline for meeting the same emission limit values as new plants to 5 years. None of these suggestions was retained, although deadlines with

[8] OJ C 75/4-11, 23.3.1988; OJ C 318/3-6, 12.12.1988; and OJ C 69/215-223 a and b, 20.3.1989 respectively.

[9] The interviews were carried out in the course of the European research project 'The implementation of EU Environmental Policies: Efficiency Issues' (IMPOL).

respect to existing large plant (> 6 t/h) were eventually shortened to 6 years. Interviews undertaken in France suggest that this country, during the negotiation process of the Directives, participated in a coalition that tried to prolong the deadlines for compliance. It also tried to introduce a range of deadlines for plants of different capacities, so that investment costs could be spread over a longer period (Schucht, 2000; Lulofs, 2001). This does not come as a surprise when considering France's specific plant park (cf. chapters 5 and 6), which included a way above-average number of plants, many of which were of low capacity.

3.2 Differentiation of emission limit values

Next to the differentiation of compliance deadlines, the Directive also differentiates emission limit values according to plant capacity classes. This differentiation was increased in the Directive's adopted version relative to the initial proposal. While the initial proposal distinguished between capacity classes of < 5 t/h and > 5 t/h as far as eventual emission limit values are concerned, the final version distinguishes between three capacity classes: < 1 t/h, 1 t/h to < 3 t/h, and ≥ 3 t/h. Transitional requirements were initially differentiated between plants of a capacity of < 1 t/h, ≥ 1 t/h to < 6 t/h, and ≥ 6 t/h. Here, the final version distinguishes transitional requirements only between plants of a capacity below 1 t/h and those of a capacity between 1 and < 6 t/h.

The introduction of additional differentiation for lower capacity classes with respect to final emission limit values also implies a weakening of certain emission limit values, specifically those for plants with a capacity below 1 t/h. All in all, the final version of the Directive takes economies of scale in abatement more strongly into account than did the initial proposal. Whether this development was due to the influence of France could not be established.

3.3 Scope and strictness of emission limits

A further point criticised by the ESC and Parliament was that emission limits in general were not demanding enough, given that some Member States applied stricter targets. In particular, emission limits set for dust and certain heavy metals were considered as too weak. Furthermore, the range of pollutants covered was considered as too restricted. Both institutions suggested enlarging the range of pollutants covered by introducing further limit values for additional heavy metals. Amongst all these suggestions only those concerning stricter dust emission limit values found their way into the final version of the Directive. Even though the suggestions were not exactly translated into the adopted legal text, emission limit values were tightened, at least for large plants.

A central discussion has centred on the issue of dioxin emissions. The 1989 Directives were not able to specify emission limits values for dioxins, the pollutant that had first raised the alarm. The initial Commission proposal demanded the Council, following suggestions from the Commission, to establish measurement requirements for dioxins and furans at a later stage, as soon as the state of the art would allow this. According to the Commission, this was motivated by the inability of existing techniques to measure to the required precision the very small amounts (10^{-9} gram per cubic metre of air) which were of concern (European Commission, 1997: iv). In this context it should be noted that domestic German and Dutch legislation to be implemented during the early 1990s did specify dioxin emission limits for municipal waste incineration plants. The Commission's proposal was criticised by the ESC, which suggested to follow the example of German regulation and to directly integrate measurement requirements for dioxins and furans into the Directive. This was however not done and, what is more, the suggestion of a subsequent issuing of measurement requirements was dropped as well. Instead, a much vaguer formulation was adopted, which leaves countries with more discretion: Member States were given the possibility to introduce emission limit values for additional pollutants, including dioxins and furans, if they thought that necessary. As will become apparent in subsequent chapters, although no limit was specified for dioxins in the

Directive, national responses to dioxin emissions have proved particularly important to the implementation outcomes achieved.[10]

Dioxins were nevertheless taken indirectly into account in that the 1989 Directives defined so-called operational conditions, which specify combustion conditions in a way that guarantees a reduction in the likely formation of dioxin compounds (European Commission, 1997: iv; COM(2001)593 final). Given that, in principle, dioxins can be destroyed by way of incineration the combustion temperature is an -although less effective- alternative to dioxin abatement technology. For this, the process requires temperatures of over 850°C. Destruction of large amounts of contaminated material are said to require even higher temperatures (1000°C or more).[11] Interviews undertaken in France revealed that these combustion requirements had been an issue of discussion during the Directives' negotiation. While a requirement of raising the temperature to 1000 °C (for a period of 10 seconds after the last injection of combustion air) had been discussed, consensus could only be reached on the requirement to raise the temperature of combustion gases to 850 °C (during 2 seconds).

3.4 Progressive adaptation to advanced technology

An aspect with respect to which the Directive was weakened during its negotiation process are the technological standards required over time. The initial requirement to progressively adapt existing plant to 'Best Available Technology' (BAT) was dropped in the adopted version. Instead, 'Best Available Technology Not Entailing Excessive Costs' (BATNEEC) standards are required for municipal waste incineration plants: Directive 89/369/EEC states that the known pollution abatement techniques can be applied reasonably economically in new incineration plants, while Directive 89/429/EEC considers that in existing incineration plants they can be implemented on a gradual basis, bearing in mind the technical features of the plants and the advisability of not entailing excessive costs.[12]

3.5 Conclusion

Taken all this together, during the negotiation process of the Directive the forces which payed more attention to implementation costs than to the strictness of the policy measure seem to have gained the upper hand over those forces which tried to make the Directives stricter (especially the European Parliament and the ESC). A stronger *differentiation of emission limit values*, in particular for smaller capacity plant, takes into account the existence of economies of scale, by this easing implementation costs especially for those countries where small MWI plants play an important role. In the same direction go the *longer compliance deadlines* accorded to small plant, giving countries more time to search for alternative treatment options or for reorganising the treatment of municipal waste on a local scale. And finally, the requirement to apply *BATNEEC* instead of BAT allows countries to give a stronger argument to their compliance costs. Whether the decision to not define *emission limits for dioxins and furans* was rather driven by cost considerations or by technological ones (the limits to measurement equipment available

[10] Dioxin pollution originating from waste incineration has remained a focus of interest, also on an EU level. As soon as the early 1990s, discussions about a new waste incineration directive started, leading to a first draft in 1994. Negotiations took place during several years and eventually, on 4 December 2000, led to the adoption of the new waste incineration directive (2000/76/EC, OJ L 332/91, 28.12.2000). In contrast to the 1989 Directives, amongst other changes, the new legislation sets emission standards for dioxin pollution also for municipal waste, which require the installation of respective abatement and measurement technology.

[11] Source: http://www.who.int/int-fs/en/fact225.html, fact Sheet No 225, June 1999 (4 December 2001).

[12] BATNEEC is also required because the 1989 European Directives were rooted in the Council Directive on the combating of air pollution from industrial plants (84/360/EEC, OJ L 188). This Directive stipulates that authorisation may be issued only when all appropriate preventive measures against air pollution have been taken, including the application of BATNEEC. It also requires that the Council, acting unanimously on a proposal from the Commission, shall, if necessary, fix emission limit values based on BATNEEC and suitable measurement techniques and methods. For existing plants the same directive demands Member States to apply policies and strategies for the gradual adaptation to BATNEEC.

at the end of the 1980s) is open. One can nevertheless assume that controlling combustion conditions is less costly than adding a dioxin abatement step to plants.

This general result is interesting in relation to what was said in chapter 3. The political science literature presented there had suggested several reasons for why EU policy making is likely to cause implementation gaps. One of these had to do with a tendency of formulating over-ambitious regulations, which was supposedly increased by a strengthened role of the European Commission, the Parliament and the Economic and Social Committee relative to the Council of Ministers under the co-operation (and even more under the co-decision) procedure. Under the co-operation procedure the adoption of a directive requires unanimity in the Council when Parliament rejects the common position. It was indeed shown that in the negotiation of the 1989 municipal waste incineration Directives the ESC and Parliament did push for more ambitious requirements compared to the first Commission proposal, however, the majority of these more ambitious suggestions were not adopted.

The findings about the evolution of the Directive from its first proposal to the adopted version raise a couple of questions: Has the fact that more demanding requirements were dropped resulted in a policy whose implementation was an undemanding issue? Or did meeting the Directive's emission limit values, although clearly less strict than the domestic policies existing in a number of Member States, pose serious problems for other Member States, which could explain why tighter standards were not open to consensus? These questions will be traced for four Member States, France, Germany, the Netherlands and the United Kingdom throughout the following chapters.

4 Economic criteria for assessing the adequate level of policy decisions

It was suggested earlier that an implementation gap, resulting from imperfect enforcement, could restore the efficiency of an inefficient policy. In order to better judge the implementation paths chosen and the results obtained in four Member States in the implementation of the 1989 European municipal waste incineration Directive it is necessary to develop an idea about the efficiency of this policy itself. One way of assessing the efficiency of a policy is to analyse in how far the policy decision is in line with the subsidiarity principle. Recall that this principle, in the European context, suggests to decentralise policy making wherever lower government levels are better suited to deal with a specific policy problem, while no specific criteria were provided to judge when this is the case. Both the application of this principle and an evaluation of a specific policy from its point of view require optimality criteria which put the procedure of the principle of subsidiarity into a concrete form and which set a benchmark against which to assess the policy.

The question is that of the efficient allocation of competences among different levels of government. Criteria for such an allocation, applied to the case of environmental policy, were suggested by the literature frequently subsumed under the title 'environmental federalism', which belongs to the broader literature on fiscal federalism.[13] A second strand of economic literature discussing the optimal government level of policy making is the literature on strategic environmental policy making. This section briefly reviews this literature in order to identify some economic criteria that make the subsidiarity principle

[13] Literature discussing the principle of subsidiarity in the European context sometimes focuses on further criteria, such as equity and accountability (see for example Begg, et al., 1993). Theses criteria are not further discussed here, as the focus is on economic efficiency. Some authors derive criteria explicitly from institutional economics, e.g. Garbe (1996) who puts a special emphasis on transaction costs. The issue here are rather decision making rules and the related institutional framework that can best deal with the revelation problem of preferences for public goods, the social choice process to take these into account, and the management of social costs. His results, however, do not differ much from those presented here. Other authors apply a political-economy point of view (e.g. Alesina and Wacziarg, 1999) and emphasize social-choice arguments.

operational from an economic perspective.[14] It also shortly presents the political objective of a level playing field. The following section then discusses the Directive's efficiency.

4.1 The point of view of the environmental federalism literature

Following Oates (1999: 1120-1121), fiscal federalism encompasses the whole range of issues which relate to the vertical structure of the public sector. It studies both in normative and positive terms the roles of the different levels of government and the ways in which they relate to one another. It asks which functions and instruments are best centralised and which are best placed in the sphere of decentralised levels of government. The theory is often directly linked to the subsidiarity principle.

As a general result, this literature shows that the answer to the question of the optimal policy level is not simply centralised versus decentralised policy making. Instead, there is a case for tailoring programmes to the circumstances of individual jurisdictions, while at the same time there are other considerations that require centralised measures (Baumol and Oates, 1988). This case specificity in the optimal level of decentralisation arises from a trade-off between keeping the policy as close to the voters' preferences as possible and the need to correct for externalities which may spill beyond the boundaries of a given unit of political decision-making (Alesina and Wacziarg, 1999: 18).

In which cases higher or lower political levels are better suited to deal with specific policy problems is discussed in the following, first from a general point of view, and then specifically for environmental policy issues.

4.1.1 General pros and cons of centralisation and decentralisation

The fiscal federalism literature discusses the provision of public goods in general. It establishes that 'in the absence of cost-savings from the centralised provision of a [local public] good and of inter-jurisdictional externalities, the level of welfare will always be at least as high (and typically higher) if Pareto-efficient levels of consumption are provided in each jurisdiction than if any single, uniform level of consumption is maintained across all jurisdictions' (Oates, 1972: 54). With this, in terms of economic efficiency, there is a presumption in favour of a decentralised provision of public goods with localised effects. Applied to policy formulation the idea is that decentralised policy making is more likely to provide standards tailored to local circumstances in an optimal way than centralised policy making.

An interesting aspect of this literature is that it is set in a context of likely real world cases. In a setting of perfect information, both central and decentralised levels could reach efficient results. In such a world the centralised level could, for example, always replicate the outcome of a decentralised one and thus provide a set of differentiated local standards that maximise overall social welfare (see for example Laffont and Pouyet, 2000: 3). These authors note that one must introduce a degree of 'incompleteness' to create a trade-off between centralisation and decentralisation. It is generally claimed that this 'incompleteness' in the real world is mirrored by two constraints to a central provision of locally optimal patterns of output: ***information asymmetries*** and ***political constraints***.

With respect to the first point, local governments are presumably closer to the people of their jurisdictions and possess better knowledge of both local preferences and cost conditions, e.g. practical conditions affecting the implementation of policies, than a central agency. And even if the centralised government level could in principle gather the necessary information, it might have to provide local jurisdictions with substantial incentives to reveal the necessary information. In practice, informational requirements may be prohibitive for the central government to provide the optimal policy. This fact is sometimes referred to as the 'costs of the size of a jurisdiction': they emerge from the

[14] There is also a political science discussion about federalism and subsidiarity which uses different criteria. These are, for example, national interests, state sovereignty, minority rights, political participation and legitimacy (see for example Shapiro, 1996; Endo, 2001; Inman and Rubinfeld, 1998; Sinnott, 1994).

heterogeneity of preferences within the population, assuming that the extent of heterogeneity increases with the size. With respect to the second point, political pressures or even constitutional constraints may limit the capacity of a central government to provide different levels of public services in different jurisdictions, or to require different environmental standards, and may instead require some form of equal treatment across localities. Consequently, central level policies tend to involve some degree of uniformity (Oates, 1999: 1123; Begg et al., 1993: 7 and 40; Alesina and Wacziarg, 1999: 15 and 18).

As a first result one can therefore note that in the absence of spillovers and of cost savings from central provision the literature suggests decentralised policy making because policies need to be tailored to local conditions and differentiation of centrally determined policies may not be equally efficient as decentralised provision.

However, decentralised solutions are not always believed to provide the optimal policy. Where policies cause spillover effects or are subject to economies of scale, efficiency considerations suggest that centralised decision making or at least some form of co-ordination are better suited to provide optimal solutions. ***Spillover effects*** characterise a situation where policies implemented in one jurisdiction have impacts on the welfare of the population in other jurisdictions. Here, economic theory argues for centralised decision making (or some form of co-ordination). Illustrating this at the example of pollution, the reason is that the jurisdiction in which pollution is caused is likely to only internalise the externalities of the pollution in its own borders, but not to take into account the costs it inflicts on others. A second reason making a centralised provision of solutions more efficient are ***economies of scale***. The basic idea is that, for example in a federal system, administration and co-ordination costs increase with the number of decentralised decision units, and that the centralisation of the administration can reduce these costs. With respect to environmental policy making, economies of scale may hence arise from an increased ability to co-ordinate. They may also be informational and consist for example in the provision of R&D (Begg et al., 1993; Garbe, 1996).

But can centralisation and coordination be considered as substitutes? To be more precise, can, where centralised policy making is suggested, co-ordination between jurisdictions reach similar results, given that the advantages of centralisation are essentially related to a co-ordination of individual policies? Theoretically, co-ordination or bargaining could provide similar results as centralised policy making. Nevertheless, in practice, co-ordination between jurisdictions may be problematic for reasons of reaching agreement and implementing and monitoring such agreements. The difference between centralisation and co-ordination lies in the fact that under the latter, jurisdictions retain the right to determine policies as they wish, subject to negotiation with others, while under centralisation they can be overruled. It is also argued that agreements might be less credible, giving jurisdictions more possibilities to behave as free riders. The credibility of coordination will therefore depend on the possibilities of monitoring compliance with agreements and on available sanctions (Begg et al., 1993: 38).

In this context it should however be noted that for credibility of centralised policy involving many countries one needs to assume that a central planner would be empowered to enforce national policies, which is not trivial. This is not the case in the greenhouse gas case, and it is only in a limited way the case in European policy enforcement. Therefore, there could be no way out to negotiation and co-ordination even where centralised policy making is in order.

Summing up, decentralisation is justified by heterogeneous local preferences and different local conditions (resource endowments). But if there are externalities to other jurisdictions, these may result in an under-provision of a public good. Using Alesina and Wacziarg's (1999: 18) words, as a general result provision should therefore be allocated to the level of government with which the frontier of the externality corresponds.

4.1.2 Optimal government levels corresponding to specific environmental problems

In order to assess in a stylised way which level of government corresponds best to externalities caused by environmental pollution it is often suggested to distinguish several benchmark cases for which the policy suggestions vary (see for example Oates, 2001; Garbe, 1996). These benchmark cases refer to different types of pollutants in terms of the geographic extent of the externalities they cause. The discussion therefore is based on the notion of a spatial dimension of the provision of public goods, where competences should be delegated to that jurisdiction which best fulfils the criteria of economic efficiency (Garbe, 1996). Oates (2001: 2-5) distinguishes three benchmark cases.

i. environmental quality is a pure public good (global effects)
ii. environmental quality is a purely local public good
iii. environmental quality is subject to spillover effects (limited trans-frontier pollution)

While the first two constitute extreme cases, the third can be considered as an intermediate case where pollution entails local effects as well as externalities to other, generally neighbouring localities, so-called spillover effects. It is immediately obvious that the differentiation between the first and the third case is a matter of scale. When talking about local pollutants, the assumption is that their negative external effects are limited to a local political jurisdiction, i.e. that they cause externalities only within this jurisdiction's borders, while spillover effects affect several jurisdictions. This is obviously only an approximation. Nevertheless, it allows to structure the issue and to develop criteria for optimal policy-making, both in terms of the allocation of competences -given that levels of government are primarily distinguished by the extent of their geographic jurisdiction (Alesina and Wacziarg, 1999: 17)- and in terms of the definition of standards. For the sake of completeness, optimality criteria are presented for the three benchmark cases, although only cases 2 and 3 are relevant for an assessment of the 1989 European municipal waste incineration Directive.

The following discussion first presents the three cases in their simplest form[15] and then considers some complications. Simplicity reasons suggest to imagine a system with two levels of government. A central government, which may establish environmental standards to be met in each jurisdiction the country is made up of, and local governments, which make policies for their own jurisdiction. This system can be easily applied to the European context, considering EU level policy making, on the one hand, and Member State policy making, on the other hand.

(i) Benchmark case I: Environmental quality is a pure public good

Environmental quality being a pure public good for a country does not mean that environmental quality is the same in all locations. Instead it means that environmental quality is a function of the aggregate level of emissions, which is the sum of the emissions from all sources in the country. In this setting, a unit of emissions has the same effect on the national environmental quality, no matter in which jurisdiction it takes place. Emissions from different jurisdictions in a country are therefore perfect substitutes.

This can be formalised in the following simple way

$Q_i = f_i(E)$,

where:

Q_i is a vector of overall environmental quality in a jurisdiction with $i = 1, 2, \ldots n$

$E = \sum e_i$ is the aggregate level of emissions

[15] The presentation of the three benchmark cases is based on Oates (2001).

Decentralised jurisdictions here do not have control over the level of environmental quality within their own boundaries, and emissions caused in one jurisdiction spill over to other jurisdictions and affect the overall environmental quality. For this case the environmental federalism literature suggests that environmental standards should be determined on a central government level.[16]

(ii) Benchmark case II: Environmental quality is a local public good

If environmental quality is a purely local good, emissions cause externalities solely within the jurisdiction where they are emitted. The environmental quality within a jurisdiction *i* then depends only upon the level of emissions in that jurisdiction. This can be formalised as

$Q_i = g_i(e_i)$,

where:

Q_i is a vector of environmental quality in jurisdiction i

i = 1, 2, … n denotes the *i*th jurisdiction

e_i is the level of emissions in jurisdiction i

At first sight, this suggests that standards should be determined in a decentralised way, i.e. at the level of each jurisdiction, and optimal standards are likely to entail differing levels of environmental quality across jurisdictions.[17] Possible complications with respect to the optimality of decentralised standard setting are discussed below (cf. section 4.1.3).

(iii) Benchmark case III: Environmental quality is a matter of spillover effects

In this case, emissions entail both local pollution and external effects on neighbouring jurisdictions. The level of environmental quality in jurisdiction *i* then depends on the specific pattern of emissions in all *n* jurisdictions. But note that here emissions in different jurisdictions, unlike to case I, do not just add up to the jurisdiction's environmental quality. Instead, more complex functional relationships are at play that can be written as

$Q_i = h_i(e_1, e_2, \ldots, e_n)$,

where:

Q_i is a vector of environmental quality in jurisdiction i

i = 1, 2, … n denotes the *i*th jurisdiction

e_i is the level of emissions in the n jurisdictions

It is generally suggested that the existence of such jurisdictional externalities, in a setting of decentralised decision making, leads to distorted outcomes, potentially involving excessive pollution. This is so because the government of jurisdiction *i*, when setting standards, will generally not pay attention to the environmental costs caused by its emissions outside its boundaries. While the efficient pattern of standards here will generally imply differing levels of environmental quality across jurisdictions, it will also require some kind of central intervention or co-ordination between jurisdictions, given that jurisdictions are in a way at the mercy of polluters elsewhere. Local governments

[16] Next to being determined on a central level, from an efficiency point of view, the optimal standard would have to meet two further requirements. Firstly, it would be set so that the marginal benefits from environmental quality improvement summed over everyone in the country equal marginal abatement costs. Secondly, this overall standard should be met through an allocation of abatement effort between pollution sources that equates countrywide marginal abatement costs.

[17] Here the efficient standard demands the level of environmental quality for which the sum of benefits from emission reductions summed over the residents of the jurisdiction *i* equals marginal abatement costs.

here need to be induced to internalise the interjurisdictional benefits from pollution control.[18]

In practice, this most common type of pollution constitutes a complicated case. Note that spillovers may be unidirectional (e.g. upstream river pollution affects downstream users; conveying of air pollutants from one set of jurisdictions to those downwind) or reciprocal (e.g. different jurisdictions occupying the same lake), which requires different solutions. While in principle one could imagine central standard setting for the case of spillovers (first-best policy), it is unlikely that such policy would provide sufficiently differentiated policy objectives (for example because of limited information available to the central level). One might therefore consider whether co-operation between jurisdictions is a valuable alternative. When spillovers are reciprocal there exist mutual gains from trade and one might imagine that jurisdictions are willing to co-ordinate their policies so as to take account of the externalities they cause to others. In the case of unidirectional spillovers, only one party stands to gain from reduced emissions, while it is the other which will need to undertake the costly activity of reducing emissions. In order to create mutual gains from interjurisdictional co-operation, some sort of compensatory mechanism would be required. It is easily understandable that in practice such co-operative measures represent quite a challenge. As a result, given that all three alternatives -local standard setting, centralised standard setting, and interjurisdictional co-operation- are not free of problems, the literature suggests case dependent solutions. One general conclusion, however, is that some form of co-ordination is justified, and -applied to the EU case- that some form of European level intervention seems appropriate.

4.1.3 Concern over a 'race-to-the-bottom' – only a theoretical complication of benchmark case II?

Coming back to the case of purely local pollution, it was suggested that for this type of pollution decentralised standard setting seems the most adequate. As Garbe (1996) notes, this is true as long as a static approach of fiscal federalism is applied. Whether the result remains the same when a dynamic approach is applied, which analyses the consequences of decentralised policy making, is questioned by a vast literature (reviewed for example in Oates, 1999, and Oates, 2001) that argues that decentralised decision making might result in distorted outcomes typically entailing excessive levels of local pollution. The general idea is that under open economy assumptions producers located in different jurisdictions compete with each other. Eager to encourage new business and to create new jobs, local officials may be incited to behave strategically and to introduce measures to reduce costs to local business in the form of low taxes or lax environmental standards. If all jurisdictions were behaving in the same non-co-operative way, the result would be 'destructive interregional competition' and a 'race-to-the-bottom' in the environmental quality.

If this were the case, benchmark case II would require reconsideration. Decentralised decision making would then seem less appropriate and also this case of pollution would be more complicated to deal with than it seemed on first sight. Indeed, advocates of the race-to-the-bottom view claim that central level standards for environmental quality are needed to prevent excessive environmental degradation. In any case, if there were a race-to-the-bottom the environmental federalism literature would imply a choice between two alternatives, where it is not clear *a priori* which of these would entail the higher level of social welfare. One is to accept suboptimal local environmental standard setting, the other is to go for central state standard setting, which is likely to be suboptimal as well, given that centrally determined standards will tend towards uniformity (Oates, 2001: 9).

[18] The efficient standard here would equalise marginal benefits of improved quality within the jurisdiction where it is caused and within the other jurisdictions to which it spills over with marginal abatement costs of the polluting sources within the first jurisdiction.

Theoretical attempts to evaluate the risk of a race-to-the-bottom

Attempts have been made in theoretical models to establish evidence for or against the likeliness of a 'race-to-the-bottom' under interjurisdictional competition. A general result is that the model outcomes are sensitive to the underlying assumptions. A number of models which, amongst other things, assume that governments maximise the well-being of their jurisdiction's residents and have access to the needed fiscal and regulatory policy instruments show that interjurisdictional competition can yield efficient levels of public goods (e.g. Oates and Schwab, 1988; Baumol and Oates, 1988). That interjurisdictional competition for plants via environmental taxes need not be destructive is also shown in a paper by Pech and Pfaffermayr (1998) who analyze strategic environmental taxation of two countries in the presence of involuntary unemployment and endogenous location choice in an international duopoly. Contrary results are found by models assuming, for example, that governments maximise the size of their local budget (bureaucrats of the Niskanen type); that they are restricted in their access to policy instruments, or that the political process is biased (e.g. Oates and Schwab, 1988; Baumol and Oates, 1988; Nordström and Vaughan, 1999; for short surveys cf. Oates, 1999 and Oates, 2001).

An argument brought up by Markusen et al. (1993 and 1995) is worth noting. Not all kinds of investments are equally polluting as the pollution intensity varies between industries. This insight raises the question of why governments would compete for polluting industries if they have the option of specialising in clean industries and importing goods that are polluting to produce. In a two-region model they argue that, if the disutility of pollution is high enough, governments would compete increasing their environmental taxes or standards to drive the polluting firm from the market. Neglecting the possibility that the government, for some reason, might have no alternative to competing for polluting industries, the only rational reason to attract the dirty industry would be that the income gain is large enough to offset the pollution costs. In the context of such NIMBYism (not-in-my-backyard phenomenon) Nordström and Vaughan (1999) discuss the possibility of the opposite of a race-to-the-bottom: a race-to-the-top. If regions or countries are able to pick and choose between industries, governments may be inclined to bid up standards and set their policies with a view to deterring polluting industries in favour of clean industries. Finally, there are also authors that claim that in a dynamic model strict and properly drafted environmental regulations can trigger innovation and thus spur a highly competitive industry (Porter and Van der Linde, 1995). Pushing this argument further, governments could have an interest in entering into a race-to-the-top rather than into a race-to-the-bottom.

All in all the large literature modelling local public decision making in a setting of competition amongst governments is inconclusive on whether local governments will seek to promote the well-being of their residents by also caring about local environmental quality, or whether they will sacrifice the environment for economic development.

Empirical studies of the race to the bottom

While empirical studies on the possible magnitude of such distortions are lacking, Oates (2001) notes that empirical evidence of standard setting in the US does not give any compelling support for the race-to-the-bottom view. He also reviews studies analysing whether the stringency of local environmental regulation has an impact on plant location decisions both in the US and the international context, and finds both positive and negative evidence. As the author notes care is needed when interpreting the results. Even if firms in their location decisions react to the stringency of environmental regulation, this is no proof that states or localities actually use environmental measures as a competitive instrument. Other authors report empirical evidence suggesting that the impact of environmental regulation in determining the location of (polluting) industries is small.

Evidence in this line is provided by Zarsky (1999) who studies the impact of foreign direct investment on the environment and, more in particular, whether and how environmental regulation influences industry location, by reviewing various statistical and case studies. He finds that environmental standards and/or abatement costs have not made a significant difference to firm location decisions. The argument that pollution control is not a critical cost factor in location decisions is also supported by Wheeler (2001). This author claims that these costs do not provide OECD industries with strong incentives to move to developing countries. More in general, he tests the race-to-the-bottom argument by studying the evolution of particulate emissions and of regulation directed at airborne pollutants in the US and in China, Mexico and Brazil, three countries from whom US industrial imports have been expanding for a long time. By showing empirically that air quality in all four countries has improved over the last 10 to 15 years, and that governments display a consistent tendency to tighten regulation with economic growth, he provides evidence in favour of a race-to-the-top rather than for a race-to-the-bottom. As far as evidence of a NIMBY attitude reflected in policies is concerned, Nordström and Vaughan (1999) find examples only with respect to hazardous waste. More precisely they refer to US state policies that document an upward drift in hazardous waste disposal taxes.

While there is so far little evidence of the empirical relevance of the race-to-the-bottom concern, most authors agree that more empirical evidence is needed. Ulph (1996) furthermore argues that the likelihood of environmental policy affecting plant location just by considering abatement costs as a proportion of total costs might be too simplistic. In his opinion, what is of relevance is the proportion which differences in abatement costs form in differences in fixed and variable costs between different locations. Moreover, it is differences in profits, and not just in costs, that matter, and these will depend on the endogenous degree of competitiveness in different markets. Finally, because of inter-industry linkages demand side considerations will affect location choices as well.

Given the existing empirical evidence which lies somewhere between the two extreme cases race-to-the-bottom or race-to-the-top, it is important to note that many models suggesting that governments might weaken environmental policies with the aim to attract business assume that the governments do not take into account local preferences for environmental protection and that they are not penalised for this. Political economy models taking re-elections into account (where voters can punish the government if it does not take their preferences into account) would give a different picture. Krutilla (1999) reviews a number of papers that study potential rent-seeking behaviour over property rights of the agents affected (industry lobbies and pollution-impacted parties).

Synthesis

Summing up these findings, one can assume with Oates (2001: 24) that decentralised standard setting in the case of local pollutants remains compelling. Nevertheless, this author further suggests that if the aim was to counter the concern about a potential danger of a race-to-the-bottom, the central level could play the role of giving guidance and information, for example by offering a menu of standards and a choice of policy instruments. In the same line of argument Baumol and Oates (1988) state that minimum standards could in certain cases be useful as guidance for regions. In sub-section 4.2 below we will however show that there exists a different stream of argumentation which gives further reasons for central government intervention in the case of local pollutants.

4.1.4 Conclusions from the environmental federalism literature

The environmental federalism literature has suggested that the optimal government level for policy making -i.e. the optimal size of the jurisdiction in which standards are set- crucially depends on the type of pollutants addressed in terms of their spatial effects. As a general rule, the jurisdiction should be of sufficient size to internalise the great bulk of the pollution. When pollution is characterised by ***cross-border spillovers*** (global or trans-frontier pollutants, such as greenhouse gases or SO_2 respectively), some centralisation of policies is needed to internalise the externality -because the local decision makers would not take into account the preferences of the neighbouring jurisdictions- and the gains from centralisation must be traded-off against the costs from imposing less differentiated policies about heterogeneous groups. Bargaining between jurisdictions might be an alternative to combine the need for co-ordination with the optimality of locally differing standards, but the feasibility and credibility of Coasian bargaining solutions will depend on the specific conditions.

Assuming that the 'race-to-the-bottom' issue as well as the impact of environmental regulation on firms' location decisions are empirically not of much relevance, ***local pollutants*** suggest decentralised policy-making. Central decision-making might lead to solutions less tailored to local preferences. Generalising this discussion, there is little support for a harmonisation of standards and in any case not for undifferentiated standards that leave no further discretion to individual countries.

4.2 Strategic environmental policy concerns

There is a second strand of economic literature which takes a different angle to the question of whether central environmental policy making is desirable or not. This literature deals with environmental policy and trade concerns and establishes the possibility that environmental policy could be used strategically to further the interests of domestic producers. With this it establishes further reasons for an implication of the central level into policy making in the case of local pollutants. This literature has been reviewed amongst others by Ulph (1996), Ulph (1999) and Rauscher (1999), on which the following presentation is based. The literature distinguishes cases where a central intervention into environmental policy setting might be justified in order to avoid competitive distortions. As a specific case, it deals with the more political concern of an 'ecological dumping', describing the concern that governments might have incentives to relax environmental policies in a process of legislative competition in order to improve the competitive advantage of domestic producers.

Ulph (1999: 433) distinguishes three reasons why governments may act strategically in setting their environmental policies with respect to those of other countries:

- The first one refers to transboundary pollutants and the possibility of a country setting its environmental policy based only on the damage caused domestically and ignoring the damage to other countries, thus not maximising collective welfare (cf. also Hoel, 1999). This case was discussed above (cf. sub-section 5.1).
- A second reason is that governments might not be concerned with welfare maximisation but rather with the objective of preserving the domestic share of world markets. Such behaviour does not aim at economic efficiency and is thus not dealt with here.
- The third reason, considered in the following, arises when international trade is imperfectly competitive. The question is whether, in a setting of trade liberalisation agreements which prohibit governments from using trade or industrial policies in a strategic way, governments may have incentives to set environmental policies strategically to gain competitive advantages for domestic firms.

With the creation of a Single European Market triggered by the Single European Act of 1986 and related trade liberalisation processes, the question raised under the third point is relevant to European environmental policy making. In this case 'environmental dumping' can be defined as an environmental policy which internalises domestic social costs only incompletely in order to achieve trade policy objectives (Rauscher, 1999).[19] The

[19] In a first-best world, trade policy instruments (e.g. export subsidies, import tariffs) would be used to achieve trade related policy objectives, and environmental policy instruments to achieve environmental objectives. To put this in a more general form, under imperfect competition, optimality would require different instruments for different market failures (Carraro, 1999), for example optimal standards according to local preferences and the nature of the pollutant, and optimal competition policy as well.

likeliness of such strategic behaviour has been studied for different settings of competition and market contexts and leads to the distinction of four cases.

4.2.1 Cases where producers act on competitive markets

In order to investigate the third reason for why governments might act strategically, Ulph (1999) considers a partial equilibrium analysis of a homogeneous good industry, with one producer in the home country, which is selling its product on the world market. Production of the good is assumed to cause pollution which is strictly domestic. The government sets its environmental policy and the firm makes its production decision based on the marginal social cost curve implied by that policy. The relevant question is whether, in this setting, governments might have incentives to set environmental policies too lax, i.e. at a level where marginal abatement costs are below marginal damage costs, in order to protect their domestic industry (Ulph, 1996).

As a first case, Ulph (1999) shows that in a setting of ***perfect competition***, the government would have no incentive to distort its environmental policy, because neither the government nor the firm can affect the world price. If the government set an environmental policy which is weaker than optimal, and the firm would consequently increase its output, the related marginal damage from pollution would exceed the increased marginal private profit from the extra output. This would not be offset by increased sales on the international market as prices would not change.

Other authors have studied strategic government behaviour in a setting where ***producers act competitively but where countries are sufficiently large so that they can influence the terms of trade*** they face, and where countries produce two goods, one which is clean, the other being dirty. This constitutes a second case. The focus is on the behaviour of the government depending on whether the country is an exporter or importer of the goods. This literature has been reviewed by Ulph (1996) and Rauscher (1999). As a result, countries will again set environmental policies at the optimal level when they cannot influence the terms of trade between the clean and the dirty product. However, if they can influence the terms of trade, an exporter of the dirty good will have an incentive to set a policy which is stricter than the optimal level, while an importer will set a laxer than optimal policy. The rationale behind this is that exporters of the pollution intensive good wish to drive up the price of the good for which they have a competitive advantage. This holds when pollution is caused by production.[20]

4.2.2 Cases where markets are imperfectly competitive

When producers act on imperfectly competitive markets, results may be different. The basic difference is that producers then can earn rents, which gives governments (and also producers) incentives to use environmental policy strategically in order to increase their share in the world market (Ulph, 1996). It is however necessary to distinguish two more cases. Firstly, coming back to the first model, where there is one firm per country producing one good, government has no interest in strategic environmental policy either if the firm holds a monopoly position, serving the entire world market with its product. While the ***firm in this third case can affect prices, it is acting as a profit maximiser*** and the role for the government is again restricted to ensuring the firm's internalisation of the external costs of the pollution it causes. A relaxed environmental policy would again lead to marginal pollution damage exceeding the marginal private profit (Ulph, 1999).

Contrary to this, in a fourth case, governments may have an incentive to set environmental policy strategically if competition is imperfect, such as in the case of oligopolistic competition and if firms compete with each other taking as given the output of their rivals. If the government here relaxes its policy from the first best policy, the domestic firm will expand its output because abatement costs are reduced and so are the

[20] If pollution is not related to production but to consumption, net exporter countries will set too lax policies, while importers set too strict policies.

overall costs of production. It will gain additional profits and cause additional damage, which in the simple model just compensate each other.[21] However, as a third effect, the expanded output of the domestic firm will reduce the marginal revenue of the foreign firm, which will consequently reduce its output. This increases the marginal revenue of the domestic firm and increases its profits (Ulph, 1999).

If all governments acted in the same way, 'ecological dumping' might be a likely outcome. However, if they did, total output would increase, which would in turn reduce total industry profits and increase pollution in every country, with little changes in market share (Ulph, 1996; Ulph, 1999). This has two implications. Firstly, the governments were better off if they entered into co-operation to prohibit 'ecological dumping'. Secondly, the optimal co-operative policy would be to set stricter than first best environmental policies in order to reduce output. Finally, when giving up the assumption of only a single firm producing the specific good in each country, a weak environmental policy would result in the domestically competing firms producing too much output relative to the foreign firms. This would imply that government had a reason to set environmental policies stricter than optimal. Governments, therefore, would have conflicting objectives (Ulph, 1999; Ulph, 1996).

4.2.3 A short summary on strategic environmental policy making

Summing up, even though governments may have incentives to distort environmental policies, it is not a priori clear that they would relax policies below the optimal level. There is thus no general case for environmental dumping. Furthermore, even if there were, policy implications are not obvious. Uniform harmonization of policies across countries is not necessarily the best response, as optimality requires marginal damages to equal marginal abatement costs, and both can vary across countries. Nevertheless, if national governments have incentives to distort environmental policies strategically, Ulph (1996) points out that there is a case for having some centralised authority overseeing environmental policy, such as the Commission in the EU context. Given that it will be difficult for such an institution to obtain sufficient information to provide optimal differentiated standards for all countries concerned, this author suggests a different solution. The centralised authority (e.g. the European Commission) might allow countries to set environmental standards within a range of minimum and maximum standards, where countries could be given the possibility to choose a standard outside this range only if they could prove that damage costs can justify this. However, he also suggests that careful empirical analysis would be needed to test whether welfare losses that might arise from strategic environmental policy making are likely to be as large as those that may arise from a supranational authority imposing limits on national governments' environmental policies in the light of the authority's imperfect information about true damage costs in different countries.

It should be noted that empirical evidence shows that there is not much evidence for a close relationship between trade and environmental regulation (Rauscher, 1999). This author argues that the main reason for insignificant results in many studies is that environmental policies in most sectors imply costs that are too small as compared to production costs (energy, labour) and other policies. He reports an average of costs of environmental regulation in total production costs of about 2%. Ulph (1996) [22] argues with comparable results. For the 24 most pollution-intensive industries in the US a range of the percentage of costs entailed by environmental regulations in total costs of 1.9% to 2.9% was found. Nordström and Vaughan (1999), finally, report that pollution abatement costs in developed countries are no more than 1% of production cost for the average industry, rising to perhaps 5% for the worst polluters, and question whether a regulatory

[21] The case would be different if instead firms took as given the prices of their rivals (products are imperfect substitutes). Here non-co-operatively acting governments would have an incentive to set policies stricter than first best.

[22] Citing Tobey (1990).

cost-disadvantage of a few percentage points could turn competitive advantage round. Therefore, environmental policy differences in the past did not have a significant impact on competitiveness, which is why environmental dumping is not so likely nor observed in practice.

4.3 When is European environmental policy making justified from an economic point of view?

Summing up the discussion in sections 4.1 and 4.2, the following recommendations for when European policy making with respect to emissions from municipal waste incinerators is justified or not can be drawn.

From the point of view of the literature dealing with strategic environmental policy making ***EU intervention is not necessary***, and hence neither justified, where competition between incinerators or between waste producers having treated their waste by these plants is perfect and countries are too small to affect the terms of trade. The same conclusion was established in a situation where firms hold monopolistic positions. The only exception to these findings is when the pollutants concerned cause spill over effects from one country to another. This issue was also discussed by the environmental federalism literature, which has established that the spatial extension of the externalities should coincide with the spatial extension of the jurisdiction. Pollutants from municipal waste incinerators with only local spatial effects would therefore be better regulated at a lower policy level. In the European case this would mean at a domestic or even regional level.[23]

In the same line of argument ***EU wide policies are adequate*** where incinerators emit pollutants that spill over from one to other EU countries in order to correctly internalise external costs across countries.[24] According to the strategic environmental policy making literature EU intervention can furthermore be justified in order to avoid the strategic setting of national environmental policies in a context of free trade but imperfect competition. Because here national governments might be led to set policy standards that diverge from those environmental policy objectives that would be optimal in order to gain competitive advantages in international trade. In such a case, EU intervention is, from an economic point of view, justified even when local pollution is at stake.

4.4 A political-juridical criterion - The policy objective of a 'level playing field'

The preceding discussion established that the need for some sort of central policy making and with this the justification of EU environmental policies was largely limited to those cases where a) trans-national externalities exist and/or b) national governments have incentives to distort environmental policies strategically.

In this context it is worth mentioning that article 100 (new numbering article 94) of the Treaty has provided an alternative legal basis for policy making at the EU level, next to article 130r (new numbering 174). This article legalises community action in order to further 'the establishment and functioning of the common market', thus aiming at a harmonisation of legal bases in the Member States. It could thus constitute an alternative rational for EU regulation also towards local pollution. The underlying aim of harmonisation as furthered by article 100 is to ensure that competition in the common market is not distorted, a principle also outlined in article 1g (formerly article 3) of the Treaty. The idea here would be to aim at one internal market. Distortions of the 'level

[23] However, this latter recommendation is unambiguous only as long as one cannot expect interjurisdictional competition, in which case local level policy-making could result in suboptimal standards.

[24] For simplification of the argument, it is neglect here that such pollutants may spill over also to third countries, which would require co-ordination of policies also with other countries, which in fact takes place, as shown by the example of the Long-Range Transboundary Air Pollution (LRTAP) protocol. Similarly it is neglected that in certain cases not necessarily all EU countries are affected by spillover pollutants from other Member States. To justify EU policy making one would assume that at least most EU countries are affected by the transboundary pollution issue.

playing field', as Woerdman (2001: 6 and 10) puts it, refer to an inequity which is primarily defined or perceived in terms of an inequality or asymmetry, and aims at treating firms on an equal footing.

Translated into the environmental policy sphere, the rationale behind this political-juridical objective is to level out the regulatory costs of the regulated economic competitors in the different Member States. The fear is that unequal regulation across the European Union may entail cost differences that lead to a distortion of competition between Member States. To avoid this, the idea is to create a level playing field by limiting poor environmental practice as a competitive factor between countries. In a strict sense, this rationale does *a priori* not apply to all environmental policies directed at local pollution but rather to those environmental policies which could have a major effect on producers' costs and thus on competition. Furthermore, as Glachant (2001:14) argues, while article 100 can serve as a basis for EU legislation aimed at local pollution control where the regulated agents compete between countries, it is not sure that article 100 could be applied where international competition is absent.

But is harmonisation in this case efficient? From the point of view of international trade the rationale is that larger markets without distortions may lead to economies of scale or a more efficient allocation of production factors (Garbe, 1996: 5). But while this may be used as an argument for preventing the strategic use of environmental policy, the issue discussed in the previous section, it does not provide any rationale, from an economic point of view, to internationally harmonise environmental standards. In this context, Rauscher (1999: 396-398) states that differences in environmental endowments, covering the degree of physical scarcity of environmental resources and the preferences of the population for environmental quality, should be reflected in environmental regulation. These differences in endowments, according to the author, constitute the basis of mutual gains from trade and should not be artificially eliminated (see Ulph, 1996: 240 for a similar argumentation). The fact that local conditions may differ significantly between Member States requires locally adapted solutions. In the same line of argument, harmonisation is necessary only in cases of global pollutants, where the impact of pollution is independent of the location of the source.

Therefore, what may seem efficient in order to guarantee fair competition may not be efficient from the perspective of environmental policy, at least not where pollution is an entirely domestic concern. The objective of providing a 'level playing field' is not an economic but a political approach to an equalisation of regulatory costs and hence not an economic efficiency principle.

Finally, as far as the 1989 municipal waste incineration Directive is concerned, levelling out the costs of waste disposal across the European Member States was neither the objective nor the effect of the Directive. As will become obvious throughout the following two chapters, the adoption of the MWI Directive setting emission limit values which were similarly applicable across all Member States did not result in the national regulations becoming homogeneous, but rather entailed non-uniform results of regulation on a higher level. National regulations were and remained heterogeneous both before and after this Directive's implementation (cf. as an example the emission limit values applicable in Germany and France prior to, and after, the adoption of the Directive in annex 4.C). This was made possible by the fact that this European policy defined what is called 'minimum standards'.

5 Is the regulation of municipal waste incinerators at the European level an efficient policy decision?

In the light of the discussions above, this section investigates whether there are reasons that can justify, from an economic point of view, that Europe created a common policy regulating air emissions from municipal waste incineration plants. As will be shown below, neither did market distortions between countries nor transboundary movements of the pollutants regulated provide sufficient justification for a centralised European policy in this area. Regulating airborne emissions from municipal waste incinerators at a European level was therefore probably not an optimal policy decision from the point of view of the efficient allocation of tasks between the European and the national levels.

5.1 Market distortions between countries cannot justify the European policy because waste incineration markets are local

A question arising from the economic literature on strategic environmental policy making is whether differing costs entailed by heterogeneous domestic MWI regulations may have caused distortions to European competition and whether governments may wilfully create such distortions to enhance the market situation of local or domestic firms. The major precondition for such distortions is obviously that the respective market is international. Below it is argued that this condition does not apply and that waste incineration markets are, in general, local. As far as waste incineration is concerned, two complementary aspects determine the size of the geographic market: regulative barriers to waste shipments and transport costs for waste for disposal. Both aspects are investigated in the following.

5.1.1 Regulations and institutional constraints prohibit waste transports

The first aspect which suggests that incineration markets are in general local are regulations or institutional constraints prohibiting waste transport. A number of European policies set the framework for, and limit, waste shipments. There is, firstly, the European Waste Framework Directive (75/442/EEC), modified by Directive 91/156/EEC, which formulated the principles of proximity and auto-sufficiency with respect to waste elimination. While the principle of proximity asks for a limitation of distance and volume of waste transport, the principle of auto-sufficiency requires Member States to establish an integrated and adequate network of waste disposal facilities, to create self-sufficiency at a Community level, and, if possible, at a Member State level as well. This network must enable the waste to be disposed off in one of the nearest appropriate installations. The Directive also demanded the Member States to establish waste management plans, considered as essential to implement this policy, and authorised them to prevent movements of waste which are not in accordance with these plans. As far as the requirement for waste management plans is concerned, countries had, amongst other things, to outline the waste volumes expected and the treatment and disposal facilities foreseen.

Secondly, Council Regulation (EEC) No 259/93 of 1st February 1993 on the supervision and control of shipments of waste within, into and out of the European Community[25], makes shipments of waste subject to national supervision and control, and to prior notification to the competent authorities. On the basis of the principles of proximity and self-sufficiency the Directive invites Member States to take measures in accordance with the Treaty to prohibit generally or partially, or to object systematically to, shipments of waste for disposal (exceptions apply to hazardous waste). Waste shipments for disposal between Member States require authorisation. In this context it is interesting to note that the EU Waste Framework Directive of 1975 declared waste incineration to belong to the waste disposal operations. Recently, on 13 February 2003, the European Court of Justice has ruled in that respect and declared that waste incineration is a disposal operation regardless of whether energy is recovered.[26]

That waste incineration markets can be considered as local was recently confirmed by the European Environmental Bureau (EEB, 2003) when commenting on the draft European Parliament follow-up report on Council Directive 75/442/EEC (Waste Framework Directive). In fact, the EEB argues that with the framework Directive proclaiming the proximity principle as well as the self-sufficiency principle for waste disposal both at the

[25] OJEC No L 30/1- L30/28 of 6.2.1993.

[26] ESA Briefings (September 2003), http://www.esauk.org/information/briefing/luxembourgopinion.asp, 3.12.2003.

level of the Community and of the Member States individually, European waste policy was not seeking a level playing field and an internal market for waste disposal options.

What can be concluded at this stage is that if countries comply with the principles established by the above stated European policies, incineration markets for municipal waste are in general local. Even if the application of such policy principles is legally less binding than would be regulations completely prohibiting waste transports, the principles constitute attempts at the European level to restrict waste transports as far as possible, and with this to limit waste incineration markets to a locally restricted area. If the 1989 municipal waste Directives had had the aim to limit market distortions or to prohibit strategic environmental policy making between European countries, this would hence have only been justified if there was reason to assume that the above outlined European policy was not effective, i.e. that the Member States did not comply with the principles and that they systematically shipped large amounts of municipal waste for incineration between countries. The following empirical examples suggest however that the EU countries actually make attempts to comply with this policy.

Example 1: waste planning in France

In France, the transposition of the EU waste Directive, law n° 75/633 of 15 July 1975, demanded the establishment of waste management plans at the level of 'départements' or groupings of 'départements' for municipal and assimilated waste. Its amendment, law n° 92-646 of 13 July 1992, made these plans obligatory.[27] Such plans limit the reception of municipal waste, as far as residual waste and not sorted waste is concerned, to the area to which the plan applies.[28] Exceptions are possible for municipalities in the close vicinity of the department's borders. Sometimes also green waste may be treated in areas outside, but close to, that covered by the plan. Waste which was previously sorted, waste consisting of secondary materials and toxic or special waste may be transported out of the area. This implies that municipal waste for incineration has to be treated within the area covered by the plan, generally a 'département', sometimes including zones in this area's vicinity. Exceptions are possible, for example in the case of a break-down of operations of a plant, but require the consent of the prefect. Markets for household waste, in France, are therefore local.

These rules do not hold for non-hazardous industrial waste, which is generally treated together with household waste. In France, for treatment of such waste other than its deposition on landfills, a free exchange between areas is possible. The waste management plans stress, however, that as far as such waste is treated in a way identical to municipal waste and at equivalent costs, the proximity principle is to be applied.

Example 2: Countries block exports of municipal waste for incineration

There is some evidence that European Member States do actually block exports of municipal waste for incineration to other Member States. Two of these cases were recently investigated, and justified by, the European Court of Justice, constituting the basis of the above mentioned court ruling. In one case Luxembourg blocked exports of municipal waste for incineration to France, justifying its decision with the proximity principle. In a second case Germany attempted to prevent the export of waste for incineration in Belgian cement kilns[29].

It should not be hidden though that, previously, there were also cases of European countries actually carrying out cross-border waste shipments that were problematic. As

[27] Precisions on the modalities and procedures to establish and revise these plans are formulated in the decree of 18 November 1996.

[28] As examples see the municipal waste management plans of the region Hérault (http://www.herault.pref.gouv.fr/34/grandsdossiers/dechets/synthese2_2.shtm or of the region Puy-de-Dôme (http://www.auvergne.pref.gouv.fr/environnement/dechets); (19/12/2003).

[29] ESA Briefings (September 2003), http://www.esauk.org/information/briefing/luxembourgopinion.asp, 3.12.2003.

one example, a German municipality situated close to the French border and whose incineration capacities were exhausted shipped its municipal waste to the Strasbourg area to have it incinerated there instead of investing immediately in own higher incineration capacities. The reason why this co-operation was particularly problematic is that the Strasbourg incinerator operated at considerably lower abatement standards than were required in Germany and, what is more, which were not in compliance with the EU Directive's requirements either. In general, such problematic waste transfers seem to have been limited to border regions, and should therefore only present a marginal problem. Such cases would rather call for some kind of bilateral co-ordination between the respective countries involved, and do not directly justify EU level policy making.

5.1.2 Transport costs limit the distance over which waste shipments for incineration are economically viable

Even if the national and international regulations limiting shipments of waste for disposal were not implemented, there is a second aspect that implies that markets for waste incineration are local: the level of transport costs which determines the distance to waste incinerators over which it is economically sensible to transport the municipal waste. Such shipments would be economically rational if incinerators outside the area the municipal waste stems from operated at lower costs than the local waste disposal facilities, if they consequently demanded lower prices for waste treatment, and therefore made cost savings possible. What is decisive as to whether shipments of municipal waste for incineration over long distances, and hence to foreign countries, are an economic undertaking and with that a possible reason for international market distortions, is thus the level of transport costs relative to incineration cost differentials between regions and countries.[30]

As far as the first element is concerned SOFRES Conseil (1998a and 1998b) estimated the average costs for transporting municipal waste of a standard density of 0.4 t/m^3 by road at approximately 0.15 €/(t*km) for France.[31] The same study estimated that a direct transporting of waste by the collection vehicles of the municipal waste collection service were economical only up to a maximum distance of 40 to 50 km between the point of collection and the treatment plant.

In order to determine whether incineration cost differences between European countries were large enough to set incentives for countries to attract foreign waste to incinerators operating at lower costs and possibly lower environmental standards, and whether this might hence justify policy making at the European level, one would ideally compare incineration cost differentials across Europe before the adoption of the 1989 Directives.

Comparable data on waste incineration costs across European countries are not easily available. No data at all could be found referring to treatment costs before the implementation of the 1989 EU Directive for those countries that had no advanced domestic regulation concerning airborne emissions from MWI plant. Nevertheless, a number of studies compare incineration costs for different capacity of plant and for more recent abatement requirements in France (cf. SOFRES Conseil, 1998a and 1998b) and across European countries (for example European Commission, 1997; Eunomia research & consulting, 2002). For illustration, the incineration cost data provided by these studies together with the underlying assumptions -concerning for example the amount of waste incinerated, the plant capacities, the regulatory requirements met by the plants and the

[30] Municipal waste incinerators represent regulated geographic monopolies. Unlike non-regulated monopolies, they cannot set prices to earn monopoly rents. Instead, they are obliged to set prices so as to balance out payments and revenues of municipal waste management. Regulated monopolies are assumed to set prices equal to average costs. It is hence in order to focus on cost data rather than on gate-way prices.

[31] This calculation is based on the distance between a transit centre and the treatment facility and takes into account the costs for the return of the empty vehicles. This means that this distance between a transit centre and the treatment facility is 1 km (while the distance to which the cost estimate applies is 2 km, taking into account the return journey of 1 km).

cost items covered- are presented in annex 4.D. With the international studies either presenting pre-tax costs net of revenues or focussing only on the costs of the abatement equipment, it is difficult to assess municipal waste incineration costs across countries in a comparable way. We therefore focus on available data from French studies.

For France, SOFRES Conseil (1998a and 1998b) estimates the range of net total costs for new plant complying with the French transposition of the Directive -depending on the technology to abate acid pollution- to vary between: 75 and 85 €/t for plants of a capacity of 5 t/h; 69 and 79 €/t for plants of a capacity of 10 t/h; and 59 and 69 €/t for plants of a capacity of 20 t/h. For existing plant the same study estimates the range of net total costs to vary between 60 and 77 €/t for plants of a capacity of 10 t/h.

Taking the maximum cost differential amongst these data for France, i.e. 26 €/t and applying the transport cost estimate per tkm given above, transporting waste in France would be economical up to a distance of approximately 170 km. In practice the maximum economic distance may differ from this value -and may have differed from it before the Directive's implementation- given that France runs both higher and lower capacity incinerators than those for which data were estimated by SOFRES Conseil, which implies larger treatment cost differentials. Furthermore, today, a number of French plants comply with stricter emission limit values than those defined by the 1989 Directive.[32]

Nevertheless, Manciet (1997), in a study for the French Ministry for Economy, Financial Affairs and Industry, comes to a comparable result. This author modelled what would be the optimal waste management in a hexagonal waste collection zone. The primary aim of the exercise was to determine the optimal size of waste incinerators, given the trade-off between the economies of scale and the necessity of limiting collection transport costs, in order to minimise waste management (collection and treatment) costs. It also produced results on the optimal distance between two incinerators. The model was set both in a context of perfect competition and in the more realistic context of a strong concentration of few enterprises holding the major share of the market, which is characterised by numerous entrance barriers. Assumptions were made about parameter values such as population density, waste quantities per inhabitant, fixed costs and transport costs.[33] As a general result, the optimal distance between two incinerators is a decreasing function of the population density, and the optimal waste quantity to treat is a decreasing function of the transport costs. For tow zones with a uniform population density of 100 inhabitants/km^2, in the setting of perfect competition, the optimal distance between two incinerators is evaluated at 130 kilometres. For two zones with differing population density of 80 inhabitants/km^2 and 40 inhabitants/km^2 respectively, the optimal distance is larger: 170 kilometres. No values are given for optimal distances for the case of imperfect competition, but the author notes that particularly in rural areas a larger perimeter of collection would be needed to achieve waste quantities that allow to take full advantage of the economies of scale.

Assuming that municipalities have an interest in limiting the costs of waste management, the found distances over which transport of municipal solid waste is an economic undertaking suggest that municipal waste incinerations markets are local, and that transboundary shipments of such waste for disposal, if any, should be limited to border regions. To which extent waste incineration cost differentials have been, and are, larger between different European countries than within France is difficult to assess. If the aim were to assess cases where trans-European transport of waste for incineration would be

[32] A circular of February 1997 required new plants to meet dioxin emission limits and some plants already take account of the new incineration directive's NO_X emission limits.

[33] The following parameter values are taken into consideration: Incineration costs -made up of investment costs and fix operation costs- are estimated to amount to between 9150 and 12200 €; transport costs are estimated at 0.3€ per tonne kilometre (based on ADEME, 1996); the waste amount per capita is estimated at approximately 450 kg (estimation for 2002); and the maximum quantity of waste input for an incinerator with energy recovery is estimated at 30000 tonnes.

economically viable, further detailed information would be needed, for example on the exact treatment costs of each incinerator, on the exact location of these plants, and on the availability of incineration capacities to treat imported waste.

5.2 Market distortions between waste producers cannot justify the Directive because the impact of waste incineration costs on production costs is too low and industrial waste producers have alternative waste treatment options

In principle, nationally differing environmental regulations of the municipal waste incinerators might also have impacts on the product markets of producers of assimilated waste. The non-hazardous portion of the industrial waste which is either collected together with household waste or separately collected, but often treated in the same installations as the municipal waste, stems from activities such as commerce and markets, craft, the service sector, public services, hospitals, schools, and industrial enterprises. The necessary precondition for market distortions here are, firstly, that these waste producers operate internationally[34], secondly, that there are differences in waste treatment costs across countries and, thirdly, that the share of waste treatment in overall costs is important enough to change competition conditions of the producers. The underlying questions would be whether countries may have set weak environmental standards for municipal waste incinerators in order to improve the competitive position of domestic waste producers, and whether cost differences (to the extent that pollutants are local) do not just reflect different environmental endowments and preferences, in which case they would be economically optimal.

Although these questions can only be approached in a very general way, owing again to a lack of sufficiently comprehensive and detailed data on the situation at the end of the 1980s and across European countries[35], some general elements can be provided which support the thesis that the impact of waste incineration costs on production costs of waste producers is too low to have an impact on the product market.

Firstly, one can evoke the general empirical evidence mentioned earlier that costs entailed by environmental regulation tend to be too small in overall costs to lead to distortions in international competition. Secondly, while municipalities have the responsibility for treating household waste, non-hazardous industrial waste is in the waste producer's responsibility. This implies that producers of non-hazardous/commercial waste are not obliged to have their waste treated in the installations provided by the municipalities: they are free to, either, treat the waste themselves in appropriate installations, have it treated by private operators, or hand it over to the municipality's waste collection. In this sense they have the opportunity to choose the least cost option and are not necessarily directly affected by environmental regulations of municipal waste incinerators. It is hence not obvious why national governments would set weak environmental standards for waste incinerators in order to improve the competitive situation of domestic industry which, as a side product, produces waste.

Finally, it is worth recalling that environmental standards required from MWI plants, and with this the costs of incineration, continue to differ across Europe. In this context, searches on the internet-site of the European Competition Authority do not reveal any cases brought up to this authority because of suspected distortions in competition caused by differing environmental standards for municipal waste incineration plants.

[34] It is obvious that some of these waste producers do not compete on international product markets (e.g. local commerce and markets, supermarkets, hospitals). Others however do, such as certain services and industrial enterprises.

[35] In particular, it is not possible to provide comprehensive data, applying to the situation before the adoption of the Directive, and that could assess these producers' production costs for all relevant products across countries, in order to evaluate the possible impact of differing incineration costs on the waste producers' competitive situation on international product markets.

5.3 Transboundary spill-over effects from pollutants do not give a general justification for the European policy because most pollutants regulated are local

A second line of argumentation that may or not justify the regulation of atmospheric emissions from municipal waste incinerators on a European level is given by the literature of environmental federalism. This literature establishes that environmental policy making should be allocated to the level of government which corresponds, in a spatial view, with the frontier of the externality caused by the pollutant to be regulated. In this line of economic argumentation, European policy making is primarily justified in the case of trans-frontier pollution that affects most of its members. And indeed, the Directive in its introductory section relates to this criterion by basing its policy on the fact that municipal waste incineration gives rise to emissions of substances, causing air pollution which in some cases may have transboundary features. But in how far is this actually the case for the majority of pollutants regulated by the Directive?

The exact distance over which pollutants travel before settling out depends on local conditions, such as geographic conditions around the location of an emitting plant, but also on factors determined by the construction of a plant (e.g. stack height). Nevertheless, pollutants can generally be distinguished into those that drift widely across different countries, and those that have rather local effects. Considering the pollutants regulated by the 1989 MWI Directive from the perspective of the knowledge about the spatial dispersion of pollutants at that time, the majority of pollutants addressed have rather limited spatial effects (cf. Table 4.3).

The spatial effects of the pollutants regulated

SO_2 clearly represents a pollutant that is known for its long distance effects. In fact, this pollutant was at the origin of the Convention on Long-Range Transboundary Air Pollution (LRTAP). A further pollutant which, while having primarily local effects, may also travel over some distances, is dust. Dust particles in fact settle out under their own weight but may remain suspended for some time. The spatial effect here is related to the particle size.

Table 4. 3: Spatial scale of pollutants regulated by Directive 89/429/EEC as assessed in the early 1990s

Pollutants regulated	Emission limit values	Indirectly regulated	Spatial scale of pollutant
Dust	√		Local to medium distance
Heavy metals	√		Local
HCL	√		Local
HF	√		Local
SO_2	√		Local to long distance
CO	√		Local
Organic compounds	√		Local
Dioxins/furans		√	Local

On the other side of the scale are HCL and HF, both pollutants that were considered by the European Commission (1997) as being readily soluble and hence tending to be washed out of the atmosphere within a relatively short distance from the pollution source. In fact, the range of effect was considered to be significantly more restricted than for SO_2 which remains in the atmosphere longer. According to the same source, also the impacts of heavy metals and dioxins are considered most likely in a restricted region around a given plant. For these pollutants exposures arise from direct inhalation, but also indirectly through ingestion of contaminated food, water and soil. This means that, abstracting from a possible export of contaminated food, local conditions and behavioural factors influence total exposure much more than in the case of SO_2 (European Commission, 1997). Organic compounds also have primarily local effects.

According to this assessment, for the majority of pollutants addressed by the Directive (heavy metals, CO, HCL, HF, dioxins, and to a large extent also dust), the subsidiarity principle, made operational through assessment criteria drawn from the environmental federalism literature, did not justify the regulation on a European level. The only pollutant for which international policy was clearly justified by this literature is SO_2.[36]

Do technological inter-relationships in the abatement of spill-over and local pollutants justify a joint regulation?

Given that with SO_2 one transboundary pollutant was addressed by the Directive which can justify European level policy making, one might wonder whether reasons of technological jointness of abatement justify the regulation of a number of local pollutants under the same policy. If so, economies of scope might justify the joint regulation from an efficiency perspective -as long as no national regulation addresses the same pollutants- in order to reduce regulation costs.

To give an example, heavy metals are generally abated by help of de-dusting facilities, electrostatic precipitators (ESP) or filters. Both groups of pollutants were however considered as primarily local. Therefore, while their joint regulation seems sensible, this did not justify European level regulation. But heavy metals, for example lead, mercury and cadmium as well as organic compounds can also be removed by acid gas removal systems, principally aimed at neutralising acid compounds such as HCL, HF and the transboundary pollutant SO_2 (Milhau and Pernin, 1994, European Commission, 1997). However, in order to reduce heavy metal emissions, these technologies would normally be utilised in conjunction with ESPs or bag filters. All in all technological interrelationships do not appear a sufficient reason to explain the choice on the scope of pollutants regulated by the European Directives in 1989 either.

There is no evidence for a race-to-the-bottom in the case of European municipal waste incineration regulation

Finally, it is worth mentioning that the concern over a race-to-the bottom is not relevant in the case of municipal waste incineration. Firstly, the establishment of such installations is the result of public waste planning in relation to waste disposal capacities needed. Secondly, there is hence no reason why specific regions or countries would attempt to attract all waste incinerators to their jurisdiction.[37] And indeed, both the situation before and after the adoption of the 1989 Directives give no evidence of a-race-to-the-bottom in the domestic regulation of air emissions from municipal waste incineration plants. Some countries, such as Austria, Sweden, the Netherlands and Germany, driven by local concerns over pollution, consistently developed stricter regulations, while some countries, such as France and the UK followed at a lower pace. The same pattern continues to this day.

[36] It should be noted that more recently the assessment of the spatial effects of a number of pollutants has changed. In 1998 the Protocol on Heavy Metals (Aarhus Protocol on Heavy Metals) was adopted under the LRTAP Convention, covering three of the eight heavy metals regulated by the 1989 Directives: cadmium, lead and mercury. Also particulate matter has more recently been taken up in the list of pollutants for which the problems they entail are increasingly perceived to require international action (Reuther, 2000). This author explains the policy shift by better monitoring and an increased understanding of air pollution transport. Finally, the CITEPA (the French Centre Interprofessionnel Technique d'Etudes de la Pollution Atmosphérique) lists HCL and HF as belonging to the group of pollutants that can travel over long distances (See http://www.citepa.org/pollution/phenomenes.htm, 24/11/2003).

[37] Note however that there may be another rational in keeping cost and public spending low and hence avoiding costly abatement investment which is related to the fiscal policy concerns of local public bodies. This issue is discussed in chapter 7 with respect to the French compliance path.

6 Conclusions

The primary aim of this chapter was to assess the MWI Directive's efficiency from the point of view of the subsidiarity principle made operational using economic criteria. The guiding question was thus whether the policy decision was made at the appropriate level and hence resulted in an efficient allocation of the decision-making tasks between the European and national levels.

It has become obvious that an evaluation of policies according to economic efficiency criteria becomes rather difficult in the presence of many externalities or market failures or transboundary pollution. Limitations arise primarily from a lack in information on detailed costs, externalities and local preferences. While therefore not being so much operational in practice, the economic criteria suggested are nevertheless useful to present and discuss the issue in theory and they do allow for a qualitative assessment of the Directive's efficiency.

It was shown that from an economic point of view there is only limited justification for the establishment of the 1989 EU Directives on atmospheric emissions from municipal waste incineration plants. Firstly, as far as the pollutants regulated are concerned, from a historic perspective, the majority of pollutants were considered as having only limited spatial effects. Spill-over effects therefore could not justify the central level regulation. Furthermore, there is no evidence of a 'race-to-the-bottom', by which Member States would have kept the environmental standards low in order to reduce costs for local incinerators and business and thus create a destructive competition. The strictness of national regulations, and with this the local costs of waste incineration, remained heterogeneous also after the implementation of the Directive, with some countries applying stricter emission limit values than required by the European policy. Finally, there are also no market distortions between the countries because both other international and national regulations as well as transport costs imply that waste incineration markets are local.

The finding that the MWI Directive was probably not optimal with respect to the allocation of tasks between the European and the national level opens the possibility that national implementation processes and even an implementation gap may have enhanced the policy's efficiency. Before assessing this question empirically, the following chapter studies the efficiency of the Directive under a second focus: that of the efficiency of the Directive's contents.

Annex 4.A: The Directives' requirements in detail

Emission limit values for new plant

The Directives distinguish between nominal capacity categories of plant, where capacity refers to the sum of incineration capacities of furnaces of which the plant is composed. As far as new plants are concerned, the following requirements were to be met by each Member State (Table 4.A.1):

Table 4.A. 1: EU emission limits in mg/m3 for new incinerators

	To be met by plants authorised from 1 December 1990 onwards		
	< 1 t/h	1 t/h- < 3 t/h	≥ 3 t/h
Dust	200	100	30
Pb+Cr+Cu+Mn	-	5	5
Ni+As	-	1	1
Cd+Hg	-	0.2	0.2
HCl	250	100	50
HF	-	4	2
SO_2	-	300	300
CO	100	100	100
Organic Compounds	20	20	20
Residence time of combustion gases	The gas resulting from the combustion of the waste is raised, after the last injection of combustion air, in a controlled and homogeneous fashion and even under the most unfavourable conditions to a temperature of at least 850 °C at least two seconds in the presence of at least 6% oxygen		

Emission limits in mg/m3 for existing incinerators. Standard conditions: 273 degrees K, 101,3 kPa, 11% Oxygen or 9% CO_2 and dry gas. - Source: Art. 3 and 4, 89/369/EEC

Monitoring requirements for new and existing plant

Table 4.A. 2: Monitoring requirements for new plants

Pollutant	< 1 t/h	≥1 t/h
Total dust	Periodically	Continuously
CO	Periodically	Continuously
Oxygen	Periodically	Continuously
HCL	Periodically	Continuously
Heavy metals		Periodically
HF		Periodically
SO2		Periodically
organic compounds	periodically	Periodically
temperature of gases resulting from the combustion of waste (and water vapour content)	continuously	Continuously
residence time of combustion gases	verifications at least once when the plant is first brought into service and under the most unfavourable operating conditions envisaged	verifications at least once when the plant is first brought into service and under the most unfavourable operating conditions envisaged

Standard conditions: 273 degrees K, 101,3 kPa, 11% Oxygen or 9% CO_2, dry gas (17% oxygen for plants < 1 t/h). - Source: Article 6, 89/369/EEC

As presented in the above table, not only emission limit values, but also monitoring requirements are stricter for larger size plants. As far as the measurement results considered as compliance for new plant are concerned (Table 4.A.3), requirements are largely identical for all plant capacity categories. The only exception concerns CO.

Table 4.A. 3: Monitoring results indicating compliance of new plant

Pollutant	Capacity < 1 t/h	capacity ≥ 1 t/h
Temperatures and oxygen content	Minimum values to be observed at all times when the plant is in operation	
CO	Limit value for the hourly average	limit value for the hourly average; 90% of all measurements taken in any 24-hour period must be below 150 mg/nm3
Other substances to be continuously measured	none of the moving seven-day averages measured may exceed the corresponding limit values; none of the daily averages measured may exceed the limit value by more than 30%	
substances to be periodically measured	The results of each of the series of measurements defined and determined according to the rules laid down by the competent authorities under articles 6 (3), (4) and (5) must not exceed the emission limit values	

Source: Article 5

Monitoring requirements as established for 'existing' plant are presented in Table 4.A.4. Final monitoring requirements for existing plants are identical to those for new plants. But again, weaker interim requirements were specified for plants of a capacity below 6 t/h.

Table 4.A. 4: Monitoring requirements specified for existing plant

Pollutant	By 1 December 1995		By 1 December 2000		By 1 December 1996
	< 1 t/h	≥1 t/h and < 6 t/h	< 1 t/h	≥1 t/h and < 6 t/h	≥ 6 t/h
Total dust	periodically	continuously	periodically	continuously	continuously
CO	periodically	continuously	periodically	continuously	continuously
Oxygen	periodically	continuously	periodically	continuously	continuously
HCL			periodically	continuously	continuously
Heavy metals				periodically	periodically
HF				periodically	periodically
SO2				periodically	periodically
organic compounds			periodically	periodically	periodically
temperature of gases resulting from the combustion of waste (and water vapour content)	continuously	continuously	continuously	continuously	continuously
reference time of combustion gases	verifications at least once after any adaptation of the plant and, in any event, before 1 December 1995		verifications at least once when the plant is first brought into service and under the most unfavourable operating conditions envisaged		

Standard conditions: 273 degrees K, 101,3 kPa, 11% Oxygen or 9% CO_2, dry gas (17% oxygen for plants < 1 t/h). - Source: Articles 6 of Directives 89/429/EEC and 89/369/EEC

Table 4.A.5 indicates the measurement results considered as compliance for 'existing' plant. Again, the Directive specifies that the same requirements as those defined for new municipal waste incinerators (MWIs) apply to 'existing' plants of a capacity > 6 t/h from 1 December 1996 and to smaller MWIs from 1 December 2000 onwards (art. 2). This implies that, from this date on, all emission values to be measured continuously have to be met on a moving 7-day-average. On a daily average the latter (except for CO) may be exceeded by up to 30%.

Table 4.A. 5: Measurement results indicating compliance of existing plant

	Applicable by 1 December 1995		Applicable by 1 December 2000		Applicable by 1 December 1996
	capacity < 1 t/h	**capacity ≥1 t/h and < 6 t/h**	**capacity < 1 t/h**	**capacity ≥1 t/h and < 6 t/h**	**capacity ≥ 6 t/h**
Temperature and oxygen content	Minimum values to be observed at all times when the plant is in operation				
CO	limit value for the daily average	limit value for the hourly average	limit value for the hourly average	limit value for the hourly average; 90% of all measurements taken in any 24-hour period must be below 150 mg/nm3	
Dust	concentration values measured in accordance with the rules laid down by the competent authorities must not exceed the limit value	none of the moving seven-day averages measured may exceed the limit value; none of the daily averages measured may exceed the limit value by more than 30%	The results of each of the series of measurements defined and determined according to the rules laid down by the competent authorities must not exceed the emission limit values	none of the moving seven-day averages measured may exceed the corresponding limit values; none of the daily averages measured may exceed the limit value by more than 30%	
Other substances to be continuously measured			none of the moving seven-day averages measured may exceed the corresponding limit values; none of the daily averages measured may exceed the limit value by more than 30%		
Substances to be periodically measured			The results of each of the series of measurements defined and determined according to the rules laid down by the competent authorities must not exceed the emission limit values		

Source: Articles 5 89/369/EEC and 89/429/EEC

The Directives furthermore indicate that the measurement results shall be recorded, processed and presented in an appropriate fashion so that the competent authorities can verify compliance with the conditions laid down in accordance with procedures to be decided upon by those authorities (Art. 6 (3)). The sampling and measurement procedures, methods and equipment shall require prior approval of the competent authorities (Art. 6 (4)). For the periodic measurements, the competent authorities shall lay down appropriate measurement programmes to ensure that the results are representative of the normal level of emissions of the substances concerned. The results obtained must be suitable for verifying that the limit values have been observed (Art. 6 (5)). Furthermore, the Directives require the results of controls to be made available to the public (Art. 8, 89/429/EEC and art. 9, 89/369/EEC).

Annex 4.B: Evolution of requirements during the Directives' negotiation process

Stated in the following two tables are only those requirements that changed during the Directive's negotiation process.

Table 4.B. 1: Evolution of Requirements for New Plant

Initial Commission Proposal (88/C75/05)	Economic and Social Committee Opinion (88/C318/02)	European Parliament Opinion (OJ C/1989/69/219)	Finally adopted Directive 89/369/EEC
Definition of new plant: plant authorised from 30 June 1989 onwards			Definition of new plant: plant authorised from 1 December 1990 onwards
Definition of municipal waste incineration plant (article 1,4): means any technical equipment used for the treatment of municipal waste by incineration, with or without the recovery of combustion heat generated, but excluding a) combustion plants conceived to use other fuels and which only burn municipal waste derived fuels additionally b) plants on land and at sea specifically used for the incineration of sewage sludge, chemical, toxic and dangerous waste, medical waste from hospital or other types of special waste, even if these plants may burn municipal waste as well		Suggestion to delete the sentence part: "medical waste from hospital or other types of special waste". Suggestion to add that the definition of MWIs covers all plants, whether they are public or private.	Definition of municipal waste incineration plant (article 1,4): means any technical equipment used for the treatment of municipal waste by incineration, with or without the recovery of combustion heat generated, but excluding plants used specifically for the incineration of sewage sludge, chemical, toxic and dangerous waste, medical waste from hospital or other types of special waste, on land or at sea, even if these plants may burn municipal waste as well
		Article 1, additional paragraph: municipal or other waste must not be incinerated in other facilities than those foreseen for it.	
Differentiation of capacity classes for emission limit values: < 5 t/h and ≥ 5 t/h			Differentiation of capacity classes for emission limit values: < 1 t/h, 1 t/h - < 3 t/h, and ≥ 3 t/h
Emission limit values (first value for plants < 5 t/h, second for plants ≥ 5 t/h) Dust 100/50 Pb+Cr+Cu+Mn 5/5 Ni+As 1/1 Cd 0.1/0.1 Hg 0.1/0.1 HCL 100/50 HF 4/2 SO2 300/300 CO 100/100 Organic compounds 20/20	Consider emission limit values as not strict enough Current state of technology for dust is 30 mg/Nm3 Suggest additional limit values for tin and cobalt Criticise lacking requirement of application of state of the art technology	Emission limit values (first value for plants < 5 t/h, second for plants ≥ 5 t/h) Dust 60/10 Pb+Cr+Cu+Mn+cobalt 1/1 Suggest to add a paragraph stating that in order to reduce atmospheric pollution to a minimum, Member States' policy with respect to waste should give priority to the following objectives: reduction of the production of waste selection and separation of waste with respect to its transformation promotion of recycling and re-use of waste yet a further paragraph should establish that Member States should not build new incinerators in sandy areas, because these are very sensitive to pollution due to the precipitation of emissions, and are already strongly loaded with heavy metals.	Emission limit values (first value < 1 t/h, second 1 t/h - < 3 t/h, third ≥ 3 t/h) Dust 200/100/30 Pb+Cr+Cu+Mn -/5/5 Ni+As -/1/1 Cd+ Hg -/0.2/0.2 HCL 250/100/50 HF -/4/2 SO2 -/300/300 CO 100/100/100 Organic compounds 20/20/20

Initial Commission Proposal (88/C75/05)	**Economic and Social Committee Opinion (88/C318/02)**	**European Parliament Opinion (OJ C/1989/69/219)**	**Finally adopted Directive 89/369/EEC**
Article 10 derogation for plants < 1 t/h subject to seasonal fluctuations, particularly if situated in touristy areas: Dust: 350 mg/Nm3	Derogation criticised and limit value considered as unacceptable		Different derogations for plants < 1 t/h finally adopted: Article 3,2 allows emission limit values to refer to an oxygen content of 17%, in this case, the concentration values may not exceed those laid down in the directive divided by 2.5. Article 3,3 allows to grant authorisation to plants provided that they meet a limit value of 500 mg/Nm3 for dust and that all provisions of directive 84/360/EEC are met.
Require establishment of measurement requirement for dioxins by the Council as soon as the state of the art allows to do so (art. 6,6)	Suggest measurement requirement for dioxins (as already applied in Germany)		Requirements not included in Directive Instead possibility that Member states introduce limit values for dioxins and furans (art. 3.4)
Combustion conditions: 850° C, 6% oxygen (art. 4)	In order to improve combustion conditions suggestion to require adding a post-combustion chamber and fixing the temperature to be achieved	Suggests to add a paragraph to article 4, stating that all incinerators must be equipped with high stacks to reduce the nuisances to the direct environment to a minimum. Furthermore: every new incinerator should be equipped with a post-combustion chamber where the temperature should be raised to at least … degrees.	Combustion conditions: 850° C, 6% oxygen
Monitoring requirements (all capacity) Dust - continuously CO – continuously Oxygen – continuously HCL – continuously Heavy metals - periodically HF – periodically SO2 – periodically Organic compounds - periodically Temperature of combustion gases – continuously		Monitoring requirements (all capacity) HF - continuously SO2 – continuously	Monitoring requirements (< 1 t/h / ≥ 1 t/h) Dust: periodically / continuously CO: periodically / continuously Oxygen: periodically / continuously HCL: periodically / continuously Heavy metals: - / periodically HF: - / periodically SO2: – / periodically Organic compounds: periodically / periodically Temperature of combustion gases: : continuously / continuously
Monitoring results indicating compliance: Temperature and oxygen content: minimum values to be observed at all times CO: limit value for daily average Other substances to be continuously measured: a) limit value for weekly average, b) none of the daily averages exceeds the limit value by more than 30% Substances to be periodically measured: concentration values measured must not exceed limit values	Criticise tolerance left with respect to limit values of continuously measured substances (other than CO, temperature, oxygen) and point to the stricter German regulation, requiring that concentrations on a daily average must not exceed the limit values.	Monitoring results indicating compliance: Other substances to be continuously measured: a) limit value for weekly average, b) none of the daily averages must exceed the limit value	Monitoring results indicating compliance (< 1 t/h / ≥ 1 t/h): Temperature and oxygen content: minimum values to be observed at all times / ibid. CO: limit value for hourly average / a) limit value for hourly average, b) 90% of measurements taken in any 24h period must be below 150 mg/Nm3 Other substances to be continuously measured: - / a) limit value for moving seven-day averages, b) none of the daily averages exceeds the limit value by more than 30% Substances to be periodically measured: results of the series of measurements defined by the competent authorities must not exceed the emission values / ibid.

Initial Commission Proposal (88/C75/05)	**Economic and Social Committee Opinion (88/C318/02)**	**European Parliament Opinion (OJ C/1989/69/219)**	**Finally adopted Directive 89/369/EEC**
New incineration plants have to be equipped with auxiliary burners. When the time these are used exceeds 5% of the operation time of the plant during 5 subsequent days, appropriate measures to re-establish sufficient combustion conditions must be taken (art. 7).		New incineration plants have to be equipped with auxiliary burners. When the time these are used exceeds 5% of the operation time of the plant during 5 subsequent days, appropriate provisions must be taken to re-establish sufficient combustion conditions without modifying the priorities fixed in the newly added paragraph of article 3 (note: concerning reduction, separation, re-cycling; re-use of waste) (art. 7).	Auxiliary burners are demanded but further limitations to their use are dropped (article 7)
When plants exceed limit values, competent authorities must take appropriate measures to re-establish compliance. Plants that don't conform to these measures cannot continue operation. (art. 8,1) In case of breakdown of purification devices, plants may under no circumstances continue to operate more than 16 hours uninterruptedly, and the cumulative duration over a year of operation in such cases shall be less than 200 hours. The dust content of off-gases must, during the periods mentioned in the preceding paragraph, not exceed 600 mg/Nm3 and all other conditions, particularly the combustion conditions, must be complied with (article 8,2).		Suggestion to tighten the dust emission limit that must not be exceeded within periods where the abatement equipment does not function properly from 600 mg/Nm3 to 350 mg/Nm3.	Plants exceeding limit values must not continue to operate any longer while failing to comply and the competent authority shall take appropriate measures to ensure it is modified or no longer operated (art. 8,1) In case of breakdown of purification devices, plants may under no circumstances continue to operate more than 8 hours uninterruptedly, and the cumulative duration over a year of operation in such cases shall be less than 96 hours (art. 8,2). The dust content of the discharges shall under no circumstances exceed 600 mg/Nm3 during the periods referred to in the preceding subparagraph and all the other conditions, in particular the combustion conditions, shall be complied with.
			Article 4,4 requires waste incineration plants to be designed, equipped and operated in such a way to prevent emissions into the air giving rise to significant ground level pollution. Waste gases shall be discharged in a controlled fashion by means of a stack, appropriate stack height shall be ensured by the competent authority.
The public must have access to the results of measurements of emissions and operation conditions (art. 9)			The final article version adds a limitation to the requirement to make control results available to the public by stating that these are subject to the respect of provisions applicable in respect of commercial secrecy. (art. 9)
Member States can impose stricter standards than the directive (art. 13)			Not included in the final version of the Directive, but article 3,4 allows for additional pollutants to be regulated.
Council directive 85/337/CEE of 27 June 1985 on the assessment of the effects of certain public and private projects on the environment should be modified with respect to point 9, annex I. It should be added "and municipal waste incineration plants whose nominal capacity is greater than 5 tonnes of waste per hour". (Article 12).		Scope enlarged. The modification should say: "and municipal waste incineration plants".	Article dropped.

Initial Commission Proposal (88/C75/05)	Economic and Social Committee Opinion (88/C318/02)	European Parliament Opinion (OJ C/1989/69/219)	Finally adopted Directive 89/369/EEC
The directive must be transposed into national law by 30 June 1989			The directive must be transposed into national law by 1 December 1990

Table 4.B. 2: Evolution of Requirements for Existing Plant

Initial Commission Proposal (88/C75/06)	Economic and Social Committee Opinion (88/C318/02)	European Parliament Opinion (OJ C/1989/69/219)	Finally adopted Directive 89/429/EEC
Definition of existing plant: plant authorised before 30 June 1989			Definition of existing plant: plant authorised before 1 December 1990
Differentiation of capacity classes for emission limit values: < 1t/h, ≥ 1 - <6 t/h, ≥ 6 t/h for transitional arrangements, < 5 t/h and ≥ 5 t/h for final requirements			Differentiation of capacity classes for emission limit values: < 1t/h, ≥ 1 - <6 t/h h for transitional arrangements, ≥ 6 t/h for final arrangements, < 1 t/h, ≥ 1 - < 3 t/h, and ≥ 3t/h for final requirements
Transitional requirements to be reached by 30 June 1994, final requirements to be reached by 30 June 1999	Delays fixed for upgrading existing plant criticised. Even though adaptation of small plants may pose technical and economic problems, there is no justification for large plants remaining behind the state of the art technology. In particular, 10 years left for upgrading existing plant are considered as too long.	Demand to reduce the transitional period for meeting intermediate emission limit values from 5 to 3 years, and to reduce the deadline for meeting the same limits as new plants from 10 to 5 years.	Transitional requirements of smaller plant to be met by 1 December 1995 Final requirements for smaller plant to be met by 1 December 2000 Final (unique) requirement for large plants (≥ 6 t/h) to be met by 1 December 1996 (article 2)
Transitional emission limit values (first value for plants < 1t/h, second for ≥ 1 - <6 t/h, third for ≥ 6 t/h) Dust 600/150/100 CO 100/100/100 (final emission limit values cf. new plant above)	The difference between emission limits for existing and new plant suggests reviewing the limit values for existing plant. In particular, the dust emission limit value of 600 mg/Nm3 for plants < 1 t/h cannot be considered a real improvement or abatement measure.	Demands to reduce the transitional dust emission limits values (first value for plants < 1t/h, second for ≥ 1 - <6 t/h, third for ≥ 6 t/h): Dust: 300/60/30	Transitional emission limit values (first value plants < 1t/h, second ≥ 1 - <6 t/h) (final limit values cf. new plant above) Dust 600/100 CO 100/100 Final (unique) equirement for plant ≥ 6 t/h Dust: 30 Pb+Cr+Cu+Mn: 5 Ni+As: 1 Cd+Hg: 0.2 HCL: 50 HF: 2 SO2: 300 CO: 100 Organic compounds: 20

Initial Commission Proposal (88/C75/06)	**Economic and Social Committee Opinion (88/C318/02)**	**European Parliament Opinion (OJ C/1989/69/219)**	**Finally adopted Directive 89/429/EEC**
Article 2 requires the progressive adaptation of existing plant to BAT. Furthermore, adaptation to directive requirements should take place as soon as possible, taking into account the remaining life span of plants.			Requirements dropped
Requires establishment of measurement requirement for dioxins by the Council as soon as the state of the art allows to do so (art. 6,6)			Dropped, instead the directive establishes that competent authorities may specify emission limit values for dioxins and furans (art. 3,3)
Combustion conditions: 850° C, 6% oxygen (art. 4)			Similar Additionally article 4,3 introduces a deroagtion stating that different combustion conditions may be applied, provided that the levels of dioxins and furans emitted are equivalent to, or lower than, those obtained with the technical conditions laid down in the directive
Transitional monitoring requirements (< 1 t/h and ≥ 1 t/h) (final requirements see new plant above) Dust: periodically/continuously CO: continuously/continuously Oxygen: continuously/continuously Temperature of combustion gases: continuously/continuously			Transitional monitoring requirements (classes: plants < 1t/h, and ≥ 1 - <6 t/h) (final requirements see new plant above) Dust: periodically/continuously CO: periodically/continuously Oxygen: periodically /continuously Temperature of combustion gases: continuously/continuously Requirement for plant ≥ 6 t/h Dust: continuously CO: continuously Oxygen: continuously HCL: continuously Heavy metals: periodically HF: periodically SO2: periodically Organic compounds: periodically
Transitional requirement: verification of residence time of combustion gases: at least once after any adaptation of the plant and under the most unfavourable operating conditions, and in any case before 30 June 1994 (final requirements: cf. new plants above)			Transitional requirement for smaller plants: verification of residence time of combustion gases: at least once after any adaptation of the plant and under the most unfavourable operating conditions, and in any case before 30 June 1994 (by 1 December 2000: cf. new plant requirements above) Requirement for large plants: verification at least once after any adaptation of the plant and under the most unfavourable operating conditions

Initial Commission Proposal (88/C75/06)	Economic and Social Committee Opinion (88/C318/02)	European Parliament Opinion (OJ C/1989/69/219)	Finally adopted Directive 89/429/EEC
Monitoring results indicating compliance during transitional period (< 1 t/h and ≥ 1 t/h) (final requirements see new plant above): Temperature and oxygen content: minimum values to be observed at all times/ibid. CO: limit value for daily average/ibid. Dust: concentration values measured in accordance with the rules laid down by the competent authorities under articles 3, 4 and 5 must not exceed the limit value / a) none of the weekly averages measured may exceed the limit value, b) none of the daily averages measured may exceed the limit value by more than 30 %	No specific point made with respect to existing plant	Dust: concentration values measured in accordance with the rules laid down by the competent authorities under articles 3, 4 and 5 must not exceed the limit value / a) none of the weekly averages measured may exceed the limit value, b) none of the daily averages measured may exceed the limit value	Monitoring results indicating compliance during transitional period (first value for plants < 1t/h, second for ≥ 1 - <6 t/h) (final requirements see new plant above): Temperature and oxygen content: minimum values to be observed at all times/ibid. CO: limit value for daily average/ limit value for hourly average Dust: concentration values measured in accordance with the rules laid down by the competent authorities must not exceed the limit value / a) none of the moving seven-day averages measured may exceed the limit value, b) none of the daily averages measured may exceed the limit value by more than 30 % Monitoring results indicating compliance for plants ≥ 6 t/h: Temperature and oxygen content: minimum values to be observed at all times. CO: a) limit value for hourly average; b) 90% of all measurements taken in any 24-hour period must be below 150 mg/Nm3 Dust: a) none of the moving seven-day averages measured may exceed the limit value, b) none of the daily averages measured may exceed the limit value by more than 30 % Other substances to be continuously measured: a) none of the moving seven-day averages measured may exceed the limit value, b) none of the daily averages measured may exceed the limit value by more than 30 % Substances to be periodically measured: the results of each of the series of measurements defined and determined according to the rules laid down by the competent authorities must not exceed the emission limit values
By 30 June 1994 existing incineration plants have to be equipped with auxiliary burners. When the time these are used exceeds 5% of the operation time of the plant during 5 subsequent days, appropriate measures to re-establish sufficient combustion conditions must be taken (art. 7).			Requirement dropped

Initial Commission Proposal (88/C75/06)	**Economic and Social Committee Opinion (88/C318/02)**	**European Parliament Opinion (OJ C/1989/69/219)**	**Finally adopted Directive 89/429/EEC**
When plants exceed limit values, competent authorities must take appropriate urgency measures to re-establish compliance. Plants that don't conform to these measures can not continue operation. (art. 8,1)			Slightly changed: article 7,1 establishes that plants exceeding emission limit values must not continue to operate any longer while failing to comply; and the competent authority shall take appropriate measures to ensure it is modified or no longer operated Additionally: in the case of breakdown of the purification devices the operator shall reduce or close down operations as soon as practicable and until normal operations can be restored (art. 7,2)
The public must have access to the results of measurements of emissions and operation conditions (art. 9)		Parliament suggests adding a further paragraph, stating that competent authorities and green groups are allowed to inspect plants and to measure emissions.	Control results must be made available to the public, subject to the respect of provisions applicable in respect of commercial secrecy (art. 8)
Member States can impose stricter standards than those established by the directive (art. 11)			Dropped, but emission limits may be laid down for other pollutants than those regulated by the directive (art. 3,3)
The directive must be transposed into national law by 30 June 1989			The directive must be transposed into national law by 1 December 1990

Annex 4.C: National emission limits applicable prior to, and after, the adoption of the MWI Directive in Germany and France

Table 4.C. 1: German and French regulation towards MWI before the Directive's adoption

Pollutant	TA Luft 1986 (all capacities)	arrêté du 9 juin 1986 (> 6 t/h or including an oven of a capacity larger than 3 t/h)
Dust	30	50
Pb+Cr+Cu+Mn	5 (inc. Sb+CN+F+Pt+Pd+Rh+V+Sn)	5 (incl. Zn, Ni, Sn, Ag, Co, Ba; excl. Mn)
Ni+As	1 (including Co+Se+Te)	1 (only As; Ni included above)
Cd+Hg	0.2 (including TI)	0.3
HCL	50	100
HF	2	
SO2	100	
CO	100	
Organic compounds	20	
Gaseous hydrocarbons		10

Table 4.C. 2: German and French regulation towards MWI after the Directive's implementation

Pollutant	German 1990 Incineration Ordinance To be met by 1 Dec. 1996		Arrêté du 25 janvier 1991 To be met by 1 Dec. 1996 by existing incinerators > 6t/h
	daily average emission limits	Half-hourly emission limits	Moving seven-day average
Dust	10	30	30
HCL	10	60	50
CO	50	100 (hourly limit)	100
			Each measurement
HF	1	4	2
SO2	50	200	300
Organic compounds	10	20	20
NO2	200	400	
	Average series of measurement		
Pb+Cr+Cu+Mn	0.5 (including Sb+As+Co +Ni+V+Sn)		5
Ni+As	(included above)		1
Cd+Hg	0.05 (Cd+TI) 0.05 (Hg)		0.2
Dioxins	0.1 ng/nm^3		

Annex 4.D: Waste incineration cost data for France and Europe

1) European Commission (1997)

Hypotheses: Cost estimates apply to new plant built on 'greenfield' sites and are calculated at UK price levels for mid 1996. The cost data are taken from core countries (Germany, the UK and France) and adjusted to the UK price basis. Covered are costs associated specifically with the emission control function of incinerators; excluded are plant items not required for pollution control. Residue handling is included. All this holds for capital and operating and maintenance costs. The capital recovery for the UK was estimated to be 15 years at 8%.

Table 4.D. 1: Estimated cost of alternative abatement technology for MWI plant

Flue gas flowrate (dry)	Nm3/h	14,67	29,34	58,68	88,01	117,35	234,7	352,05	469,41
number of streams	No.	1	1	2	2	2	2	3	4
stream capacity	t/h	2,9	5,7	5,7	8,6	11,4	22,8	22,8	22,8
plant utilisation factor	%	80	80	80	80	80	80	80	80
annual operation	hours	7008	7008	7008	7008	7008	7008	7008	7008
Waste throughput	1000 t/y	20	40	80	120	160	320	480	640
Total flue gas treatment capital costs									
Capacity	1000 t/y	25	50	100	150	200	400	600	800
control option A	Ecu/t	102,00	60,40	43,40	35,40	31,20	23,60	21,40	19,40
control option B	Ecu/t	121,60	72,20	51,80	42,20	37,20	28,20	25,50	23,20
control option C	Ecu/t	139,60	82,80	59,50	48,50	42,70	32,30	29,20	26,60
control option D	Ecu/t	139,20	82,60	59,30	48,30	42,50	32,20	29,10	26,50
control option E	Ecu/t	155,60	92,40	66,40	54,10	47,60	36,10	32,60	29,70
control option F	Ecu/t	243,60	144,60	103,80	84,60	74,50	56,40	51,00	46,40
Average annual flue gas treatment operating and maintenance costs									
control option A	Ecu/t	12,90	9,00	8,10	7,10	6,60	5,70	5,50	5,40
control option B	Ecu/t	14,80	10,30	9,20	8,10	7,50	6,50	6,30	6,10
control option C	Ecu/t	17,30	12,20	11,10	9,80	9,10	8,00	7,70	7,60
control option D	Ecu/t	22,20	16,60	15,50	14,00	13,30	12,10	11,80	11,60
control option E	Ecu/t	24,00	17,90	16,60	15,00	14,20	12,80	12,50	12,30
control option F	Ecu/t	32,10	23,70	21,70	19,60	18,50	16,70	16,30	15,90

Table 4.D. 2 Waste incineration pollution control options expected to be applied in 2000

	France							
Plant capacity (1000 t/y)	**25**	**50**	**100**	**150**	**200**	**400**	**600**	**>600**
Control option A	x	x	x	x				
Control option B			x	x	x	x		
Control option C			x	x	x	x		
Control option D			x	x	x	x		x
Control option E			x	x	x	x		x
Control option F					x	x	x	
	Germany							
Control option A								
Control option B								
Control option C				x	x	x	x	
Control option D								
Control option E								
Control option F				x	x	x	x	x
Others		x	x					
	UK							
Control option A		x	x	x				
Control option B				x	x	x	x	
Control option C					x			
Control option D								
Control option E								
Control option F								

Description of control options:

A: acid gas control through semi-dry system followed by bag filter for particulate removal; solid residues: lime and fly/ash

B: acid gas control through semi-dry system followed by bag filter for particulate removal plus carbon injection for dioxin and mercury control and NOx reduction by flue gas recirculation

C: acid gas control through semi-dry system followed by bag filter for particulate removal plus carbon injection for dioxin and mercury control and SNCR for NOx control (if residues treated to meet leaching standards the plant is compatible with the draft incineration directive)

D: wet scrubbing system for acid gas removal, ESP for dust control; plus effluent treatment plant to precipitate heavy metals as sludge for disposal; together with fly ash from the ESP as solid residue

E: ESP for fly ash removal, multi stage wet scrubber for acid gas removal and effluent treatment plant discharging saline water (in line with draft directive standards except for dioxins and NO_x)

F: bag filter for fly ash removal, followed by multi-stage wet scrubber for acid gas abatement, followed by SCR for NO_x control and activated carbon for dioxin and mercury removal. Remaining particulates are removed by a bag filter. Liquid effluents from the scrubber are evaporated to dryness (if solid residues are treated to meet the leeching standards the plant is compatible with the draft directive's standards)

2) SOFRES 1998a & 1998b

Hypotheses: new plant, located on greenfield sites, costs without subsidies; financing by credit at 100 %; operating margin taken into account; nominal capacity of x kt/a of municipal waste at 2100 th/t; rate of utilisation: 86%; 2 streams; regulation: 'arrêté' of January 1991; energy recovery: electricity; destination of slag: platform at 10 km and use in public works; destination of residues from off-gas cleaning of MWI: landfill for special waste at 200 km

Table 4.D. 3: Incineration costs for compliance with the ministerial order of January 1991 in France

total net cost (€/t)	Dry		Semi-humid		Humid	
Capacity	Min	Max	Min	max	Min	max
37,5 kt/a (5 t/h)	77,75	85,37	76,99	84,61	75,46	85,37
75 kt/a (10 t/h)	70,13	77,75	69,36	76,99	69,36	78,51
150 kt/a (20 t/h)	58,69	67,84	59,46	67,84	60,22	69,36

Hypotheses: existing plant, nominal capacity of 75 kt/a (10 t/h) of municipal waste at 2100 th/t; rate of utilisation: 86%; 2 streams; regulation: 'arrêté' of January 1991; energy recovery: case A electricity, case B cogeneration with maximum electricity production; gas washing system: case A semi-humid, case B humid; destination of slag: platform at 10 km and use in public works; destination of residues from off-gas cleaning of MWI: landfill for special waste at 200 km

Table 4.D. 4: Differences in incineration costs according to energy recovery option

	Case A		Case B	
Total net cost	min	Max	min	Max
(€/t)	69.36	76.99	60.22	71.65

3) EUNOMIA Research & Consulting (2002)

Hypotheses: current cost data, mostly from the end of the 1990s, only grate incinerators compared, total cost of plant, not only abatement equipment. In some countries there exist taxes for plant with energy recovery which have to be added. The revenues for energy supply and costs of ash treatment vary widely across countries and are not taken into account either.

Table 4.D. 5: Incineration costs across European countries

	18,7 kt/a	37,5 kt/a	50 kt/a	60 kt/a	75 kt/a	100 kt/a	120 kt/a	150 kt/a	200 kt/a	300 kt/a	600 kt/a
Austria				326				159		97	
Belgium								71-75			
France	118-129	91-101 86-101			80-90			67-80			
Germany			250						105		65
Luxembourg							97				
UK						69			47		

Chapter 5 Efficiency Properties of the Directive's Contents

1 Introduction

To complement the arguments in the previous chapter, which used an indirect approach for assessing the efficiency of the 1989 Directive, the current chapter analyses the same question by applying a direct approach focusing on the efficiency of the Directive's objectives.

In a European context, the inefficiency problem of policies is frequently related to the heterogeneity of national situations, which are difficult to accommodate under a European policy. The issue is the cost an EU policy implies for different polluion sources. An economic criterion for evaluating a policy in this respect is cost-effectiveness. Cost-effectiveness refers to the extent to which the aggregate pollution abatement costs introduced by a regulation are minimised across polluters (OECD, 1997: 32). Cost-effectiveness requires an allocation of abatement effort amongst various plants which takes into account the abatement costs of individual plants. In practice this means that the less costly it is for a plant to reduce pollution, the more abatement effort it should undertake. Where abatement costs differ across plants, homogenous pollution reduction requirements are clearly not cost-effective.

The 1989 EU Directive defined emission standards that were differentiated according to plant age ('existing' and 'new' plant) and to several incineration capacity categories. A second approach to an economic assessment of the Directive's efficiency, therefore, focuses on the policy's contents and investigates whether the Directive's differentiation of objectives was sufficient to allow for a cost-effective allocation of abatement effort amongst pollution sources. To this end, the Directive's differentiation needs to be evaluated in relation to the cost characteristics of pollution sources regulated at the time at which the EU Directive was issued. The chapter studies the following factors which are decisive for the relative compliance costs faced by the different countries: the structure of the countries' plant parks (capacity dispersion because of economies of scale in abatement; plant vintage because of investment cycles), the absolute share of municipal waste incineration in overall municipal waste treatment, and the pre-existing domestic environmental regulation and equipment of plants with abatement technology. This evaluation also constitutes the basis for assessing the countries' implementation paths with respect to the ultimate question of the impact of the implementation processes on the efficiency of policy outcomes relative to the policy objectives defined by the Directive in the following chapter.

Section 2 provides information on the country contexts prior to the Directive's implementation, discussing plant park characteristics and pre-existing domestic regulations in order to give a first idea of the challenge posed to different countries by the Directive. Section 3 discusses the cost-effectiveness of the Directive's standards relative to the heterogeneity found in France, Germany, the Netherlands and the United Kingdom, and section 4 concludes.

2 Context of municipal waste incineration in EU Member States around 1990

As discussed in chapter 3, the heterogeneity between countries is one of the major challenges EU environmental policy making faces. The initial situation in the Member States with respect to municipal waste incineration and related environmental legislation is decisive for the relative implementation costs this policy implied. Therefore, available information that gives an idea about the homogeneity or heterogeneity of the situation across European Member States prior to the implementation of the 1989 municipal waste incineration Directives is evaluated below.

As will be shown, the situation in countries was clearly heterogeneous, in terms of the role played by municipal waste incineration in overall municipal waste treatment (% of municipal waste incinerated), the number of plants, the capacity classes the plants belong to, as well as the pre-existing regulation of atmospheric pollution caused by these plants and the related pollution abatement equipment installed. While for some countries the implementation of the Directive was demanding, others practically already fulfilled its requirements when the Directive was adopted.

2.1 Importance of municipal waste incineration, plant number and capacities across Member States: a heterogeneous picture

Incineration provides an important treatment option for municipal solid waste in the EU. When with post-war economic growth, increased use of packaging material and greater consumption the waste quantities rose, incineration provided an advantage in that it reduces waste -by converting solid waste into a gaseous effluent and a solid residue (ash)- to about 10% of its original volume and about 30% of its original mass. Also, waste incineration offers opportunities for energy recovery, particularly in the larger incinerators. The recovered energy can be used for district or process heating and electricity generation. This explains why various countries established large capacities of municipal waste incineration during the 1960s and 1970s (European Commission, 1997: iii and 2-5). Nevertheless, the share of incineration in overall municipal waste treatment alternatives, in the early 1990s, varied considerably between countries (cf. Table 5.1).

Table 5. 1: Share of waste incineration in overall municipal solid waste treatment and number of municipal waste incineration plants in 1993

	% of MSW incinerated	Number of MWI plants
Austria	11	2
Belgium	54	24
Denmark	36	30
Finland	2	1
France	42	225
Greece	0	0
Germany	36	49
Ireland	0	0
Italy	16	28
Luxembourg	75	1
Netherlands	35	10
Norway	22	18
Portugal	0	0
Spain	6	14
Sweden	47	21
Switzerland	59	30
UK	8	31
Average	*26.4*	*28.5*
Total		*484*

Source: European Commission, 1997, citing Rijpkema, 1993

Focussing on those countries that were Members of the European Union in the early 1990s when the Directive came into force, the relevant group covers Belgium, France, Germany, Italy, Luxembourg, the Netherlands, Denmark, Ireland, the United Kingdom, Greece, Spain and Portugal. Austria, Finland and Sweden only joined in 1995. The above table shows that the importance of municipal waste incineration differs substantially between these countries. While Ireland, Portugal and Greece[38] are reported to not have practiced any incineration of municipal waste, approximately 35% of the municipal waste was incinerated in Germany, the Netherlands and Denmark, and in Luxembourg the share was even 75%. Belgium and France, with 54% and 42% respectively, show high shares of municipal waste incineration in total municipal waste treatment as well, while the UK with 8% is situated at a lower edge of the spectrum.

[38] Hannequart (1993: 238) reports that Greece, by the end of the 1980s, operated one incinerator of very low capacity.

Differences between countries become even more striking when considering the number of municipal waste incineration plants operated in the early 1990s. One extreme case amongst those countries that operate municipal waste incinerators is Luxembourg with only one plant. Being a small country, Luxembourg presents obviously a specific case. The fact that it has only one incinerator puts into perspective the incineration share of 75% mentioned before. Amongst the other countries, the Netherlands and Spain with 10 and 14 plants respectively are situated at the lower spectrum with respect to the number of plants, while the majority of European Union members operated between 20 and 30 plants approximately. Germany lies slightly higher, with 49 plants, while France quite obviously constitutes another extreme case, with a way above-average number of 225 reported waste incinerators. The data collected for the IMPOL project[39] comes to comparable numbers for the Netherlands and Germany, but found a higher number of plants for the UK (39). For France it suggests that the number given above is considerably underestimated, reporting about 100 plants more, even though only 'existing' plants were considered in this data base (cf. Table 5.2).

Plant capacity and economies of scale

These data give a first indication of the importance the municipal waste incineration Directives had for the different Member States. A more thorough idea about what this policy implied for the countries in terms of abatement costs, however, can only be gained when qualifying the data with information on plant capacities, pre-existing domestic environmental standards and applied abatement equipment (see below).

Table 5. 2: Number of 'existing' plants and capacity classes in the case study countries, early 1990

	Number of plants	capacity > 6 t/h	Capacity ≤ 6 t/h
France (*)	≈ 320	≈ 66	≈ 254
Germany	48	Almost all	Very few
The Netherlands	13	13	0
The United Kingdom	37	32	5

(*)This estimation of the number of plants is based on four municipal waste incinerator inventories. Two were provided by the Ministry of the Environment and represent the plants' state of compliance in 1998 (for large incinerators) and 2000 (for small incinerators). One was provided by the ADEME, presenting the MWI park in 1998, and one is the 1995 ITOM inventory which presents the park in 1993. All these inventories included different, limited information about the plant park. For an in depth description of these inventories and the hypotheses made to merge them cf. Annex 6.B. The list of all French plants is given in Annex 6.C. For a variety of plants information on capacity was lacking altogether, this is indicated in the relevant tables in chapter 6 with NV. - Source: Bültmann and Wätzold, 2000; Eames, 2000; Lulofs, 2000 and 2001; Schucht, 2000; and Schucht et al., 2001

Hannequart (1993: 238), referring to the number of municipal waste incinerators operated in the European Union at the end of the 1980s, also reports a higher number overall than given in Table 5.1. His inventory reports 525 plants, of which 38% had a capacity greater than 6 t/h of waste incinerated, 51% a capacity between 1 t/h and 6 t/h, and 11% a capacity below 1 t/h. It is worth noting that in 1986/87 88% of the waste was incinerated in those plants that had a capacity superior to 6 t/h, while only 1% was incinerated in plants of a capacity below 1 t/h. Capacity is crucial because of economies of scale prevalent in abatement investment and the limited differentiation of emission limit values in the European Directive. Focussing on the number of plants by capacity groups, Table 5.2 shows for the four countries studied in more detail throughout this thesis that France, with approximately 254 'existing' incinerators of a lower capacity (i.e. below 6 t/h), was clearly disadvantaged in terms of relative implementation costs. The United Kingdom holds a medium position, while Germany and the Netherlands basically ran large incinerators only.

[39] The project involved four research institutes: CERNA, Ecole des Mines de Paris; SPRU – Science and Technology Policy Research, University of Sussex; CSTM, University of Twente; and UFZ Leipzig-Halle.

Table 5.3 gives an indication of how control capital costs for one abatement option able to meet the Directive's requirements differ with plant capacity (for a further illustration cf. Annex 5.A). Unfortunately no cost data were provided for small incinerators. Note that as a rule, a capacity of 25 kt/y equals approximately a capacity of 6 t/h (European Commission, 1997). To some degree, differentiated emission limit values for different capacity classes of plant make up for those abatement cost differences that are related to capacity. Standards were however not differentiated for plants falling within the capacity group of '>6 t/h' for which cost data are provided in the table below.

Table 5. 3: Economies of scale for a basic emission control system able to meet the 1989 Directive's requirements

Plant capacity	**Waste throughput**	**Bag filter**		**Semi dry scrubber**		**Total flue gas treatment**	
1000t cap/y	**1000 t/y**	**1000 Ecu**	**ECU/tcap/y**	**1000 Ecu**	**ECU/tcap/y**	**1000 Ecu**	**ECU/tcap/y**
25	20	380	15,20	530	21,20	2550	102,00
50	40	450	9,00	630	12,60	3020	60,40
100	80	640	6,40	910	9,10	4340	43,40
150	120	780	5,20	1100	7,33	5310	35,40
200	60	920	4,60	1300	6,50	6230	31,15
400	320	1390	3,48	1970	4,93	9440	23,60
600	480	1890	3,15	2670	4,45	12810	21,35
800	640	2290	2,86	3240	4,05	15540	19,43

Note: the data refer to control option A presented in annex 4.D. - Source: European Commission, 1997

When only considering large plant, for which emission limit values were not differentiated in the Directive, a comparison between Germany and France allows making a first guess on relative differences in implementation costs between the countries. Given the share of waste incineration of 36% in Germany and of 42% in France as reported in Table 5.1, taking into account the population of these two countries, and comparing this to the number of incinerators suggests that the 48 large German incinerators on average must be of substantially larger capacity than the 66 large French plants. This implies that Germany would have encountered comparatively lower marginal abatement costs overall than France. However, the data presented here only give a first indication; a more thorough analysis of the capacity dispersion in these four countries is carried out in section 3 below. Let us now turn to the second important determinant of abatement cost differences imposed by the implementation of the 1989 European Directives, the pre-existing regulation in different Member States and the related pollution abatement equipment.

2.2 Pre-existing legislation differs significantly across Germany, the Netherlands, France and the United Kingdom[40]

The countries studied have not developed their domestic environmental regulation in parallel. Instead, they show considerable differences as to the speed with which regulation directed at atmospheric emissions from municipal waste incinerators has been developed. This has an impact on the requirements actually implemented on a national level in parallel with the EU Directive's implementation. It also has an impact on the abatement investment which had to be put into place to upgrade 'existing' plant to the Directive's requirements. In order to take account of the countries' variety before the implementation of the 1989 Directives, the domestic regulation in France, Germany, the Netherlands and the UK, which was in force at the time the EU Directive was adopted, is presented.

[40] If not otherwise specified, the following information is taken from Bültmann and Wätzold, 2000 (for Germany); Eames, 2000 (for the UK); Lulofs, 2000 (for the Netherlands); and Schucht, 2000 (for France) and from two comparative studies Lulofs, 2001; and Schucht et al., 2001.

2.2.1 The situation in Germany prior to the EU Directive's implementation

Amongst the countries studied here, Germany belongs to the group of environmental 'leaders' with strict pre-existing legislation on emissions from waste incinerators. This was the TA Luft (Technische Anleitung zur Reinhaltung der Luft - Technical Instructions on Air Quality Control) of 1986, an amendment of the TA Luft 1974. These instructions are specifications of the German Federal Imission Control Act[41] of 1974 (Bundesimmissionsschutzgesetz - BImSchG), the basis of all German pollution control regulation.

The emission limit values imposed by the TA Luft 1986 were almost identical to, and in some cases stricter than, those the 1989 EU Directive had specified for incinerators of a nominal capacity above 6 t/h (Table 5.4). The requirements were to be applied by plants demanding operation permission from 1 March 1986 onwards. Older plants emitting more than 1.5 times (1 time) the emission limit values defined by the TA Luft 1986 were to meet the TA Luft requirements by 1 March 1991 (1 March 1994), except for plants whose operators declared in written form that their plant was to cease operation at latest by 28 February 1994.

Table 5. 4: EU Directive and German pre-existing emission limits in mg/m3 for existing incinerators

Pollutant	EU Directive	TA Luft 1986
	Deadline: 1 Dec. 1996 (existing plant > 6t/h; 1 Dec. 1990 (new plant > 3 t/h)	Deadline/all capacities: cf. text above)
Dust	30	30
Pb+Cr+Cu+Mn	5	5 (including Sb+CN+F+Pt+Pd+Rh+V+Sn)
Ni+As	1	1 (including Co+Se+Te)
Cd+Hg	0.2	0.2 (including TI)
HCl	50	50
HF	2	2
SO_2	300	100
CO	100	100
Organic Compounds	20	20

Source: TA Luft 1986

Therefore, the provisions of the Directive were in principle covered by the TA Luft 1986 and the European Directives, hence, implied little or no additional cost to bring municipal waste incinerators into compliance. Nevertheless, for formal reasons, the TA Luft 1986 was not recognised as transposition of the European Directives (cf. chapter 6).

2.2.2 The situation in the Netherlands prior to the EU Directive's implementation

Amongst the countries studied here, the Netherlands represent the second environmental 'leader' country as far as MWI regulation is concerned. Back in 1985 the Netherlands had issued the Guideline Combustion 1985 defining emission limit values that were very close to those of the EU Directive. However, this law covered a smaller scope of pollutants and was addressed at new municipal incineration plants only.

By the time the EU Directive was adopted, the Netherlands were about to impose significantly stricter and more far reaching domestic limits under the 1989 Dutch Guideline Combustion (Richtlijn Verbranden)[42]. This was linked to ongoing German debates on regulation and rising public concern about dioxin pollution. The Guideline Combustion 1989 was issued on 15 August 1989, shortly after the EU Directive had been

[41] Immission means the impact of pollutants on plants, animals and man (cf. http://www/bmu.de).

[42] Richtlijn Verbranden 1989, Stert. 1989, nr. 188.

adopted. It defined emission limit values (applying to new and existing plant) much stricter than those required by the EU Directive and also covered the additional pollutants NO_x, dioxins and furans (Table 5.5).

Table 5. 5: EU Directive and Dutch pre-existing emission limits in mg/m3 for existing incinerators

Pollutant	EU Directive	Guideline Combustion 1985	Guideline Combustion 1989
	Deadline: 1 Dec. 1996 (existing incinerators > 6t/h); 1 Dec. 1990 (new plant > 3 t/h)	**all capacities**	**Deadline: 30 November 1993 all capacities**
Dust	30	50	5
Pb+Cr+Cu+Mn	5	5 (including Ni+As+Sb+V+Sn +Co+Se+Te)	1 (including Ni+As+Sb+V+Sn +Co+Se+Te)
Ni+As	1	(included above)	(included above)
Cd+Hg	0.2	0.1 Cd 0.1 Hg	0.05 (Cd) 0.05 (Hg)
HCl	50	50	10
HF	2	3	1
SO_2	300		40
CO	100		50
Organic Compounds	20		10
NOx			70
Dioxins and furans			0,1 ng I-TEQ/m3

Source: Richtlijn Verbranden 1985 and 1989

As in the German case, the Commission did not recognise the Incineration Guideline 1989 as formal transposition (cf. chapter 6). Nevertheless, similarly to Germany, the Netherlands were advanced in their domestic regulation of atmospheric emissions from municipal waste incineration plants. Although not all incinerators did yet comply with these emission limits, the additional costs related to implementing the European Directive can be considered as low.

2.2.3 The situation in France prior to the EU Directive's implementation

French air pollution regulation directed at municipal waste incinerators is generally laid down in ministerial orders, so called "arrêtés" or technical instructions. Prior to the integration of the European Directive into national law, two legal texts regulated atmospheric emissions from municipal waste incinerators. One was addressed at new plants, the other and older one at both existing and new plants. For ***new plant*** and plants being subject to an extension increasing their capacity, authorised from 10th July 1986 onwards, a ministerial order (arrêté) of 9th June 1986[43] defined atmospheric emission standards for a variety of pollutants (cf. Table 5.6 below). Compared to the 1989 Directive, this ministerial order covered a different package of pollutants and did not include emission limits for HF, SO_2 and CO. Emission limits were on average less strict than those defined in the Directive. However, a larger range of heavy metals was covered, and the emission limit values for these pollutants were comparable with the European standards (cf. with Table 4.2).

All in all, the 1986 ministerial order refers to the same categories of abatement technology as the 1989 European Directive, i.e. requirements for compliance with the two texts, as far as standards for atmospheric emissions are concerned, do not impose a 'jump' in technology. Plants authorised according to the 1986 ministerial order from 10th July 1986 onwards and before 1st December 1990 are 'existing' incinerators in the sense of the 1989 European Directives. However, not many incinerators where authorised during this period.

[43] Arrêté du 9 juin 1986 relatif aux installations d'incinération de résidus urbains (no longer applicable).

Table 5. 6: Limits for Atmospheric Emissions According to the Arrêté of 1986

Capacity	**≤ 1 t/h**	**1 < c ≤ 6 t/h (no oven > 3 t/h)**	**> 6 t/h (or incl. an oven of > 3 t/h)**
Dust	600 mg/Nm3	150 mg/Nm3	50 mg/Nm3
HCL		250 mg/Nm3	100 mg/Nm3
Gaseous hydrocarbon	30 ppm	10 ppm	10 ppm
Cu, Pb, Zn, Ni, Cr, Sn, Ag, Co, Ba (particulate)			5 mg/Nm3
Cu, Pb, Ni, Cr, Sn, Ag, Co, Ba (particulate)		6 mg/Nm3	
Hg+Cd (particulate and gaseous)			0,3 mg/Nm3
As			1 mg/Nm3
vertical speed of combustion gas emission must be above	8 m/s	8 m/s	12 m/s
combustion conditions	Combustion gas must for at least 2 seconds reach a temperature of at least 750 °C in the combustion or post-combustion room and during this time at least contain 7% of oxygen. In normal operation the combustion or post-combustion gas must contain at least 0,1% de carbon monoxide (expressed as 7% CO2) and more than 7% oxygen.		

Source: Arrêté du 9 Juin 1986

For ***existing incinerators*** solely emission limits for dust existed before 1991 (cf. Table 5.7), defined in a technical instruction of 1972 (Instruction technique du 6 juin 1972)[44]. Owing to this, all municipal waste incineration plants in France were equipped with off-gas de-dusting facilities (Milhau & Pernin 1994). Generally, these were electric filters for large incinerators and mechanic de-dusters (such as cyclones) for small incinerators.

Table 5. 7: Limits for Atmospheric Emissions According to the technical instruction of 1972

	< 1 t/h	**1 < c ≤ 4 t/h**	**4 < c ≤ 7 t/h**	**> 7 t/h**
Dust	1 g/Nm^3	0.6 g/Nm^3	0.25 g/Nm^3	0.15 g/Nm^3

Source: Instruction technique du 6 juin 1972

In conclusion, at the time the EU Directive was adopted and transposed into national law, France showed a significant backwardness with respect to the incinerators' equipment with clean air technology. This, together with the high number of MWI plants of low capacity in this country, means that considerable costs could be expected to result from the implementation of the 1989 EU Directive in France.

2.2.4 The situation in the United Kingdom prior to the EU Directive's implementation

The UK presents an extreme case, where MWI plants prior to 1989 were basically not subject to any system of authorisation, detailed emission limits or emission monitoring. The limited regulatory control that did exist was exercised under the general requirements of the Clean Air Acts of 1956 and 1968 and was limited to the regulation of chimney height, grit and dust equipment, and the emission of dark smoke. The government was aware of the laxity of the existing regulation. In 1986, there were suggestions to bring municipal waste incinerators under the control of an integrated (covering air, water and land issues) inspectorate (Her Majesty's Inspectorate of Pollution - HMIP), in the course of changes to be made to UK legislation in order to make it compatible with existing and prospective EC environmental Directives. However, these changes were not implemented, so that eventually, under pressure from the EC, the UK was forced to issue 'stop-gap' regulations in March 1989. In April 1989, the Health and Safety (Emissions into the Atmosphere) (Amendment) Regulations 1989 brought all incineration processes with a capacity greater than 1 t/h under HMIP control. In 1991 the Commission took formal infringement procedures against the UK and transposition of the EU Directive was reached in November of the same year (cf. chapter 6). Prior to the implementation of the

[44] Instruction technique du 6 juin 1972 relative aux installations d'incinération des résidus urbains (no longer applicable).

1989 European Directives, UK municipal waste incineration plants were equipped with only basic pollution abatement (electrostatic precipitators or cyclones), and investment necessary for upgrading plants to the Directive's standards could therefore be considered as significant.

2.3 Synthesis

From what was said before it is obvious that the Directive met with very heterogeneous situations in the Member States. This heterogeneity concerns the importance of municipal waste incineration in this waste's treatment overall, the number of plants and the structure of the plant park, as well as the pre-existing regulation and equipment of municipal waste incinerators with abatement technology.

On first sight the data suggest that Germany and the Netherlands were in a comparatively lucky position, with an on-average share of waste incineration in overall municipal waste treatment in a European wide comparison, a rather limited number of plants of high capacities, and advanced pre-existing domestic regulation. France seems to hold a medium position when it comes to pre-existing regulation, but the number of plants affected by this was low. Given the hence limited equipment with abatement technology and the way above-average number of incinerators, a high share of which were furthermore plants of very low capacity, it clearly encountered a disadvantage when it comes to implementation costs. The UK was not in a better position when it comes to pre-existing abatement equipment, but at least the majority of 'existing' municipal waste incinerators were large plants and overall waste incineration played no prominent role in municipal waste treatment. Although upgrading of existing plant would have implied heavy investment, overall costs expected could be assumed to be lower than in the French case. This is investigated in more detail below with the help of a set of indirect cost-indicators.

3 To what extent is the Directive's standard differentiation cost-effective?

In order to be able to evaluate the cost-effectiveness properties of the 1989 municipal waste incineration Directive, it is useful to recall what economists have in mind when talking about efficiency and cost-effectiveness of policies and specific policy instruments.

3.1 Why the focus on cost-effectiveness?

Ideally, environmental regulations would define efficient levels of pollution control where marginal abatement costs equal the marginal social damage of pollution (Baumol and Oates, 1988) and where net benefits (total benefits in the form of reduced damage through pollution control minus total costs of pollution abatement and remaining damages to the environment) are hence maximised. What complicates the *definition of efficient regulations* in practice is that it is generally ambient concentrations of pollution and not directly emissions that cause damage to the environment. What needs to be regulated with respect to specific firms, however, are their emissions. Unfortunately, for reasons related to space (e.g. acid rain damage to forests will be higher from pollution sources situated in the vicinity than from more distant sources or from emissions that are largely blown out to the sea) and time (e.g. damage may occur in the future, or be subject to seasonal variations) emissions are not perfectly connected to ambient concentrations and thus to damages (Kolstad, 2000).

In practice, therefore, it is often too difficult to define regulations that correctly target damage and ambient concentrations, and regulations therefore frequently set some second best overall goal for emission limits that are only imperfectly related to efficient levels of pollution. If emission targets are defined in the latter way, the goal of the regulation should be to control individual polluters in such a way as to achieve the given emission target in the least cost way, i.e. cost-effectively.

For an *analysis of environmental policies* there is an additional reason to focus on cost-effectiveness rather than on efficiency. Indeed, owing to a lack of data on environmental and health damages in monetary terms of major pollutants emitted by waste incinerators, such as dioxins, and of the related benefits of emission reductions, we cannot apply a full efficiency view in analysing the 1989 Directive but only the more modest criterion of the cost-effectiveness of the Directive's objectives. Where possible, however, we complement the cost-effectiveness view by qualitative considerations about environmental benefits.

3.2 Can emission standards be cost-effective?

Cost-effectiveness requires that differences in pollution control costs across various pollution sources are reflected and that the regulation correctly matches required emission reductions with control costs, by equating marginal control costs across pollution sources. This implies that the less costly it is for a plant to reduce pollution, the more abatement effort it will undertake. Putting this in a more general way, cost-effectiveness arises from properly taking into account local conditions, i.e. cost characteristics at the level of a pollution source.

When discussing the pros and cons of economic instruments at one side, and of command-and-control policies at the other side, economic literature generally argues that economic (or price) instruments, such as emission taxes or tradable emission permits, in terms of their cost-effectiveness properties are superior to quantity based instruments, such as pollution standards (e.g. Baumol and Oates, 1988; Baumol, 1972; Cropper and Oates, 1992; but also OECD, 2001). By leaving the choice of the abatement level to the regulated agent, economic instruments, unlike quantity based instruments, take advantage of the large differentials in abatement costs across polluters and minimise the overall abatement costs introduced by the regulation.

The theory says that emission standards are not cost-effective because they lack this mechanism to equalize costs (Baumol and Oates, 1988). What is more, standards tend to be uniform. In principle, abatement standards could be equally cost-effective, provided they were differentiated according to individual plant characteristics in such a way that abatement costs would be equalised at the margin. Such a differentiation is decisive, because here it is the regulation that determines the individual abatement effort of the regulated agent. A differentiation of pollution targets equalising marginal abatement costs, however, demands a high level of information about the abatement costs of each firm. Such detailed information about each regulated agent is generally not readily available, or the attempt to correctly design such a differentiated instrument would be subject to high information and regulatory costs. Therefore, abatement standards usually tend to be more or less uniform, differentiated only according to certain broad plant characteristics. The more uniform standards are, the less likely it is that they require the cost-effective abatement level from regulated firms. Environmental economists, therefore, generally suggest that the more uniform an environmental standard, the less likely it is that it will result in a cost-effective allocation of abatement effort between the regulated firms. It should however be added that, in practice, the degree of cost-ineffectiveness entailed by uniform standards depends on plant characteristics of the regulated community. Efficiency losses from uniform regulation are the smaller the more homogeneous the regulated plants are.

3.3 What can be said about the Directive's differentiation of emission standards in terms of cost-effectiveness?

As pointed out before, the couple of European Directives regulating atmospheric emissions from municipal waste incineration plants defines targets which are differentiated to a certain extent. Firstly, the two Directives differentiate targets between plant age, i.e. between 'new' and 'existing' plant. 'Existing' plant were left between 6 and 10 years, depending on their capacity, to adjust to the requirements defined for 'new' plant. Compared to a situation of completely uniform standards this differentiation goes in an efficient direction in that it takes account of the remaining life spans of existent equipment and by this leaves firms some flexibility for decisions in terms of the timing of necessary investment.

Focussing now on the Directive dealing with 'existing' plant, its differentiation of requirements according to plant size (capacity) is indeed important for cost-effectiveness and clearly not chosen by coincidence. Two aspects of the requirements were differentiated according to plant size: the severity of the standards and the deadlines by which plants needed to comply. The fact that emission limit values are stricter for larger plant goes in the direction of cost-effectiveness. The relationship between size and abatement costs is straightforward as economies of scale in waste incineration and the respective pollution abatement are widely acknowledged for the case of municipal waste incineration (European Commission, 1997), making abatement relatively less costly for larger plant.

It should be mentioned that the setting of weaker standards for smaller plants can additionally be justified by benefit considerations. Small incinerators are predominantly found in rural areas, where the population density is lower and with this the external costs caused by emissions. Therefore, the standard differentiation according to plant size indirectly introduces a differentiation of targets between rural and urban areas.

The fact that smaller plants were left with more time than large plants until they had to comply with final emission limits, from a theoretical point of view, is not relevant for cost-effectiveness. In practice, however, it has been relevant, at least for France, because a large number of the small plants were eventually closed. This differentiation, therefore, left smaller municipalities more time to develop alternative waste treatment capacities while at the same time making use of the remaining life-span of plants over a longer period. The Directive therefore made provisions to facilitate a phasing out of plants of inefficient size.

Compared to the benchmark of a completely uniform regulation, one can conclude that the Directive shows some elements of cost-effectiveness. Nevertheless, given that the Directive distinguishes requirements only according to several broad capacity groups of plants, the differentiation can be assumed to hardly have been sufficient to take into account the specific local circumstances in all Member States and of all plants falling under this regulation. It therefore cannot be expected to lead to an equalisation of marginal abatement costs across pollution sources.

The ultimate goal of this chapter is to investigate the relative cost-effectiveness of the Directive's standards for France, Germany, the Netherlands and the United Kingdom. This constitutes the basis to assess the cost-effectiveness of the implementation paths pursued by the four countries relative to the cost-effectiveness of the Directive's requirements in the following chapter. For this, we need to evaluate costs implied by the Directive at a level of individual pollution sources. Ideally, such an evaluation would be based on cost data on the level of each individual plant and on precise information about source characteristics, as the problem is one of efficiently coping with inter-source differences. But comparable quantitative data on actual costs per plant were not available for all four countries. The analysis therefore relies on indirect indicators more easily observable than costs, and which reflect the level of costs involved. This approach is also justified by the different time-frames of regulation in the four countries studied, i.e. the fact that pre-existing environmental legislation and the standards to be implemented during the 1990s differed between the countries. This implies that real costs of abatement incurred in the 4 countries in the 1990s are not comparable.

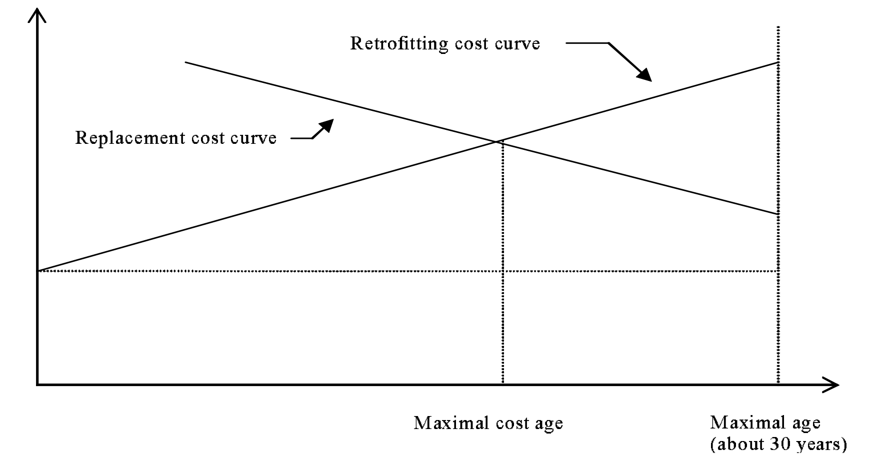
3.4 A method to analyse cost-effectiveness based on indirect cost-indicators

In order to carry out the analysis variables need to be identified which affect the costs of abatement, on the one hand, and which allow to say something about the cost-effectiveness both of the Directive for the countries studied and of the implementation paths chosen, on the other hand. As pointed out just above, the two variables according to which the municipal waste incineration Directives were differentiated -a plant's size in terms of waste incineration capacity and its age- are linked to abatement cost and as such used here as indirect indicators.[45]

The existence of economies of scale in pollution abatement means that abatement cost functions are non-linear with respect to plant and abatement capacity. As a simplified proxy one can assume that the larger the plant, the lower are the marginal and average abatement costs. ***Plant incineration capacity*** is therefore used as a first indirect indicator.

The relationship between the plant's age and costs is more complex. The municipal waste incinerators subject to the Directive's environmental requirements had two broad alternatives for compliance: operators could invest in abatement equipment to retrofit the plant to the regulation's requirements or replace an 'existing' incinerator, thus closing down the old plant. In the latter case, the replaced plant would become subject to the stricter standards for new plant. Depending on a plant's age and thus its position within the (average) life span of an incinerator, one or the other alternative will be advantageous in terms of costs. According to these compliance alternatives two cost curves have to be considered (cf. Figure 5.1): a retrofitting cost curve and a replacement cost curve.

Figure 5. 1: Abatement cost and plant age



The retrofitting cost curve indicates that compliance costs increase over time, which is due to the decreasing remaining life span of the plant, implying a decreasing period over which costs for the abatement equipment can be amortised. The replacement cost curve consists of two components. Firstly, the costs entailed by plant closure, which decrease with the plant's age until they are zero at the end of a plant's life time when the plant has to be closed down for economic reasons anyway. The second cost component consists of the costs for the abatement equipment of the new plant. The installation of abatement equipment in a new plant is sometimes less costly than equipping an old plant with abatement technology which fulfils similar standards because in the former case the abatement equipment can be directly planned into the plant. However, given that requirements for new plant are generally stricter, the costs for the technology itself are generally higher. This effect, here, is assumed to outweigh the former. An age of maximal compliance costs can be identified, located in the intersection of the replacement and the retrofitting cost curve, which, according to experts, is about 15 years for MWIs (while the normal life span of an incinerator is about 30 years). From this it is directly obvious that a plant's compliance costs will be the higher, the closer its age comes to the maximum cost

[45] The indicators applied were developed in the European project IMPOL in the course of which the municipal waste incineration case studies were carried out.

age. The second indirect indicator for assessing abatement costs is therefore the ***deviation of a plant's age from the maximum cost age***.

With the help of these two indicators, qualitative estimates not only about compliance costs of individual plant but also about overall compliance costs, entailed by the Directive's requirements in a Member State, can be made. The smaller the average age deviation (of all plants) from the maximum cost age, and the lower the average capacity of the overall plant park, the higher are **a *country's aggregate compliance cost***. National plant park characteristics, however, not only determine aggregate compliance costs, but also the ***potential for cost savings*** achievable through a cost-effective allocation of abatement efforts. Following from what was discussed above, the more heterogeneous costs are across plants, the more differentiated should be the abatement effort. Using the same base of information -data about plant age and capacity- the potential for cost savings is assessed with the help of two further indirect indicators measuring the cost heterogeneity within each country. These indicators are the value for the ***plant capacity dispersion*** and the ***standard deviation value of plant age***.

3.5 Cost-effectiveness of the Directive in relation to the heterogeneity of compliance costs

3.5.1 Heterogeneity of aggregate compliance costs across countries

Tables 5.8 to 5.11 present the indicator results describing the cost heterogeneity within and across the four Member States as far as it depends on plant capacity and age. Table 5.8 and Table 5.10 refer to large plants, Table 5.9 and Table 5.11 to small incinerators. Note that the last two rows of each table rank the four countries according to the cost characteristics of aggregate costs and cost reduction potential.

Table 5. 8: Cost heterogeneity according to plant size of large MWIs

Large municipal waste incinerators - Indicator: size				
	France	**UK**	**Germany**	**NL (1)**
Min	6,8	7	18	6
Max	100	55	105	227
standard deviation	17,4	10,2	28	59
capacity dispersion (2)	0,87	0,57	0,55	1,09
Mean	20	18	51	54
ranking of countries on a scale from 1 to 4				
Highest aggregate costs	2	1	3	4
Highest cost reduction potential	2	3	4	1

(1) Capacity in t/h calculated from reported capacity in t/year. Conversion factor used: 25000 t/year ≈ 6 t/h. – (2) capacity dispersion = standard deviation/average capacity. - Source: own calculations

The last line in the first part of Table 5.8 shows that the average ***incinerator capacity*** in France and the UK was much lower than in Germany and the Netherlands. In this table, the average capacity only of large incinerators is taken into account. France is the only country that had a high number of small incinerators when the Directive was adopted. Taking these into account as well shows that the cost disadvantage for France outweighs that of the UK (cf. Table 5.9). The Netherlands and Germany did not operate any small incinerators.

Table 5. 9: Cost heterogeneity according to plant size of small MWIs

Small municipal waste incinerators – Indicator: size				
	France (1)	**UK**	**Germany**	**NL**
Min	0,5	3	-	-
Max	6	6	-	-
standard deviation	1,4	1,1	-	-
Capacity dispersion	0,6	0,3	-	-
Mean	2	4	-	-
ranking of countries on a scale from 1 to 4				
Highest aggregate costs	1	2	-	-
Highest cost reduction potential	1	2	-	-

(1) The sample of French small plants used here is not complete (cf. Annex 6.B). Excluded were all those plants for which no information on either the opening year or the capacity was available. Furthermore, for 19 plants the capacity was reported as <1 t/h, <2 t/h and <3 t/h. In order to make the above calculation possible, the reported capacity was transformed into 0.8 t/h, 1.8 t/h and 2.8 t/h respectively. The calculation therefore presents only an approximation. Given that 194 small plants are covered under the sample, the calculation should however give a more or less appropriate view of the situation of the French plant park in the early 1990s in comparison to the UK plant park. - Source: own calculations.

However, to compensate for this disadvantage, the Directive differentiated its requirements according to plant size and defined less stringent requirements for smaller incinerators. For consistency, therefore, the further analysis is restricted to large incinerators. Doing this, the UK is the most disadvantaged country, directly followed by France, whereas Germany and even more the Netherlands, with more than twice the average capacity of the UK and France, were subject to much lower aggregate compliance costs, as far as meeting the Directive's abatement requirements is concerned.

For ***plant age*** Table 5.10 shows that France is in the worst position, with an average age of plants which was closest to the maximum cost age by the time the Directive's compliance requirements had to be met. In 1996 (the deadline for compliance of large incinerators), the average age of large incinerators was 18 years in France, whereas it was 21 years in Germany, 22 in the UK and 24 in the Netherlands. For the latter three countries, the deadline was therefore more in line with investment cycles, being closer to the incinerators' end of lifetime.

Table 5. 10: Cost heterogeneity according to plant age of large MWIs

Large municipal waste incinerators - Indicator: age				
	France	**UK**	**Germany (1)**	**NL (2)**
Min	1965	1968	1965	1963
Max	1990	1981	1987	1986
standard deviation	7,2	3,2	7,4	5,9
Mean	1978	1974	1975	1972
Average age in 1996	18	22	21	24
ranking of countries on a scale from 1 to 4				
Highest aggregate costs	1	3	2	4
Highest cost reduction potential	2	4	1	3

(1) For Germany, when available information suggests that a plant had been replaced before the starting point of the implementation of the EU Directive, this more recent opening year was used for the calculation. The same was done in the French plant data base. (2) Capacity in t/h calculated from reported capacity in t/year. Conversion factor used: 25000 t/year ≈ 6 t/h. - Source: own calculations

Looking at small incinerators (Table 5.11), we find that the UK plants' average age came even closer to the end of an incinerator's normal life span. It would have been 27 years by 2000, the final compliance deadline for small municipal waste incinerators, while the respective average age of 19 years for French plants was much closer to the maximum cost age.

Table 5. 11 Cost heterogeneity according to plant age of small MWIs

Small municipal waste incinerators - Indicator: age				
	France (1)	**UK**	**Germany**	**NL**
Min	1967	1970	-	-
Max	1990	1975	-	-
standard deviation	6,1	1,8	-	-
Mean	1981	1973	-	-
average age in 2000	19	27	-	-
ranking of countries on a scale from 1 to 4				
highest aggregate costs	1	2	-	-
highest cost reduction potential	1	2	-	-

(1) The sample of French small plants used here is not complete. Excluded were all those plants for which no information on either the opening year or the capacity was available. 194 small plants are included in the calculation. - Source: own calculations

3.5.2 Cost heterogeneity within the countries

The cost heterogeneity within countries determines the potential for cost savings. The abatement cost is specific to every plant. Generally, the higher the cost heterogeneity, the higher the potential for cost savings owing to a cost-effective allocation of pollution abatement effort between plants. Looked at it from the other side this also means that the higher the cost heterogeneity, the more detrimental are uniform abatement standards if implemented as prescribed.

The indicators capacity dispersion and standard deviation of age in the four tables above indicate that this was indeed an important issue, as the degree of cost heterogeneity differs significantly between the countries. Only taking into account large incinerators, the potential for cost savings owing to differentiated pollution abatement effort was the highest in the Netherlands and in France as far as capacity is concerned (cf. Table 5.8). With respect to age, the need for time flexibility in the UK was clearly less crucial than in the other three countries (cf. Table 5.10). For small plants, the cost saving potential was quite restricted with respect to both capacity and age of plants in the UK, while in France the opposite was the case (cf. Table 5.9 and Table 5.11).

The found heterogeneity suggests that the detrimental impacts on cost-effectiveness of the Directive's uniform standards -for all plants larger than 6 t/h- differ between the four countries studied, being the lower the less heterogeneous national plant parks are.

4 Conclusions

The aim of this chapter was to assess the Directive's efficiency with respect to its contents and, in particular, to come to some qualitative conclusions about the relative cost-effectiveness of the policy's requirements.

The analysis has shown that owing to the heterogeneity of country contexts and the existence of economies of scale in pollution abatement, the Directive implied quite different relative compliance costs for the four Member States. France, especially, but also the UK, were subject to relatively higher aggregate compliance costs than Germany and the Netherlands. What is more, the Directive's standard differentiation was insufficient to equalise marginal costs across sources and was not similarly cost-effective for all countries. An analysis of the cost-saving potential, reflected in the cost heterogeneity within countries, showed that, as far as capacity is concerned, the limited differentiation of standards was more detrimental for the Netherlands and France than it was for the UK and Germany.[46] This finding constitutes the basis for assessing, in the following chapter, whether and in what way the countries have applied local discretion

[46] For plant age the picture is different but, bearing in mind that plants may have been modernised since the year of construction, our data on the indicator 'capacity' are more reliable than the data on the indicator 'age'.

during the implementation process, thus altering the cost-effectiveness of the actual implementation outcomes relative to the initial policy objectives.

Overall, the finding of the limited cost-effectiveness of the Directive supports an assumption made earlier: the two approaches for studying the efficiency of a policy, focussing either on the efficient allocation of tasks or on the policy's contents, are related. The efficiency of the level on which the policy decision was taken is the reason for the efficiency properties of the policy's contents.

Annex 5.A Economies of scale in abatement technology for municipal waste incinerators

The following two tables show for a range of capacity classes above 6 t/h data for two control options which were presented in Annex 4.D. One is capable of meeting the 1989 Directive's requirements (option A), the other is capable of meeting the requirements of the draft incineration directive as it was discussed in the early 1990s (option C).

1) Control option A (capable of meeting the 1989 Directive's requirements)

Table 5.A. 1: Costs of control option A

Plant capacity	Waste throughput	Total average flue gas treatment cost	Total average annual operating and maintenance cost	Total average flue gas treatment cost	Total average annual operating and maintenance cost
Kt(cap)/y	1000 t/y	1000 ECU	1000 ECU	ECU/t(cap)/y	ECU/t(cap)/y
25	20	2550	258	102,0	10,3
50	40	3020	358	60,4	7,2
100	80	4340	649	43,4	6,5
150	120	5310	851	35,4	5,7
200	60	6230	1050	31,2	5,3
400	320	9440	1829	23,6	4,6
600	480	12810	2645	21,4	4,4
800	640	15540	3435	19,4	4,3

Source: European Commission, 1997

2) Control option C (capable of meeting the draft Directive's requirements)

Table 5.A. 2 Costs of control option C

Plant capacity	Waste throughput	Total average flue gas treatment cost	Total average annual operating and maintenance cost	Total average flue gas treatment cost	Total average annual operating and maintenance cost
Kt(cap)/y	1000 t/y	1000 ECU	1000 ECU	ECU/t(cap)/y	ECU/t(cap)/y
25	20	3490	346	139,6	13,8
50	40	4140	489	82,8	9,8
100	80	5950	885	59,5	8,9
150	120	7270	1171	48,5	7,8
200	60	8530	1455	42,7	7,3
400	320	12920	2563	32,3	6,4
600	480	17540	3718	29,2	6,2
800	640	21270	4836	26,6	6,0

Source: European Commission, 1997

Assuming a discounting of capital costs over 10 years (Table 5.A.3) and 20 years (Table 5.A.4) results in the following total annual costs for the two abatement options:

Table 5.A. 3: Total annual capital and operating and maintenance costs, assuming the abatement equipment's life span is 10 years

Plant capacity	Total FGT costs per annum (10 years) and total annual O&M costs	
Kt(cap)/y	A	C
25	20,52	27,80
50	13,20	18,06
100	10,83	14,80
150	9,21	12,65
200	8,37	11,54
400	6,93	9,64
600	6,54	9,12
800	6,24	8,70

Source: Based on European Commission, 1997; own estimations

The shaded areas comparing control options A and C in Table 5.A.3 (Table 5.A.4) indicate that a plant with a capacity of 600 kt/y (800 kt/y) could comply with the draft directive's requirements (control option C) at average annual compliance costs comparable to those of a plant with a capacity of 150 kt/y which only meets the weaker 1989 Directive's requirements (control option A).

Table 5.A. 4: Total annual capital and operating and maintenance costs, assuming the abatement equipment's life span is 20 years

Plant capacity	total FGT costs per annum (20 years) and total annual O&M costs	
Kt(cap)/y	A	C
25	15,42	20,82
50	10,18	13,92
100	8,66	11,83
150	7,44	10,23
200	6,81	9,41
400	5,75	8,02
600	5,48	7,66
800	5,27	7,37

Source: Based on European Commission, 1997; own estimations

The Directive did not introduce any standard differentiation for all those plants whose capacity is larger than 6 t/h, although the exact size of plants within this capacity group, in the early 1990s, differed largely both within and across Member States.

Chapter 6 An Empirical Evaluation of Environmental Compliance and Cost-Effectiveness - the Implementation of Directive 89/429/EEC

1 Introduction

This chapter together with the following one undertakes an *ex post* evaluation of the implementation paths chosen and outcomes obtained by France, Germany the Netherlands and the United Kingdom[47] in the implementation of the Council Directive of 21 June 1989 on the reduction of air pollution from existing municipal waste-incineration plants (89/429/EEC). The goal is to see whether implementation has (partly) reduced the inefficiencies of the Directive. The criteria used in this analysis to evaluate the implementation processes *ex post* are goal attainment and cost-effectiveness.

Goal attainment relates to the achievement of objectives defined by a policy. The present chapter analyses goal attainment by setting the implementation outcomes in relation to the initial policy objectives. Goal attainment *a priori* is relatively straightforward to apply as the Directive represents a 'classical' piece of regulation defining emission limits for a variety of airborne pollutants. Comprehensive emission data, however, were not available for all four countries, so in practice more indirect indicators had to be applied. Next to emission limit values the Directive also formulates administrative requirements. The countries' compliance with these is also studied. With respect to *cost-effectiveness*, defined in the previous chapter, the analysis focuses on the properties of the actual implementation outcomes, i.e. the cost-effectiveness of the implementation paths relative to the cost-effectiveness of the Directive's objectives.

The chapter is structured as follows. Section 2 presents the Directive's transposition on a national level in the 4 Member States and evaluates the implementation outcomes in terms of goal attainment. Sub-sections 2.1 to 2.4 deal with the four countries respectively. Section 3 evaluates these outcomes in terms of cost-effectiveness, and section 4 concludes.

2 Compliance differs between the four countries

As outlined in chapter 2, the implementation of EU environmental policy generally comprises various objectives and by this addresses also different actors. The 1989 municipal waste incineration Directive not only defines emission limit values and combustion conditions to be met by the incinerators. It also demands emissions to be measured, abatement and monitoring equipment to be authorised, and compliance with the conditions imposed by the Directive to be monitored by competent authorities. And, finally, Member States are required to bring into force the laws, regulations and administrative provisions necessary to comply with the Directive and to communicate these to the Commission.

In order to take account of this variety of what compliance or goal attainment includes we distinguish three categories of compliance:

- ***administrative compliance***, referring to the process of legally transposing the Directive into national law,
- ***regulatory or initial compliance***, referring to the process of operators demonstrating the technological capabilities necessary to receive an operation licence in accordance with requirements specified in the EU and national law,

[47] The empirical parts of this chapter and the following are to a large extent based on case studies on the implementation of Directive 89/429/EEC in the four countries published as: Bültmann and Wätzold, 2000; Eames, 2000; Lulofs, 2000 and 2001; and Schucht, 2000, which were prepared in the context of the IMPOL project, as well as on Glachant (2001) and Schucht et al. (2001).

- ***operational or continuing compliance***, referring to the operational performance of a plant and its maintenance, where monitoring and enforcement are to ensure that the requirements of its licence are fulfilled or enforced where breaches are found to occur.

As will be demonstrated throughout the following sections, these categories are partly independent of each other, i.e. compliance with these three categories can differ within a country. In particular, on time administrative compliance does not appear to be an indispensable prerequisite for regulatory and operational compliance, nor does it ensure the latter.

As was previously shown, the countries studied have not developed their domestic environmental regulations in parallel. Instead, they show considerable differences as to the speed with which regulation directed at atmospheric emissions from municipal waste incinerators (MWI) has been developed. This has an impact on the requirements actually implemented on a national level in parallel with the EU Directive's implementation. It also has an impact on the abatement investment necessary in order to comply with the Directive. Not only did the countries' domestic regulations in force at the time the EU Directive was adopted vary, also the requirements of those regulations officially serving as the Directive's transposition vary across the four countries. In order to take account of this variety, it is in order to shortly discuss in how far the national regulations serving as transposition differed from the EU Directive's requirements.

2.1 The Directive's implementation in France

2.1.1 Transposition of the Directive's minimal requirements into French regulation

As far as pre-existing regulation is concerned, chapter 5 concluded that at the time the EU Directive was adopted and had to be transposed into national law, France, in general, showed some backwardness with respect to the incinerators' equipment with clean air technology. This is at least true as compared to other European countries such as the Netherlands and Germany where stricter standards for existing and new incinerators had existed before 1990. What was the French approach in the transposition of the Directive?

France transposed the EU Directive into national law by the ministerial order (arrêté) of 25th January 1991[48], specifying atmospheric emission limits for both new and existing municipal waste incinerators. With this, France showed only a slight delay in the transposition of the Directive into national law. The ministerial order was published not even a month after the date prescribed by the Directive. As far as atmospheric emission standards and deadlines are concerned, France transposed the EU Directive's requirements as they were (cf. table 4.2; and table 6.A.1 in annex 6.A). It did this by keeping to the minimum requirements demanded by the Directive and did therefore not follow the Directive's suggestion to define additional emission limit values for further pollutants, in particular, for dioxins and furans.[49] Unlike the Directive, however, the French ministerial order takes an integrated approach, defining further requirements with respect to solid residues of waste incineration (such as slag) and residues of the flue gas treatment, in particular for their disposal (landfill), transport and elimination. The ministerial order furthermore defines requirements and standards for the prevention of water pollution and noise. Its implementation was therefore likely to cause additional costs, not directly resulting from the EU Directive.

48 Arrêté du 25 janvier relatif aux installations d'incinération de résidus urbains.

49 Although France has not specified dioxin emission limits in the Directive's transposition, some requirements concerning this pollutant were developed during the 1990s. From municipal waste incinerators licensed after 24 February 1997, France demands to meet the dioxin emission standard of 0.1 Ng/m3, by this anticipating the future European emission standard (circular of 24th February 1997). As this concerns 'new' plants only, it does not affect the investigation undertaken in this case study. However, measurement of dioxin emissions from large 'existing' incinerators was required by a circular of 30th May 1997.

Also the French monitoring requirements are overall in line with the EU Directive's, although not always identical. In some cases (total dust and CO), the French law requires additional weight controls and periodical spot checks where the Directive requires only continuous measurement. But in some cases (heavy metals, HF, SO_2 and organic compounds) where the Directive requires periodical measurement, the *arrêté* only asks for spot checks. With respect to the reference time of combustion gases, the *arrêté* -unlike the Directive- does not refer to adaptations of plant and neither does it specify a deadline for the measurement campaign. The monitoring requirements specified in the 1991 ministerial order are presented in annex 6.A (cf. table 6.A.2 with table 4.A.4 in annex 4.A).

In line with the Directive's further requirements, the exact periodicity of periodical measurements and the requirements of communicating the results to the competent authority were to be specified in the plants' individual operation permits (art. 13 and 28, arrête 1991). However, the ministerial order transposing the Directive makes no reference to the Directive's requirement of informing the public about measurement results.

2.1.2 Delayed goal attainment in France

In the early 1990s, France operated a considerable number of municipal waste incinerators, approximately 320 altogether, many of which were of very small capacity (Table 6.1).[50] There were 66 large incinerators, i.e. plants of a capacity of more than 6 t/h, and supposedly 254 plants of a lower capacity[51].

Table 6. 1: Compliance choice of French MWIs by 2003

Capacity group	Number of plants	Upgraded	Downsized	Closed	Replaced	Closed and/or replaced	Initially compliant	NV
all plants	320	75	3	206	17	223	7	12
< 1 t/h	45	3	-	37	0	37	1	4
1 to 3 t/h	104	11	-	84	2	86	3	4
3 to 6 t/h	47	16	-	19	7	26	1	4
> 6 t/h	66	45	3	9	7	16	2	-
NV	58	-	-	57	1	58	-	-

NV: no value available. - Source: MATE, 1998, 2001a, 2001b and 2002; MEDD, 2002 and 2003; ITOM, 1995; ADEME, 1998

Table 6.1 furthermore indicates that a majority of plants, mostly small plants, was closed or replaced. It were predominantly the large incinerators that were upgraded to comply with the Directive's requirements. It should be noted that there are 58 plants for which hardly any information is available. It can, however, be assumed that these were of very small capacity and that they were closed before the final compliance deadline for small plants. Also within the group of plants for which the capacity size is known, there are 12 plants whose eventual compliance path is not known (Table 6.1). Also for these it can be assumed that they were closed. Detailed information on the French plant park is given in annex 6.B.

50 The data presented in the following two tables have been updated until 2003. For the remaining countries the analysis stops in 1999 because in these countries almost all plants were compliant by that time.

51 This estimation of the plant number (and of the compliance decisions) is based on a comparison of 8 municipal waste incinerator inventories. Four were provided by the previous Ministry of the Environment (MATE, Ministère de l'Aménagement du Territore et de l'Environnement) and represent the plants' state of compliance in 1998 (for large incinerators), in 2000 and 2001 (for small incinerators) and in 2002 for all incinerators. Two inventories were provided by the new Environmental Ministry (MEDD, Ministère de l'Ecologie et du Développement Durable). The first of these was published in 2002 and presents data on all incinerators which were non-compliant in that year, the second dates of 2003 and covers all incinerators in operation at that time. Another inventory was provided by the ADEME, presenting the MWI park in 1998 and, finally, the 1995 ITOM inventory presents the park in 1993. All these inventories included different, limited information about the plant park. For a more detailed description of these inventories and the hypotheses made to merge them, cf. annex 6.C. For a variety of plants information was lacking altogether, this is indicated in Table 6.1 and Table 6.2 with NV.

Table 6.2 gives information about the plants' timing of compliance, i.e. on the number of plants complying on time or late, and for the late complying plants also on the compliance delay in years. Note that there are again a number of plants for which no data are available, in this case information is lacking on the exact timing of compliance. As far as large MWIs (> 6 t/h) are concerned, the deadline set by the Directive was 1996 and the compliance delay is assessed as difference to this year. For smaller plants there were two successive deadlines, one for 1995, setting intermediate requirements, and a final deadline for 2000. It is impossible to assess the compliance delay with respect to the 1995 deadline, due to the unavailability of a complete plant inventory presenting the compliance status or the status of abatement equipment in 1995. The data for small plants in Table 6.2 refer to the 2000 deadline.

Table 6. 2 Timing of Compliance Behaviour by French MW incinerators

		Timing of compliance overall				**Compliance delay in years to respective deadline (1)**						
Capacity group	number of plants	Early	on time	early or on time	late	1	2	3	4	5	6	NV
all plants	320	76	83	159	83	21	41	12	8	0	1	78
< 1 t/h	45	11	15	26	13	4	6	3	-	-	-	6
1 to 3 t/h	104	37	35	72	25	7	17	0	1	-	-	7
3 to 6 t/h	47	11	22	33	10	2	4	3	1	-	-	4
> 6 t/h	66	17	11	28	35	8	14	6	6	0	1	3
NV	58	-	-	Probably all	-	-	-	-	-	-	-	58

(1) Official deadline for compliance of large plants was 1 December 1996, for small plants 1 December 2000. NV: no value available. - Source: MATE, 1998, 2001a, 2001b and 2002; MEDD, 2002 and 2003; ITOM, 1995; ADEME, 1998

We find that a large number of plants complied on time, but also that the number of plant closures of small plants included in this is high. Moreover, the table shows that more than 50% (35 out of 66) of the large incinerators complied late, i.e. after the 1996 deadline fixed by the Directive. Compliance was primarily achieved through upgrading, although some plants were closed, replaced, or reduced in size[52]. All but one of the remaining large incinerators complied by end-2000. Altogether, compliance of large plants was reached with up to 6 years' delay.

A study of the Environment Ministry undertaken in 1998 showed that about 3/4 of the approximately 190 smaller ('existing' and 'new') plants remaining in operation at that time did not comply with the Directive's 1995 requirements. By 2000, approximately 90 small plants had been closed or were expected to do so. The majority of plants still operating at this time had not yet completed or even started retrofitting measures. In more detail, 27 small plants were expected to comply through retrofit, in approximately 10 cases upgrading measures were under way, and about 55 plants were still non-compliant with respect to the Directive's 2000 requirements.[53] However, the majority of waste was already burnt in large plants around big cities, and therefore in compliant plants[54]. As indicated by the previous table, by 2003, 207 of the initially 254 small plants had been closed, 10 of which were replaced. All in all, 48 small plants complied late with respect to the 2000 deadline for compliance, and for 17 plants the timing of compliance is not known. The current environmental Ministry (Ministry for Ecology and Sustainable

52 Thus becoming subject to less strict environmental standards and extended deadlines for compliance.

53 Source: http://www.environnement.gouv.fr/dossiers/dechets/incineration/010122-incinerateurs-petits.htm; (18/5/2001).

54 The following can serve as a proxy: A 1993 plant inventory including all plants with a capacity above 3 t/h showed that these plants represented only about 1/3 of all French municipal waste incineration plants which however treated about 92% of total waste incinerated (TSM N° 9, 1994).

Development, MEDD) assumes that by the end of 2003 all 'existing' MWIs were compliant. Compliance of small incinerators, therefore, was reached with 3 years' delay.

For France, operational compliance in terms of meeting the emission limit values at any time as defined by the Directive is impossible to assess. There are no comprehensive emission inventories including all incinerators and all pollutants regulated and which give information on breaches of emission limits. It consequently remains unclear whether all municipal waste incinerators continuously comply with the Directive's requirements. In fact, interviews carried out with the French Agency of the Environment and Energy Control (ADEME) at the end of the 1990s indicated that some large incinerators which had previously been upgraded to meet the requirements of the French ministerial order of 1986 (cf. chapter 5) were considered as complying also with the EU Directive. No information is available on whether these plants were subsequently forced to make changes necessary to continuously meet the Directive's requirements.

2.2 The Directive's implementation in Germany

2.2.1 Domestic regulation enables early transposition in Germany

As shown in chapter 5, amongst the countries studied here, Germany belongs to the group of environmental 'leaders' in terms of strict pre-existing legislation on emissions from waste incinerators. As was already mentioned, the provisions of the EU Directive were in principle covered by the TA Luft 1986 (Technical Instructions on Air Quality Control), the pre-existing national legislation. But the European Court of Justice does not recognise a transposition of European Directives by means of technical instructions and requires instead a transposition by either act or ordinance. Technical instructions have the status of administrative guidelines, binding within the public administration but not directly affecting citizens and courts. The latter may always examine the appropriateness of such instructions before they make them the basis of their decisions. Consequently, emission limits of the TA Luft were not directly binding on plants. Plants only had to comply with these limits after they had been incorporated into individual plant authorisations. Hence, in order to have the TA Luft accepted as transposition of the EU Directive, the German government would in any case have had to translate the Directive's provisions into an act or ordinance.

This, however, was not necessary any more, as by the time the EU Directive was adopted Germany was already about to impose significantly stricter and more far reaching domestic limits, for both new and existing municipal waste incinerators. These standards were introduced under the German Ordinance on Incineration Plants for Waste and Similar Combustible Substances (Verordnung über Verbrennungsanlagen für Abfälle und ähnliche brennbare Stoffe – 17. BImSchV) of 23 November 1990, issued under the Federal Imissions Control Act (BImSchG - Bundesimmissionsschutzgesetz). With this, Germany is the only country amongst the four studied that has managed to integrate the Directive's requirements on time. Table 6.A.3 (cf. annex 6.A) allows for a direct comparison of the German emission limit requirements with those of the EU Directive.

In contrast to the Directive, Germany does not differentiate emission limits according to capacity. All municipal waste incinerators have to comply with the same emission limits as new plants by 1 December 1996. Neither did Germany allow for any transitional arrangements for smaller plants. Furthermore, the ordinance does not distinguish between existing and new plants. And last but not least, German emission limit values are much stricter than the EU requirements and the scope of pollutants covered is larger. In particular, Germany defined emission limits for dioxins and furans. Except for the deadline for compliance, which is similar to the EU Directive's deadline for large plant, the German regulation is thus in every respect stricter than the EU Directive.

Its implementation was even further tightened in North Rhine-Westphalia (NRW), the German *Land* which accounts for the highest incineration capacity in Germany and also for the second highest number of municipal waste incineration plants, and which is studied in greater detail here.[55] In NRW, changes to the implementation time-schedule were introduced by a negotiated agreement between the NRW government and the waste incineration sector, agreed upon in February 1990, i.e. several months before the 17. BImSchV came into force. This voluntary agreement, the ‘Emission Reduction Plan for Dioxins from Waste Incineration Plants’ (Emissionsminderungsplan für Dioxine aus Abfallverbrennungsanlagen - EMDA), specified a dioxin and furan emission limit value of 0.1 ng TE/m3 and a corresponding retrofitting deadline for 1 December 1995 for incinerators for municipal, hazardous and industrial waste. It was based on the TA Luft 1986 requirement to reduce dioxin emissions as far as possible. Compared to the 1990 Ordinance, the EMDA therefore imposed a stricter compliance deadline. Moreover, it became soon clear that the abatement equipment for dioxin emissions could not sensibly be separated from that for the other pollutants regulated under the Ordinance. The EMDA's retrofitting deadline, therefore, indirectly applied to the entire abatement technology necessary to comply with the 17. BImSchV.

Turning to monitoring requirements (cf. table 6.A.4 in annex 6.A with table 4.A.4 in annex 4.A) specified under the 1990 German Ordinance, one finds that -with the exception of heavy metals, dioxins and furans, which have to be measured at least on 3 days per year by recognised measurement institutions- all emissions and operating parameters have to be continuously measured (article 11 (1) 17. BImSchV). Furthermore, except for CO, the German law requires emission limits to be met on a daily or half-hourly average where the Directive only requires them to be met on a moving 7-day average. The German ordinance is thus stricter than the Directive also with respect to monitoring requirements.

Finally, in line with the Directive’s requirements, the German Ordinance specifies the periodicity in which the results of all measurements have to be recorded, analysed and reported to the competent authority. It also opens the possibility for competent authorities to prescribe the telemetric transfer of data. Measurement equipment is required to match the best available technology. In line with the Directive's requirements, the data must be made publicly available.

2.2.2 High goal attainment level in Germany/North Rhine-Westphalia

Germany operated 48 ‘existing’ incinerators, almost all of which were equipped with abatement technology adequate to meet the national standards. Apart from a small number of plants that needed a few extra months, all German plants were retrofitted by 1 December 1996, the deadline set in the German ordinance. An in-detail study was carried out for North Rhine-Westphalia (Table 6.3).

In NRW, 12 out of 13 ‘existing’ incinerators were upgraded and only one smaller plant was closed down, and this not directly due to environmental regulation[56]. All NRW plants met the retrofitting deadline of 1 December 1996 or were upgraded earlier. 11 incinerators participating in the EMDA managed to meet the EMDA retrofitting deadline of 1 December 1995 for the installation of devices to abate dioxins and furans, and 7 plants had their entire pollution abatement equipment in place by that time.

[55] In the early 1990s Germany had 48 existing incinerators, 14 of which were situated in North Rhine-Westphalia. Most German incinerators are large plants, only very few have a nominal capacity below 6 t/h (3 out of 53 new and existing incinerators in 1994) (Anonymous, 1995; Schmidt-Tegge, 1993). In 1999, NRW had 16 municipal waste incineration plants (accounting for 26% of the total number of plants in Germany and for 36% of the total incineration capacity). The other German *Land* with a high number of municipal waste incinerators is Bavaria, running 17 plants in 1999 (which made up for 28% of the total number of plants and for 20% of the total incineration capacity).

[56] This plant ceased operation in July 1991 after an operation defect. Generally, in Germany only those plants were shut down that were not economical in operation, either because of their age or insufficient size. Shut downs were therefore not directly caused by the 17. BImSchV, although the Ordinance may in some cases have speeded up closure decisions.

Table 6. 3: Compliance path of existing municipal waste incinerators in NRW

Plant	Initial opening year and modernisation of combustion units	Capacity in tonnes/hour	Energy Recovery	Compliance through … in …
Aachen	NV	NV	NV	shut down in 1991
Bielefeld	1981 1995/96 modernisation	3x16	yes	upgraded in 1995/1996, one oven temporarily switched off in 1996 until its connection to abatement facility
Bonn	1991	3x11	yes	New plant in operation since 1992
Düsseldorf	1965 1996 modernisation	6x12.5	yes	upgraded in 1993, modernised in 1996
Essen	1960 1987 new (replaced) 1993 modernisation started	4x26.3	yes	upgraded in 1996
Hagen	1966 1988/89 modernisation	3x6	yes	upgraded in 1996
Hamm	1985 1994 modernisation	4x9.4	yes	upgraded in 1995
Herten	1982 1990 modernisation	2x20	yes	upgraded in 1994
Iserlohn	1970 1993-1996 capacity increase and modernisation	2x8 1x16	yes	upgraded in 1996
Krefeld	1975/76 1994 modernisation	2x12 plus 2x1.2 (sewage sludge) 1x18.4	yes	upgraded in 1995
Leverkusen	1970 1986 enlarged 1996 modernised	2x10 1x12	yes	upgraded in 1996
Oberhausen	1972 1997 modernised	2x22 2x25	yes	upgrade in 1996
Solingen	1969 1993 modernisation	1x12 1x7.6	yes	upgrade in 1995
Wuppertal	1976 1995 enlarged	4x15 1x10	yes	upgrade in 1995

NV: no value available. - Source: EMDA, 1996; UBA, 08/1999

Measurements of emissions in German waste incinerators showed that emission limits, generally, have only been exceeded for a few minutes, due to emission peaks, and that such problems could be adjusted by means of process optimisation. Only shortly after the instalment of the abatement equipment were emission limits exceeded more often and for longer periods. In such cases, the German Ordinance limits the time operation may continue to a maximum of 8 successive hours and a total of 96 hours per year. Operators were almost always able to meet this dual time limit. Only one example of a municipal waste incinerator was reported where the operator did not manage to solve the technical problems with its activated carbon filters within this time limit and was subject to enforcement measures, having to reduce waste throughput until the problem was resolved. Overall, therefore, German waste incineration plants rarely exceeded their emission limits and were normally well below these limits. Given that these emission limit values are much stricter than those defined by the Directive, operational compliance in Germany in the sense of the Directive is high. Incineration plants, in general, largely over-comply with the European standards (detailed data for 10 NRW incinerators in 1994 and 1996 is provided in annex 6.D). Goal attainment in terms of compliance with environmental requirements in Germany (NRW) can be judged as high.

2.3 The Directive's implementation in the Netherlands

2.3.1 Late but ambitious transposition into Dutch regulation

The Netherlands represent the second environmental 'leader' country in terms of pre-existing regulation. But although the Netherlands were about to impose significantly stricter and more far reaching domestic emission limits under the 1989 Dutch Guideline Combustion (Richtlijn Verbranden) when the EU Directive was adopted, the Dutch government faced considerable problems with the formally correct incorporation of EU requirements into Dutch law. The Commission did not recognise the Guideline Combustion 1989 as formal transposition, as it left some discretion to the provincial administration. For acceptance by the Commission, the Dutch emission limits were to be issued under an Order in Council (Algemene Maatregel van Bestuur - AMvB), which is comparable with a German ordinance, and not under a guideline. This required an amendment to the Dutch law, necessary to authorise the Environment Minister to issue an AMvB, which was a time consuming process.

This explains why formal transposition of the EU Directive was only reached with one year's delay, when a draft AMvB was pre-published on 3 April 1992. And finally, the 1989 Incineration Guideline was withdrawn on 7 January 1993 and replaced by the Air Pollution from Waste Incineration Decree (Besluit luchtemissies afvalverbranding), constituting the official transposition of the EU Directive. This official transposition of the Directive came therefore with more than two years delay. Emission limit values are the same as in the 1989 Guideline Combustion, however, the deadline for compliance, compared with the 1989 regulation, was extended to 1 January 1995 (cf. table 6.A.5 in annex 6.A).

Contrary to the Directive's requirements, the Dutch regulation does not differentiate emission limits according to plant capacity or age (new or existing plant). All incinerators had to meet the emission limit values by 1 January 1995 and with this almost two years before the date required by the Directive. Furthermore, Dutch emission limit values are stricter and the scope of pollutants covered is larger, including in particular dioxin and furan emission limit values.

Monitoring requirements specified by the Air Pollution from Waste Incineration Decree are largely in line with the Directive's requirements (cf. table 6.A.6 in annex 6.A with table 4.A.4 in annex 4.A). They are stricter with respect to SO_2 and organic compounds, which in the Netherlands have to be continuously monitored. Furthermore, the Dutch regulation requires compliance with hourly averages where the Directive demanded compliance with moving 7-day-averages or daily averages.

2.3.2 High goal attainment level in the Netherlands

In the Netherlands 5 out of 13 'existing' plants were equipped with abatement technology adequate to meet the national standards (Table 6.4). 8 plants were closed, the majority of which was considered too old-fashioned and poorly designed to be retrofitted. The capacity of these plants was replaced, but not necessarily on the same location and not every single plant. In those plants that were upgraded, abatement equipment was retrofitted by 1 January 1995 as required by the stricter Dutch law, except in the case of one plant which was granted an extension until 1 December 1996. Regulatory compliance with the EU Directive was therefore reached, in the majority of cases with almost 2 years' advance.

In 1995, a comprehensive inventory was carried out for the Waste Board of the Dutch Ministry of the Environment, as part of the monitoring of new standards for existing incinerators. Of the 212 controls taken, 206 were within the limits set by the Dutch regulation. While there were thus 6 breaches with Dutch regulation, none exceeded the limits set by the European regulation. Usually, emission levels were much lower than even the strict Dutch limits. What was stated for Germany also holds for the Netherlands: not only regulatory compliance, also operational compliance is high. A comparison of actual emission levels of Dutch incinerators with the European Directive's requirements shows large over-compliance. Detailed data on a selection of Dutch incinerators are presented for 1991 and 1995 in annex 6.E.

Table 6. 4: Compliance path of existing municipal waste incinerators in the Netherlands

Plant	Opening Year	Capacity in tonnes/year (early 1990s)	Energy Recovery (1)	Compliance through … in …
Zaanstadt (AVI)	1976	140,000	NV	closure, 1990
Alkmaar (VVI)	1973	125,000	Yes	closure between 1990 and 1991, re-opening after adjustment and final closure in 1996 - replaced by HVC-Alkmaar
Leiden (Gevulei)	1966	90,000	NV	closure, 1990
Leeuwarden (OLAF)	1973	80,000	NV	closure, 1991
Amsterdam (AVI-Noord)	1969	234,000	NV	closure, early 1990s, replacement by AVI-Amsterdam in 1993
Den Haag (VVI)	1967	330,000	NV	closure, early 1990s
Philips (very small private plant)	NV	26,000	NV	closure, early 1990s
Duiven (Avira)	1975	315,000	Yes	retrofit on time (1995)
Nijmengen (ARN Weurt)	1986	69,000	Yes	retrofit on time (1995)
Roosendahl (HVR)	1974	35,000	Yes	closure on time, replacement in 1996
Rozenburg (AVR-Rijnmond)	1973	945,000	Yes	retrofit on time (1995)
Dordrecht (GEVUDO)	1973	158,000	Yes	retrofit by 1996
Rotterdam (ROTEB)	1963	375,000	Yes	retrofit on time (1995)

(1) The Environmental Management Act of 1993 prescribes energy recovery for all municipal waste incinerators. NV: no value available. - Source: Lulofs, 2000; Eberg 1997; Ministry of Housing, Spatial Planning and the Environment, 1998.

2.4 The Directive's implementation in the United Kingdom (England and Wales)

2.4.1 A late transposition into UK regulation

As pointed out in the previous chapter, the UK presents an extreme case, where MWI plants prior to 1989 were basically not subject to any system of authorisation, detailed emission limits or emission monitoring.[57] It was only in 1989, that incineration processes with a capacity greater than 1 t/h were brought under control of the national inspectorate (Her Majesty's Inspectorate of Pollution – HMIP) under the Health and Safety (Emissions into the Atmosphere) (Amendment) Regulations 1989. Formal transposition of the EU Directive into UK law, however, was only reached in November 1991, with almost a year's delay -and after the Commission had taken formal infringement procedures against the UK in May 1991- by issuing the Municipal Waste Incineration Directions (secondary legislation) under the Environmental Protection Act 1990.

The 1990 Act had introduced a more integrated and centralised approach to the control of industrial pollution in the UK. It established two separate pollution control regimes for (prescribed) industrial processes: Integrated Pollution Control (IPC) and Local Air Pollution Control (LAPC). The allocation of processes between IPC and LACP regimes is dealt with in secondary legislation: the Environmental Protection (Prescribed Processes and Substances) Regulations 1991.

Municipal *waste incinerators < 1 t/h* were classified under LAPC, generally regulating smaller, less complex and less polluting processes. Authorisation requirements for plants < 1 t/h were specified in guidance note PG 5/4 (91), which was revised in 1995 as PG 5/4 (95). Specifications of regulation applying to plant < 1 t/h were in parts stricter than the Directive's requirements for this plant category. They are not presented in the annex as the UK, in the relevant period, did not operate any municipal waste incinerator of a capacity below 1 t/h.

57 Note that the analysis for the United Kingdom focuses on England and Wales, as Scotland has its own distinct legal and administrative system.

Municipal waste incinerators with a capacity of *greater than 1 t/h* are regulated under IPC, which generally regulates more complex and polluting industrial processes. This requires the authorisation of such plants, and that all such plants comply with 'Best Available Technology Not Entailing Excessive Costs' (as required by the Directive) and, where releases to more than one environmental medium may occur, with BPEO. BPEO indicates Best Practicable Environmental Option and aims at ensuring that damage to the environment as a whole is minimised[58]. Detailed advice on what constitutes BATNEEC for a particular process is set out in process guidance notes, issued by the Chief Inspector of HMIP. Standards relevant to MWI were defined under 'Process Guidance Note IPR 5/3, Waste Disposal & Recycling Municipal waste Incineration', published on 1 June 1992. Requirements were stricter than the Directive with respect to both emission limits and range of pollutants covered (cf. table 6.A.7, Annex 6.A). Existing incinerators had to comply with the IPR 5/3 standards by 1 December 1996, in line with the Directive's compliance deadline.

When further comparing the UK regulation with the Directive one finds that the UK does not distinguish between capacity categories for incinerators above 1 t/h. Emission limit values are partly stricter than required by the Directive (eg. for dust, HF and heavy metals for plants with a capacity between 1 and 3 t/h, and for HCL for all plants). Furthermore, just as Germany and the Netherlands, the UK defined additional emission limit values for NO_x and dioxins and furans. These are, however, less strict than in the other two countries. In addition to the Directive's requirements, the process guidance note (IPR 5/3), as part of the IPC regime, also suggested limits on the release of prescribed substances to water, and included provisions for setting limits on the release of waste to land.

UK monitoring requirements for emissions to air from MWI plant are in line with the EU Directive in that periodical measurements must be undertaken for heavy metals and hydrogen fluoride, and potentially weaker in that such measurements 'should' only be made for sulphur dioxide, oxides of nitrogen and volatile organic carbon (cf. table 6.A.8, annex 6.A and table 4.A.4 in annex 4.A). The UK's monitoring requirements are in line with the Directive's and even slightly stricter where continuous measurement results must comply to 95% with hourly averages (e.g. dust) instead of moving 7-day-averages as in the case of the Directive.

2.4.2 Reasonable goal attainment level in the United Kingdom

As can be seen in Table 6.5, in 1989, the UK operated 32 existing plants of a capacity above 6 t/h, 5 of a capacity between 3 and 6 t/h, and none of a capacity below 3 t/h. 33 out of the 37 existing municipal waste incinerators were closed, and the waste largely transferred to landfill. The remaining 4 plants were upgraded to ensure compliance with both the Directive and UK BATNEEC standards.

At three of the four plants which were kept in operation, at the Coventry, Nottingham and Edmonton plants, upgrading was completed successfully - although in the case of the Edmonton plant waste throughput was reduced temporarily through the 1996 deadline whilst upgrading was completed. However, work on the fourth plant, the Sheffield incinerator, has suffered considerable delays, with commissioning problems, breaches of emission limits and frequent shutdowns ongoing by the time this study was carried out in 1999/2000. Therefore, all except one plant showed regulatory compliance with the Directive. The plant that has continuously encountered problems was forced to shut down operations until compliance would be reached.

58 For a discussion of differences between the concepts of BATNEEC and BPEO see for example Sorrell (2001).

Table 6. 5: Compliance path of existing municipal waste incinerators in the UK

Plant	Opening Year	Capacity in tonnes/hour	Energy Recovery	Compliance through … in …
Alloa	1975	1x3	No	closed in time for 1996 deadline
Altrincham	1973	2x5	No	closed in time for 1996 deadline
Basingstoke	1969	1x9	No	closed in time for 1996 deadline
Belfast	1973	1x4	No	closed in time for 1996 deadline
Birkenhead, Wirral	1978	2x14	No	closed in time for 1996 deadline
Bolton	1971	1x16	No	closed in time for 1996 deadline
Bristol	1972	2x15	No	closed in time for 1996 deadline
Coventry	1975	3x11	Yes	retrofit in 1996
Derby	1969	2x8	No	closed in time for 1996 deadline
Dudley	1969	2x6	No	closed in time for 1996 deadline
Dundee	1979	2x7	No	closed in time for 1996 deadline
Edmonton, London	1970	5x11	Yes	stayed in operation through 1 December 1996 deadline but with reduced throughput until retrofit
Exeter	1970	1x9.5	No	closed in time for 1996 deadline
Gateshead	1973	2x10	No	closed in time for 1996 deadline
Havant	1974	1x14	No	closed in time for 1996 deadline
Huddersfield	1975	2x6	No	closed in time for 1996 deadline
Jersey	1977	2x5	No	closed in time for 1996 deadline
Lichfield	1970	1x5	No	closed in time for 1996 deadline
Mansfield	1973	1x6	No	closed in time for 1996 deadline
Marchwood, Fawley, Hants	1975	1x9	No	closed in time for 1996 deadline
Middleton	1968	1x8	No	closed in time for 1996 deadline
Nottingham	1973	2x12	Yes	retrofit in 1995
Portsmouth	1972	2x10	No	closed in time for 1996 deadline
Renfrew	1974	2x8	No	closed in time for 1996 deadline
Rochdale	1981	1x8	No	closed in time for 1996 deadline
Scillies	1978	1x7	No	closed in time for 1996 deadline
Sheffield	1976	2x10	Yes	upgrading and substantial rebuilding work caused persistent problems leading to temporary closure of plant
South Shields	1975	2x10	No	closed in time for 1996 deadline
Stoke on Trent	1977	2x11	No	closed in time for 1996 deadline
Sunderland	1972	2x10	No	closed in time for 1996 deadline
Swindon	1974	1x12	No	closed in time for 1996 deadline
Teeside	1973	1x12	No	closed in time for 1996 deadline
Tynemouth	1971	2x16	No	closed in time for 1996 deadline
Tyseley, Birmingham	1978	2x15	No	closed in time for 1996 deadline
Winchester	1972	1x9	No	closed in time for 1996 deadline
Wolverhampton	1973	2x10	No	closed in time for 1996 deadline; replaced
Upton-upon-Severn, Worcester	1972	2x2	No	closed in time for 1996 deadline

Source: House of Lords, 24 January 1989; Eames, 2000

As in the other 4 countries, UK plant operators are required to monitor their emissions - through continuous or regular measurements- and to report any breaches of their authorisation to the regulator. Almost 500 breaches of emissions limits were reported at incinerators (not only municipal waste incinerators, but also including hazardous waste and sewage sludge incinerators) between January 1996 and November 1998. The actual number of breaches may be higher, as this estimate is based on self-monitoring. The majority of these breaches are thought to have occurred at MWI plants. However, most relate to very short hydrogen chloride (HCl) ‘spikes’ of a few seconds’ or minutes’ duration. They are in the main caused by the incineration of large amounts of plastics generating large volumes of acid gas and temporarily overcoming the abatement equipment. Such breaches have occurred at both ‘new’ and ‘existing’ plants and led to enforcement measures (cf. chapter 7).

It can be concluded that initial compliance in the UK was overall high, while operational compliance might be lower than for example in Germany or the Netherlands. But without more information available about short-term breaches in all Member States, a comparison between countries on such a detailed level is not possible.

3 Were the actual implementation paths more cost-effective than the Directive's requirements?

The analysis in the previous chapters gave evidence of inefficiencies in the Directive. This holds both with respect to the allocation of tasks between the European and national levels and with respect to the contents of the Directive. In particular, this policy failed to equalise marginal abatement costs across all sources and therefore to minimise the overall abatement costs. This is the basis to analysing whether and in how far countries have, during the implementation process, decreased the inefficiencies of the Directive. In this context, it is important to recall that countries may, during the implementation process, change or further differentiate the prescribed abatement standards. They can do so either by anticipating future requirements, thus leading to more or earlier abatement than required by the concerned policy. But they can also do so by 'cheating', thus remaining behind what is required in terms of pollution abatement. Given that within the overall policy process adjustments to local conditions typically take place at the implementation stage, the next sections attempt to analyse the cost-effectiveness of the actual compliance paths in the four countries studied.

Table 6. 6: Cost-characteristics of incinerator parks and cost-performance of implementation paths

	France	UK	Germany	Netherlands
Aggregate compliance cost (large plant) (*) Age size (capacity)	 high (1) high (2)	 lower (3) high (1)	 lower (2) low (3)	 low (4) low (4)
Potential for cost savings (large plant) (*) age size (capacity)	 high (2) high (2)	 low (4) low (3)	 high (1) low (4)	 lower (3) high (1)
compliance reached (large plant)	by 2002	by 1996	by 1996	By 1995/96
differentiation of strictness of standards according to size in national transposition in actual implementation	 yes (exactly as defined in Directive) yes (assumedly as defined in Directive)	 yes, but more limited than in Directive no (because there existed no plants < 1 t/h)	 no no	 no no
differentiation of timing of compliance according to size in national transposition in actual implementation	 yes (exactly as defined in Directive) additionally increased through selective enforcement (whilst some plants also complied early)	 no (deadline 1 December 1996 for all plant) not as compared to Directive's requirements (but compliance partly early)	 no (deadline 1 December 1996 for all plant) not as compared to Directive's requirements (but compliance partly early)	 no (deadline 1 December 1995 for all plant) not as compared to Directive's requirements (but compliance partly early)

(*) The numbers in brackets rank the countries on a scale from 1 (highest) to 4 (lowest) for the respective indicator.

From what was said before it is obvious that, firstly, the higher the overall compliance cost, the more important becomes a cost-effective implementation in order to save costs where possible. Secondly, heterogeneity between pollution sources in a specific country makes compliance cost-savings possible through an application of differentiated solutions. In order to analyse the cost-performance of the countries' compliance paths relative to the cost-effectiveness of the Directive's standards, Table 6.6 (above) sets the findings in terms of overall abatement costs and the potential for cost-savings across the four countries in relation to information on the extent to which the countries actually differentiated abatement requirements in their national transpositions of the Directive. In this, the table also includes information on whether they applied further differentiation of requirements -unforeseen by the legislation- during subsequent implementation stages.

A reminder: The countries' relative cost positions

As to ***aggregate compliance costs***, the table indicates that France was in the worst position with respect to plant age. The UK was most disadvantaged when it comes to plant size, closely followed by France. When also taking into account small incinerators, France was most disadvantaged with respect to both plant age and size. On the other end of the scale, it is the Netherlands that seem to have been subject to the lowest aggregate compliance costs overall.

Considering capacity, the ***potential for cost savings*** was the highest in the Netherlands, quite closely followed by France. The potential for cost savings was clearly lower in the UK, and even more so in Germany. The indicator age seems to indicate that Germany had the highest potential for cost savings, but that this potential was also quite high in France, while it was lower in the Netherlands and low in the UK. It should be noted, however, that this picture may be blurred by the fact that a number of German plants were modernised after their initial start of operation (cf. Table 6.3 above), an information which was not available for the UK and the Netherlands and consequently not used in the calculation of the indirect indicators above. The age indicator is thus less reliable than the capacity indicator.

Focusing therefore on the capacity indicator one can conclude the following. France holds the second position with respect to high aggregate compliance costs and to a high cost saving potential; while the Netherlands, with the overall highest cost saving potential were subject to quite low aggregate compliance costs.

Cost-effectiveness of the countries' compliance paths

In ***France***, the identical transposition of the Directive's differentiated emission limit requirements according to plant capacity and of the respective deadlines into national regulation is in line with this country's high need and potential for cost savings. In line with this is also the differentiated compliance path this country followed, stretching compliance deadlines over the time-span allowed by the Directive. In fact, some large French incinerators kept operating up to 6 years over the Directive's deadline until finishing their upgrading work or closing down operations. Some also reduced their capacity, by this becoming subject to the later compliance deadline for small incinerators. For many small incinerators, the final deadline was passed, in some cases by up to 3 years, while an assessment of the compliance delay with respect to the transitional deadline of 1995 is not possible.

Opposite to France, ***the Netherlands***, on first sight, did not make use of the high potential for cost savings their plant park characteristics implied, which may be explained by their comparatively low aggregate compliance costs. All plants had to comply by 1 January 1995 -almost two years before the Directive's compliance deadline- and in practice this was met by all plants except one. A slightly more differentiated picture is however found when taking a closer look at the exact compliance choice taken by the Netherlands. As outlined before, compliance can be reached through two alternative ways: upgrading existing plant to the policy's requirements or closing down operations, often replacing the closed capacity by a new plant.

Comparing the indirect indicators for those Dutch plants that were closed (Table 6.7) with those that were upgraded (Table 6.8) we find that the average capacity of the closed plants is lower than that of the retrofitted plants.

Table 6. 7: Closure of plants in the Netherlands

Large municipal waste incinerators that were closed			
	Opening Year	**Capacity in t/year**	**Capacity in t/h**
Mean	1971	132500	32
Min	1966	26000	6
Max	1976	330000	79
standard deviation	4	103394	25
average age in 1996	25		
capacity dispersion		0,8	0,8

Source: own calculations

Also, on average, the plants which were closed were slightly older and hence closer to the end of their normal life span at the time compliance was required (but cf. the restrictions with respect to the reliability of the indicator 'age' above). Looking at the Dutch implementation path from this angle, one can conclude that this country actually did to some extent make use of the cost saving potential implied by its incinerator park. Unlike France however, the Netherlands did this without bypassing the Directive's requirements.

Table 6. 8: Retrofit of plants in the Netherlands

Large municipal waste incinerators that were retrofitted			
	Opening Year	**Capacity in t/year**	**Capacity in t/h**
Mean	1974	372400	89
Min	1963	69000	17
Max	1986	945000	227
standard deviation	8	342488	82
average age in 1996	22		
capacity dispersion		0,9	0,9

Source: own calculations

Turning now to the two remaining countries, the rather low aggregate compliance costs and the very restricted potential for cost savings in ***Germany*** (the lowest amongst the countries studied as far as capacity is concerned) suggests that the inflexible, uniform implementation, where all plants had to comply with identical requirements by 1 December 1996, a deadline in fact met, was less detrimental in terms of cost-effectiveness than it would have been otherwise. The ***UK*** finally, presents a case quite opposite to the Netherlands. Here the aggregate compliance costs were highest with respect to plant capacity, while the cost saving potential was rather low. This country chose a differently extreme compliance path, closing down existing capacity to a large extent. This is not astonishing, considering that a large number of UK plants came close to the end of their life span at the time the Directive required compliance. Further driving factors explaining why plant closure was the least cost strategy available to the UK are presented in the following chapter.

4 Conclusions

The objective of this chapter was an *ex post* evaluation of the implementation paths chosen and outcomes obtained by France, Germany, the Netherlands and the United Kingdom in the implementation of the 1989 Directive both with respect to goal attainment and cost-effectiveness. In particular, the question was whether implementation has (partly) reduced the inefficiencies of the Directive.

Starting with ***goal attainment***, compliance with the ambitious and highly uniform national requirements -and hence automatically with the less strict EU Directive's requirements- was quasi-perfect in the Netherlands and Germany. The UK encountered some problems in actually meeting the emission limits. Nevertheless, abatement technology in the few plants that were retrofitted was either installed on time or the operation of plant was suspended. Furthermore, given that the national regulations in Germany, the Netherlands and the UK were stricter than the EU Directive, plants on average over-complied with the EU Directive. France is the only country where compliance of large numbers of plants was delayed. This implementation gap occurred despite the application of the EU Directive's minimum requirements and its differentiated emission limit values and deadlines. The four countries are found to have followed distinct compliance paths that can be summarised as follows (cf. also Table 6.9): marginalisation of incineration (from 'existing' plant) and compliance through the closure of the majority of plants in the UK; retrofit or replacement of plants in Germany and the Netherlands; and late compliance, primarily by upgrading existing large, and closing down a high number of small, incinerators in France.

Table 6. 9: Summary of countries' compliance paths and implementation outcomes

	France	UK	Germany	Netherlands
Goal attainment strategy	Partly late upgrade or replacement of big incinerators; frequently late upgrade or closure of many small plants	Closure of majority of incinerators and shift to landfill; upgrade of remaining plants; mostly on time	Retrofit of almost all plants; on time	Retrofit or closure of plants; on time
Strictness of transposition relative to EU Directive	Largely identical	Slightly stricter and larger scope	Much stricter and larger scope	Much stricter and larger scope
Differentiation of emission standards	Differentiation equal to Directive	No differentiation amongst capacity classes effectively in use	No differentiation	No differentiation
Average goal attainment relative to EU Directive	Partly late compliance	Over-compliance	Strong over-compliance	Strong over-compliance
Average goal attainment relative to national law	Partly late compliance	(Over-)compliance	(Over-) compliance	(Over-) compliance

Source: Bültmann and Wätzold, 2000; Eames, 2000; Lulofs, 2000 and 2001; Schucht, 2000; Schucht et al., 2001

Two further findings are interesting with respect to the Commission's implementation gap statistics. These often put a strong focus on the transposition of EU law into national regulation and, generally, do not reveal whether non-compliance is a general problem or a rare event, or whether breaches occur for a few seconds only or are more long-lasting events. Firstly, in this study, administrative compliance was shown not to be an indispensable prerequisite for environmental goal attainment. Similarly, administrative compliance did not necessarily imply compliance in terms of environmental goal attainment. While formal compliance in the Netherlands and the UK was considerably delayed but environmental goal attainment high, France complied with the administrative requirements but delayed regulatory and operational compliance. This supports the earlier formulated suggestion that formal compliance with transposition requirements is not a good indicator to assess compliance with environmental requirements and does not necessarily allow us to deduce results in terms of practical application of a policy. In this sense, the Commission's statistics are likely to give a biased picture of compliance in terms of meeting substantial regulatory requirements. This is even more so where, as in the case of the Netherlands, the problems encountered with transposition were formal ones, relating primarily to the legal type of the text transposing the Directive, rather than to problems of the substantial contents of the respective text. Secondly, this chapter presented evidence that even once abatement technology is installed, plants may encounter continuing problems with respect to operational compliance. This may be due to peaks in pollution caused by inhomogeneous inputs, but also result from a malfunctioning of abatement equipment, especially in the starting phase. It is therefore important to distinguish between issues of initial and continuing non-compliance.

When focussing on the cost side of implementation, the analysis showed that in terms of ***cost-effectiveness*** the Directive's standards were not similarly detrimental for all countries studied. Furthermore, implementation led to a partial reduction of the inefficiencies of the policy objectives that were to be implemented. To illustrate this, let us first look at Germany and the Netherlands, the two countries that implemented quite similar strict and undifferentiated emission limit values. In Germany, the implementation of undifferentiated targets can be judged as comparatively little harmful with respect to cost-effectiveness, because the German plant park, in terms of plant size, was quite homogenous and therefore offered little potential for cost savings through differentiated abatement efforts. In the Netherlands, the situation was different. Here, the Directive's transposition into national policy did not make use of the cost-saving potential implied by the Dutch plant park characteristics. At first sight, this suggests that implementation in the Netherlands was less cost-effective than in Germany. However, to some degree, the Netherlands did make use of the cost-saving potential in that they applied different compliance solutions to plants with different characteristics. On average the larger plants were retrofitted, while the smaller plants were closed (and replaced). The Netherlands, therefore, made use of the small degree of decision-making foreseen by their national policy: the decision on whether to reach compliance through retrofit or plant closure.

France, with a plant park implying high aggregate compliance costs and a high potential for cost-savings, is the only country out of the four that implemented differentiated requirements. This was, firstly, achieved through application of the differentiated emission limit values and compliance deadlines as suggested by the Directive - thus taking advantage of flexibility foreseen by the European and national policy makers. Secondly, despite the fact that the Directive formulated clear and specific targets, France, furthermore, made use of unforeseen flexibility and discretion by introducing further differentiation through selective enforcement, thus not keeping to the official deadlines. Being a country for which the Directive was not sufficiently adapted to the national situation and where it created problems for implementation by leading to high costs, France fine-tuned the policy objectives to its local political and economic exigencies by controlling the speed and scope of implementation. By this, France allowed for the time necessary to develop cost-effective waste treatment alternatives and brought the upgrading investment closer in line with investment cycles, thus increasing the cost-effectiveness. The selective enforcement in France, however, had a negative impact on environmental goal attainment. The UK's specific compliance path, finally, can be judged as cost-effective as well. Here, the closure of a large number of municipal waste incinerators and the transfer of waste to the cheaper treatment alternative landfill clearly made compliance less costly than it would have been had the incinerators been retrofitted to meet the Directive's requirements. The UK thus reached the Directive's target at overall lower costs.

Summing up, the specific compliance paths of France and the UK, to some extent, did reduce the Directive's inefficiencies. For Germany and the Netherlands, a judgement is more difficult, because the influence of the Directive on the policy objectives implemented was rather low. This will become more obvious in the following chapter which studies the driving forces behind the implementation paths of the four countries and in this aims at a certain amount of disentangling between the impacts of the policy itself and other factors influencing the behaviour of regulated agents.

Annex 6.A Emission standards and monitoring requirements in the transposition of the Directive in France, Germany, the Netherlands and the United Kingdom

Table 6.A. 1: Emission standards for existing municipal waste incineration plants according to the arrêté of 25th January 1991

Pollutants	Plants < 1 t/h		Plants ≥ 1 t/h and < 3 t/h		Plants ≥ 3 t/h and < 6 t/h		Plants ≥ 6 t/h
	1995	2000	1995	2000	1995	2000	1996
Dust	600	200	100	100	100	30	30
Pb+Cr+Cu+Mn				5		5	5
Ni+As				1		1	1
Cd+Hg				0,2		0,2	0,2
HCL		250		100		50	50
HF				4		2	2
SO2				300		300	300
CO	100	100	100	100	100	100	100
organic compounds		20		20		20	20
vertical speed of combustion gas leaving the chimney	> 8 m/s		> 8 m/s		> 12 m/s		> 12 m/s
Combustion conditions	The combustion gas must, after the last injection of combustion air, and even under the most unfavourable conditions, reach a temperature of at least 850 °C for at least 2 seconds in the presence of at least 6% of oxygen.						

Source: Arrêté du 25 janvier 1991

Table 6.A. 2: French monitoring requirements for MWI

Pollutant	monitoring requirement				
	by 1 December 1995		by 1 December 2000		by 1 December 1996
	< 1 t/h	≥ 1 t/h and < 6 t/h	< 1 t/h	≥ 1 t/h and < 6 t/h	≥ 6 t/h
total dust	at least once a year	continuously Weight controls at least once a year	at least once a year	continuously spot checks (at least once a year) by external organism	
CO	at least once a year	Continuously	at least once a year	continuously spot checks (at least once a year) by external organism	
Oxygen	at least once a year	Continuously	at least once a year	Continuously	
HCL		Weight controls at least once a year	at least once a year	continuously spot checks (at least once a year) by external organism	
heavy metals				spot checks (at least once a year) by external organism	
HF				spot checks (at least once a year) by external organism	
SO2				spot checks (at least once a year) by external organism	
organic compound			at least once a year	spot checks (at least once a year) by external organism	
CO2		Weight controls at least once a year			
	general requirements				
reference time of combustion gases	a complete measurement has to be carried out under most unfavourable conditions				
gas temperature	Continuously				

Source: Arrêté du 25 janvier 1991

Table 6.A. 3: EU and German emission limits in mg/m3 for existing incinerators

Pollutant	EU Directive Deadline 1 Dec. 1996 for incinerators > 6t/h; 1 Dec. 2000 for smaller incinerators			German 1990 Incineration Ordinance To be met by 1 Dec. 1996	
	< 1 t/h	1 t/h to 3 t/h	> 3 t/h	all capacities - daily average emission limits	all capacities - half-hourly emission limits
Dust	200	100	30	10	30
HCl	250	100	50	10	60
HF	'divided'	4	2	1	4
SO_2	-	300	300	50 (SO2 + SO3)	200 (SO2 + SO3)
CO	100	100	100	50	100 (hourly limit)
Organic Compounds	20	20	20	10	20
NO_2	-	-	-	200	400
dioxins and furans	-	-	-	0.1 ng/nm^3	0.1 ng/nm^3
				average of measurement series	
Pb+Cr+Cu+Mn	-	5	5	0.5 (including Sb+As+Co +Ni+V+Sn)	0.5 (including Sb+As+Co +Ni+V+Sn)
Ni+As	-	1	1	(included above)	(included above)
Cd+Hg	-	0.2	0.2	0.05 (Cd+TI) 0.05 (Hg)	0.05 (Cd+TI) 0.05 (Hg)

Emission limits in mg/m3 for existing incinerators. Standard conditions: 273 degrees K, 101,3 kPa, 11% Oxygen or 9% CO_2. - Source: 17. BImSchV

For a direct comparison of the German emission limit requirements with those of the EU Directive cf. columns 4 and 5 in table 6.A.3.

Table 6.A. 4: German monitoring requirements for MWI independent of plant capacity

Pollutant	monitoring requirement	results indicating compliance
Dust	continuously	no daily average value and no half-hourly average value must exceed the emission limit value
CO	continuously	no daily average value and no half-hourly average value must exceed the emission limit value 90% of all measurements taken within 24 hours must not exceed 150 mg/M3
Oxygen	continuously	no daily average value and no half-hourly average value must exceed the emission limit value
HCL	continuously	no daily average value and no half-hourly average value must exceed the emission limit value
heavy metals	periodically	no measurement must exceed the emission limit value
HF	continuously	no daily average value and no half-hourly average value must exceed the emission limit value
SO2	continuously	no daily average value and no half-hourly average value must exceed the emission limit value
organic compound	continuously	no daily average value and no half-hourly average value must exceed the emission limit value
NOx	continuously	no daily average value and no half-hourly average value must exceed the emission limit value
dioxins/furans	periodically	no measurement must exceed the emission limit value
temperature of flue gases	continuously	
water vapour	continuously	
Combustion conditions	continuously	

Heavy metals are to be measured over a period of 0.5 to 2 hours, dioxins/furans over a period between 6 and 16 hours. The average value of measured emissions must not exceed the emission standard value. - Source: 17. BImSchV 1990, article 11 to 14.

Table 6.A. 5: EU and Dutch emission limits in mg/m3 for existing incinerators

Pollutant	1989 EU Directive Deadline 1 Dec. 1996 for incinerators > 6t/h; 1 Dec. 2000 for smaller incinerators			1993 NL Waste Incineration (Air Emission) Decree Deadline 1 Jan. 1995
	< 1 t/h	**1 t/h- 3 t/h**	**> 3 t/h**	**All capacities**
Dust	200	100	30	5
Pb+Cr+Cu+Mn	-	5	5	1 (including Ni+As+Sb+V+Sn +Co+Se+Te)
Ni+As	-	1	1	
Cd+Hg	-	0.2	0.2	0.05 (Cd) 0.05 (Hg)
HCl	250	100	50	10
HF	'divided'	4	2	1
SO_2	-	300	300	40
CO	100	100	100	50
Organic Compounds	20	20	20	10
NO_2	-	-	-	70
dioxins and furans	-	-	-	0.1 ng/nm^3

Source: Waste Incineration (Air Emission) Decree 1993, article 3

Table 6.A. 6: Dutch monitoring requirements for MWI independent of plant capacity

Pollutant	**monitoring requirement**	**results indicating compliance**
Dust	Continuously	no discretely measured result must exceed the emission limit (1)
CO	Continuously	97% of the calculated hourly average concentrations by weight of a calendar year must not exceed the emission limit
Oxygen	Continuously	97% of the calculated hourly average concentrations by weight of a calendar year must not exceed the emission limit
HCL	Continuously	no discretely measured result must exceed the emission limit (1)
heavy metals	periodically	no result measured must exceed the emission limit
HF	periodically	no result measured must exceed the emission limit
SO2	continuously	97% of the calculated hourly average concentrations by weight of a calendar year must not exceed the emission limit
organic compound	continuously	97% of the calculated hourly average concentrations by weight of a calendar year must not exceed the emission limit
NOx	continuously	97% of the calculated hourly average concentrations by weight of a calendar year must not exceed the emission limit
dioxins/furans	periodically	no result measured must exceed the emission limit
Temperature of flue gases	continuously	97% of the calculated hourly average concentrations by weight of a calendar year must not exceed the emission limit
water vapour	continuously	97% of the calculated hourly average concentrations by weight of a calendar year must not exceed the emission limit

(1) Although these emissions have to be measured continuously and discontinuously, the results of the continuous measurement of these components must for the time being be regarded as indicative only. This is so because the results from the continuous method are not (yet) considered to be always reliable (Annex A, 4.1.a) - Source: Waste Incineration (Air Emission) Decree 1993, article 4., 5.4 and 5.5

Table 6.A. 7: EU and UK emission limits in mg/m3 for existing incinerators

Pollutant	EU Directive Deadline 1 Dec. 1996 for incinerators > 6t/h; 1 Dec. 2000 for smaller incinerators			EU Directive Transitional arrangement from 1 Dec. 1995 to 1 Dec. 2000 for incinerators < 6t/h		UK Municipal Waste Incineration Directions 1991 Deadline 1 Dec. 1996
	< 1 t/h	**1 t/h-3 t/h**	**> 3 t/h**	**<1t/h**	**1-6t/h**	**> 1 t/h**
Dust	200	100	30	600	100	30
Pb+Cr+Cu +Mn	-	5	5	-	-	1 (including As+Ni+Sn)
Ni+As	-	1	1	-	-	
Cd+Hg	-	0.2	0.2	-	-	0.1 (Cd) 0.1 (Hg)
HCl	250	100	50	-	-	30
HF	'divided'	4	2	-	-	2
SO_2	-	300	300	-	-	300
CO	100	100	100	100	100	100
Organic Compounds	20	20	20	-	-	20
NO_2	-	-	-	-	-	350
dioxins and furans	-	-	-	-	-	1 ng/nm^3
combustion conditions	the gas resulting from the combustion must be raised, after the last injection of combustion air and even in the most unfavourable conditions, to a temperature of at least 850°C for at least two seconds in the presence of at least 6% oxygen for large incinerators (by 1 December 1996) and for a sufficient period of time to be determined by the competent authority (Inspector) for smaller plants (by 1 December 1995)					

Source: Directive 89/369/EEC and Guidance note IPR 5/3, 6 3.2.3, 4.1 - 4.3

Table 6.A. 8: UK monitoring requirements for MWI > 1 t/h

Pollutant	monitoring requirement	requirements indicating compliance
particulate matter (dust)	Continuously	95% of the hourly average readings for each rolling 24 hours do not exceed the emission value and the peak hourly average does not exceed 1.5 times the limit values
CO	Continuously	100 mg/m3: hourly average; 150 mg/m3: 95% of all measurements determined as 10 minute average values taken in any 24 hour period
HCL	Continuously	95% of the hourly average readings for each rolling 24 hours do not exceed the emission value and the peak hourly average does not exceed 1.5 times the limit values
heavy metals	periodically (quarterly measurements, as a minimum)	no measurement must exceed the emission limit value
HF	periodically (quarterly measurements, as a minimum)	no measurement must exceed the emission limit value
SO2	periodically (should be undertaken)	no measurement must exceed the emission limit value
NO	periodically (should be undertaken)	no measurement must exceed the emission limit value
VOCs (organic compound)	periodically (should be undertaken)	no measurement must exceed the emission limit value
Dioxins	Annually	no measurement must exceed the emission limit value
water vapour	generally continuously	
temperature of gases and pressure	Continuously	

Source: IPR 5/3 1992, article 4.4, 4.5 and Annex A4, 4.1

Annex 6.B French ‘existing’ municipal waste incineration plant park - plants in operation in the early 1990s

For information on how these data bases were established cf. annex 6.C.

Table 6.B. 1: Compliance information on the French plant park - capacity group c < 1 t/h

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Briord	<1	1989	I	0	closed
Chanay	<1	1984	I	0	closed
Echallon	<1	1982	I	0	closed
Jujurieux	<1	1982	I	2	closed
Vieu d'Izenave	<1	1982	I	0	closed
Pierrefitte-sur-Loire	0,8	1986	I	< or = 0	assumedly closed
Peyruis	<1	1989	NV	2	closed
St-Marcel-les-Annonay	0,95	1984	I	NV	NV
Vanosc	0,9	1987	I	< or = 0	assumedly closed
Conques	<1	1980	I	1	closed
Quillan	<1	1984	I	1	closed
Camares	<1	1986	I	NV	probably to be closed
Aurillac	0,98	1989	IR	3	upgraded?
Rochefoucauld	0,7	1985	I	<0	assumedly closed
Clerac	0,9	1979	I	<0	assumedly closed
Luri	0,95	1991	I	NV	NV
San Lorenzo	0,5	1988	I	1	closed
Bréhat (Ile de Brehat)	0,95	1988	I	3	closed
Cadours	0,95	1987	I	0	closed
Mauvezin	0,95	1989	I	1	closed
Taussac-La-Billiere	0,6	1989	I	<0	assumedly closed
Saint-Sauveur-en-Rue	0,6	1981	I	<0	assumedly closed
Gramat	0,7	1985	I	<0	assumedly closed
Chirac	<1	1988	I	3	closed
Florac	<1	1988	I	2	probably to be closed
Dieuze	0,9	1990	I	NV	NV
Sichamps	0,97	1982	I	0	closed
Guarbecque	<1	1989	I	0	upgraded
Tarare (Saint Forgeux)	<1	1982	IR	2	upgraded
Cluny	<1	1986	I	NV	NV
Vendenesse-Les-Charolles	0,9	1978	I	<0	assumedly closed
Moutiers (Villarlurin)	<1	1990	I	0	directly compliant?
St-Paul-en-Chablais (Lyaud)	0,95	1988	I	NV	probably to be closed
Caylus	0,95	1989	I	2	closed
Tonnerre	<1	>1986 (??)	I	2	closed
Broons	0,9	1986	I	-2	closed
Chapelle-en-Vercors	0,9	NV	I	< or = 0	assumedly closed
Die	0,9	NV	I	< or = 0	assumedly closed
Morcenx	0,9	1989	I	-2	closed
Aigueblanche	0,9	1984	I	< or = 0	assumedly closed
Lautrec	0,9	1984	I	< or = 0	assumedly closed
Bellac	0,9	1980	I	< or = 0	assumedly closed
Contrexeville	0,9	1979	I	-1	closed
Lerrain	0,9	1979	I	-1	closed
Xertigny	0,9	1979	I	-1	closed

Table 6.B. 2: Compliance information on the French plant park - capacity group 1 t/h </= c </= 3 t/h

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Oyonnax-Groissiat	2,9	1970	I	2	closed
Allos	1,5	1989	I	> or = 1	assumedly upgraded
Mison	<2	1990	I	2	closed
Briancon	2	1974	I	<0	assumedly closed
Château-Ville-Vieille	1	1981	I	> or = 2	closed
Cannes	1,8	1971	I	< or = 0	assumedly closed
Isola	1,5	1987	I	<0	assumedly closed
Malamaire (Valderoure)	1	1976	I	2	closed
Tende	1,5	1987	I	2	closed
Valberg (Guillaumes)	1,5	1981	I	2	closed
Coucouron	1	1988	I	<0	assumedly closed
Cros-de-Géorand	1	1989	I	0	closed
Montpezat	1	1987	I	0	closed
St-Côme-d'Olt	<2	1988	I	> or = 2	closed
Arles	3	1975	I	4	closed
Jonzac	2,8	1982	IR	2	upgraded
Surgères	2	1979	I	2	upgraded
Catteri	2	1988	I	NV	NV
St-Florent (Pieve)	2	1989/90	I	NV	NV
Venaco	2	1987	I	< or = 0	assumedly closed
Chatillon/Seine (Sainte Colombe sur Seine)	2	1985	IR	NV	NV
Montbard (Nogent-les-Montbard)	1,5	1980	I	2	closed
Saulieu	1,5	1984	I	> or = 2	closed
Pleumeur-Gautier	1	1977	I	<0	assumedly closed
Bernay	1	1974	I	<0	assumedly closed
Pont-Audemer	2	1972	I	1	closed
Confort-Meilars (Meilars)	2,75	1974	I	> or = 1	assumedly upgraded
Tornac	1	1989	I	<0	assumedly closed
Villefranche-de-Lauragais	1,3	1974	I	0	closed
Montgaillard-de-Salies	1	1983	I	<0	assumedly closed
Olonzac	1	1984	I	<0	assumedly closed
Cesson-Sevigne	1	1979	I	<0	assumedly closed
Saint-Benoît-la-Forêt	2,8	1982	IR	0	upgraded
Crolles	1,8	1974	I	<0	assumedly closed
Livet (Livet et Gavet)	1,8	1978	I	-2	assumedly replaced
Pontcharra	3	1977	I	1	upgraded
Brevans	3	1975	I	<0	assumedly closed
Messanges	3	1976	I	1	upgraded
Peyrehorade	1	1988	I	<0	assumedly closed
Vernou-en-Sologne	2,3	1986	IR	0	upgraded
St-Bonnet-le-Château (Estivareilles)	1	1982	I	NV	probably to be closed
Courtenay	1	1979	I	1	closed
Château-Gontier (Aze)	2	1984	IR (?)	0	closed
Pontmain	3	1983	IR	0	upgraded
Marville	1	1983	I	<0	assumedly closed
Corbigny	2,4	1981	I	<0	assumedly closed
Cosne	2,5	1974	I	1	closed
Saint-Georges -sur-l'Aa	3	1972	I	<0	assumedly closed
Saint-Hilaire-sur-Helpe	2	1981	I	<0	assumedly closed
Saint-Ouen-sur-Iton	2	1973	I	<0	assumedly closed

Table 6.B.2 continued

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Bayonne	2,5	1991	IR	0	directly compliant
Mourenx	2	1990	IR	0	directly compliant
Argelès-sur-Mer	3	1975	I	2	Closed
Saillagouse	1	1967	I	<0	assumedly closed
St-Féliu-D'Avall	1,5	1981	IR	2	Closed
St-Paul-de-Fenouillet (Lesquerde)	1	1990	I	2	Closed
Aspach-le-Haut	<3	1990	IR	0	directly compliant
Saint-Germain	1,3	1984	I	<0	assumedly closed
Vinzelles	1	1981	I	<0	assumedly closed
Bathie	1	1983	I	<0	assumedly closed
Tignes	3	1985	I	0	Upgraded
Faverges	1	1977	I	<0	assumedly closed
Fécamp (Senneville-sur-Fecamp)	3	1974	I	NV	NV
Le Tréport	3	1972	I	2	closed
Auvillar	2	1984	I	2	Closed
Apt	2,5	1984	IR	NV	probably to be closed
Orange	3	1977	I	NV	probably to be closed
Bessines-sur-Gartempe	2	1985	I	<0	assumedly closed
Sens	3	1988	IR	2	upgraded
Feche l'Eglise	1,5	1970	I	<0	assumedly closed
Vendeuvre	2	1980	I	< or = 0	assumedly closed
Is-sur-Tille	1,5	1983	IR	< or = 0	assumedly closed
Vercel-Villedieu-le-camp	1,5	1983	I	-2	closed
Dangeau	1,5	1993	I	-1	closed
Nogent-le-Rotrou	3	1976	I	-2	closed
Ouarville	1,5	1976	I	< or = 0	closed/replaced
Plougoulm	2,1	1973	I	-1	closed
Sauve	1,5	1977	I	< or = 0	assumedly closed
Cazeres	1,5	1974	I	-2	closed
Pauilhac	1,4	1990	I	-2	closed
Agde	3	1974	I	< or = 0	assumedly closed
Pezenas	3	1981	I	< or = 0	assumedly closed
Baguer-Pican	1	1980	I	-2	closed
Redon	3	1974	I	< or = 0	assumedly closed
Tinteniac	3	1984	I	< or = 0	assumedly closed
Pont-de-Beauvoisin	1	1983	IR	< or = 0	assumedly closed
Saint-Laurent-du-pont	1	1982	I	< or = 0	assumedly closed
Saint-Marcellin	2,5	1979	I	< or = 0	assumedly closed
Nouan-le-Fuzelier	2,3	1984	I	< or = 0	assumedly closed
Lasse	1,3	1981	I	< or = 0	assumedly closed
Preporche	1	1983	I	< or = 0	assumedly closed
Rouy	1,5	1981	I	-2	closed
Sainte-Austreberthe	1,7	1974	I	-2	closed
Bolquere	1	1972	I	-2	closed
Echenoz-la-Meline	3	1970	I	< or = 0	assumedly closed
Melisey	1	1985	I	-2	closed

Table 6.B.2 continued

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Saint-Martin-de-Belleville	1,5	1975	I	< or = 0	assumedly closed
Coulommiers	3	1970	I	< or = 0	assumedly closed
Samoreau	3	1968	I	< or = 0	assumedly closed
Eppeville	2	1973	I	-2	closed
Aussillon	1,6	1976	I	< or = 0	assumedly closed
Saint-Juery	1,5	1972	I	< or = 0	assumedly closed
Negrepelisse	1,4	1982	I	< or = 0	assumedly closed
Sillans-la-Cascade	1,5	1973	IR	< or = 0	assumedly closed
Morville	2,5	1975	I	< or = 0	assumedly closed

Table 6.B. 3: Compliance information on the French plant park - capacity group 3 t/h < c </= 6 t/h

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Sandrans	4,5	1977	I	<0	assumedly closed
Touques	6	1974	I	<0	assumedly closed
Angoulême (Couronne)	5	1986	I	3	upgraded
Echillais	5	1990	IR	0	assumedly upgraded
Paille	3,5	1981	I	NV	NV
St-Pierre-d'Oléron	5	1974	I	NV	NV
Plouisy	3,2	1972	I	<0	assumedly closed
Taden	4	1976	I	-2	Closed/replaced
Besançon	5	1971	IR	2	upgraded/downsize/replaced
Pontarlier	5	1989	IR	0	upgraded
Chateaudun	3,5	1976	I	0	upgraded
Saint-Martin-de-Valgalgues	3,2	1980	I	<0	assumedly closed
Vitré	4	1988	I	0	upgraded
Pontenx	4	1974	I	-3	closed/replaced
Amilly	4,8	1969	I	-9	replaced
Pithiviers	3,25	1985	IR	0	upgraded
Agen (Passage)	4,2	1982	IR	0	upgraded
Cholet (La Séguinière)	4	1983	IR	0	upgraded
Tronville-en-Barrois	3,5	1983	IR	0	upgraded
Plouharnel	4,2	1971	I	0	upgraded
Pontivy	4,5	1990	IR	0	upgraded
Fourchambault (Nevers)	3,5	1978	I	0	closed/replaced
Nogent-sur-Oise	4	1976	I	1	closed
St-Omer	4,2	1976	I	2	closed
Villefranche-sur-Saône	4,5	1984	IR	2	upgraded
Gilly-sur-Isère	4	1984	I	NV	NV
Valezan	3,3	1991	I	0	assumed as directly compliant
Thonon-les-Bains	5	1988	IR	0	upgraded
Dieppe (Rouxmesnil-Bouteilles)	5	1973	IR	3	upgraded
Lillebonne	3,6	1974	IR	4	closed
Montereau (Montereau-Faut-Yonne)	4,2	1971	I	NV	NV
Ozoir-la-Ferrière	4,2	1969	I	0	closed
Vaux-le-Penil	4	1974	I	3	probably to be closed
Montauban	5	1986	IR	0	upgraded

Table 6.B.3 continued

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Avignon (Vedene)	5	1971	I	-6	replaced
Loriol-du-Comtat	3,8	1973	I	1	closed
Belfort	4	1989	IR/I	2	replaced
Lisieux	3,5	1973	I	-1	closed
Medis	5	1986	I	< or = 0	assumedly closed
Mainvilliers	6	1971	I	-2	closed
Teste	6	1974	I	< or = 0	assumedly closed
Issoudun	4	1978	I	< or = 0	assumedly closed
Gien	3,6	1974	I	-1	replaced
Tilloy-les-Mofflaines	4	1976	IR	< or = 0	assumedly closed
Perpignan	4	1974	I	< or = 0	assumedly closed
Niort	6	1972	I	< or = 0	assumedly closed
Calvaire-sur-Mer	4	1978	I	< or = 0	assumedly closed

Table 6.B. 4: Compliance information on the French plant park - capacity group c > 6 t/h

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Peyrieu	11,4	1982	I	<0 or max 2	assumedly closed
Bayet	9	1982	IR	2	Upgraded
Antibes (Vallauris)	19	1970	I	2	Upgraded
Nice	36	1977	IR	2	Upgraded
Caen (Colombelles)	16	1971	IR	0	Upgraded
La Rochelle	8	1988	IR	4	Upgraded
Brive (Saint Pantaleon de Larche)	10,5	1972	IR	1	Upgraded
Dijon	23,2	1974	IR	2	Upgraded
Besançon	7	1971	IR	2	Upgraded/downsized
Montbéliard	8	1988	IR	-8	assumedly built to meet new requirements
Chartres	8	1970	NV	2	Closed
Brest	18	1988	IR	0	Upgraded
Concarneau	7,8	1989	IR	0?	Upgraded
Toulouse	38	1969	IR	-2	Upgraded
Bordeaux (Cenon)	16	1981	IR	-2	Upgraded
Rennes	18	1968	IR	NV	Upgraded
Bourgoin-Jallieu	11	1986	IR	0	Upgraded
Grenoble (Tronche)	24,75	1974	IR	-1	Replaced
Benesse-Maremne	7,5	1972	I	-2	Replaced
Blois	7	1968	IR	3	replaced
Nantes (Valorena)	19	1987	IR	1	Upgraded
Angers (Sainte-Gemmes-sur-Loire)	15	1973	IR	4	Upgraded
Reims	13	1987	IR	1	Upgraded
Nancy (Ludres)	6,8	1974	IR	-1	Replaced
Metz	12	1970	IR	<0	assumedly closed
Douchy	10	1977	I	> or = 4	Upgraded
Dunkerque	8	1970	I	2	Closed
Halluin	10	1967	I	2	Closed
Maubeuge	11	1980	IR	6	Upgraded
Sequedin	30	1974	I	2	Closed

Table 6.B.4 continued

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Valenciennes (Saint-Saulve)	15	1977	IR/I	0?	Upgraded
Wasquehal	30	1974	I	2	Closed
Nogent-sur-Oise	8	1970	I	2	Downsize
Hénin-Baumont	12	1972	I	2	Closed
Labeuvrière	20	1976	IR	0	Upgraded
Noyelles-sous-Lens	13,4	1972	I	3	Upgraded
Pau (Lescar)	17	1976	IR/I	2	Upgraded
Haguenau (Schweighouse-sur-Moder)	10	1990	IR	-6	NV
Strasbourg	44	1974	IR	-1	Upgraded
Colmar	12	1988	IR	0	Upgraded
Mulhouse (Didenheim)	9	1972	IR	3	Closed/replaced
Lyon Nord (Rillieux-la-Pape)	24	1989	IR	0	Upgraded
Lyon Sud	36	1989	IR	1	upgraded
Le Mans	28	1973	IR (I)	4	upgraded
Chambéry	14	1977	IR	0	upgraded
Annecy (Chavanod)	14,4	1986	IR	1	upgraded
Marignier (Cluses)	7,5	1982	IR/I	NV	downsize
Le Havre	24	1970	IR	4	upgraded until replaced
Rouen	20	1970	IR	4	Closed/replaced
St-Thibault-des-Vignes	20	1985	IR	0	upgraded
Carrières-sur-Seine	19	1977	IR	0	upgraded
Thiverval-Grignon	34,9	1975	IR	-2	upgraded
Toulon	38	1984	IR	-3	upgraded
Poitiers	8	1984	IR	1	upgraded
Limoges	15	1989	IR	2	upgraded
Epinal (Rambervilliers)	7	1982	IR/I	-3	upgraded
Belfort	8	1974	IR/I	3	downsize
Massy	11	1986	IR	-4	upgraded
Villejust	14	1972	IR/I	3	upgraded
Issy-les-Moulineaux	76	1965	IR	-3	upgraded
St-Ouen	84	1990	IR	1	upgraded
Créteil	16	1979	IR	1	closed of existing plant park, downsized
Ivry	100	1969	IR	-1	Upgraded
Rungis	17	1984	IR	3	Upgraded
Argenteuil	24	1974	IR	-2	Upgraded
Sarcelles	20	1978	IR	-4	Upgraded

Table 6.B. 5: Compliance information on the French plant park - capacity group: NV (probably very small plants)

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Courmangoux	NV	1985	I	< or = 0	assumedly closed
Feillens	(10 t/j)	1981	I	NV	Closed
Hotonnes	NV	1982	I	<0	assumedly closed
Injoux-Genissiat	NV	1982	I	NV	Closed
Lalleyriat	NV	1980	I	< or = 0	assumedly closed
Chauny	NV	1975	I	NV	Closed

Table 6.B.5 continued

Plant/Commune	Plant capacity	Date of operation start	With energy recovery (IR), without (I)	Compliance delay	Compliance choice
Flavigny-le-Grand-et-Beaurain	NV	1987	I	< or = 0	assumedly closed
Saint Quentin	(47 t/j)	1969	IR	<0	assumedly closed
Tergnier	NV	1972	I	NV	Closed
Digne-Les-Bains	NV	NV	I	<0	assumedly closed
Lurs	NV	1989	I	< or = 0	assumedly closed
Privas	NV	1976	I	< or = 0	assumedly closed
Saint-Alban-d'Ay	(11 t/j)	1987	I	< or = 0	assumedly closed
Liart	NV	NV	IR/I	< or = 0	assumedly closed
Tarascon-sur-Ariege	(40 t/j)	1974	I	<0	assumedly closed
Grans	NV	1973	I	<0	assumedly closed
Yffinac	NV	1988	I	<0	assumedly closed
Genolhac	(14,4 t/j)	1989	I	<0	assumedly closed
Arsac	NV	NV	I	<0	assumedly closed
Castelnaud-de-Medoc	NV	NV	I	<0	assumedly closed
Cussac-Fort-Medoc	NV	NV	I	<0	assumedly closed
Vaulnaveys-le-Haut	NV	1981	I	<0	assumedly closed
Saint-Viaud	(11 t/jour)	1983	I	0	closed
Haussonville	NV	1985	I	<0	assumedly closed
Nivillac	NV	1991	I	<0	assumedly closed
Charite-sur-Loire	NV	1978	I	<0	assumedly closed
Trouillas	(10 t/j)	1986	I	<0	assumedly closed
Sainte-Foy-l'Argentiere	NV	1986	IR	<0	assumedly closed
Entremont-le-vieux	(20 t/j)	1979	I	<0	assumedly closed
Villiers-Ecalles	NV	1974	I	<0	assumedly closed
Chateaurenard	NV	1975	I	NV	closed
Maurs	NV	1984	I	< or = 0	assumedly closed
Dun-sur-Auron	NV	1989	I	NV	closed
Vitteaux	NV	1083	I	NV	closed
Ploufragan	NV	1989	I	< or = 0	assumedly closed
Ile-de-Sein	NV	1977	I	< or = 0	assumedly closed
Cabrières	NV	1985	I	< or = 0	assumedly closed
Mejannes-le-Clap	(12 t/j)	1982	I	< or = 0	assumedly closed
Roquemaure	(15 t/j)	1984	I	< or = 0	assumedly closed
Sommieres	NV	1983	I	NV	closed
Villeneuve-les-Avignon	(6 t/j)	1988	I	NV	closed
Bassens	NV	NV	IR	< or = 0	assumedly closed
Aigurande	(16 t/j)	1987	I	< or = 0	assumedly closed
Sainte-Eulalie-en-Born	NV	NV	IR	< or = 0	assumedly closed
Bussières	NV	NV	I	< or = 0	assumedly closed
Noiretable	NV	1986	I	< or = 0	assumedly closed
Sail-sous-Couzan	(75 t/j)	1983	I	< or = 0	assumedly closed
Strazeele	(10 t/j)	1973	I	< or = 0	assumedly closed
Arudy	NV	1982	I	< or = 0	assumedly closed
Soumoulou	NV	1990	I	< or = 0	assumedly closed
Bourg-de-Thizy	(38 t/j)	1978	I	< or = 0	assumedly closed
Echenans-sous-Mont-Vaudois	(12 t/j)	1986	I	NV	closed
Rumilly	NV	1976	I	< or = 0	assumedly closed
Ham	NV	NV	IR	< or = 0	assumedly closed
Vert-le-Grand	NV	NV	IR	< or = 0	closed/replaced
Saint-Barthelemy	NV	1989	I	< or = 0	assumedly closed
Sousville	Check	1987	I	< or = 0	assumedly closed
Blanc	Check	1975	I	< or = 0	assumedly closed

Annex 6.C: Information Covered by Various French Plant Inventories

Eight plant inventories were used to analyse the French municipal waste incinerator park (cf. chapters 5-7). All those published by the Environmental Ministry (previously MATE, now MEDD) were successively available at the Ministry's web-page (http://www.environnement.gouv.fr) but not published in print form. The eight inventories refer to different points of time and cover different samples of the overall French plant park:

- the ITOM 6 (1995) plant inventory, presenting information on the incinerator plant park in 1993; includes only those plants that were operating in 1993 (ITOM, 1995)
- the large plant inventory provided by the Ministry for the Environment (MATE), established in September 1998, presenting the current and expected situation of the large incinerators' (capacity > 6 t/h) compliance behaviour; includes only those plants that were operating in 1998 (MATE, 1998)
- the ADEME (Environmental Agency) inventory, established in 1998, and presenting information about the incinerator plant park in 1998; includes only those plants that were operating in 1998 (ADEME, 1998)
- the small plant inventory provided by the MATE, established in January 2001, presenting the (expected) situation of the small incinerators' (capacity ≤ 6 t/h) compliance behaviour; includes only those plants that were operating in 2000 (MATE, 2001a)
- the MATE inventory of November 2001, presenting the current and expected compliance situation of all small incinerators; includes only those plants that were operating in 2001 (MATE, 2001b)
- the MATE inventory of January 2002, including large and small incinerators and presenting the current and expected compliance situation of all incinerators; includes only those plants that were operating in 2002 (MATE, 2002)
- the MEDD (new Ministry for the Environment) inventory of June 2002, including non-compliant large and small incinerators and presenting the expected compliance situation of these; includes only those plants that were operating in 2002 (MEDD, 2002)
- the MEDD inventory of January 2003, including large and small incinerators and presenting the current and expected compliance situation of all incinerators; includes only those plants that were operating in 2003 (MEDD, 2003)

For the econometric analysis in chapter 9 only the inventories MATE (1998), ADEME (1998) and ITOM (1993) were used.

The information included in each of these inventories is presented in a synthetic way in Table 6.C.1.

Table 6.C. 1: Synthetic presentation of information covered by the inventories

	ITOM (1993)	MATE large plant (1998)	ADEME (1998)	MATE small plant (2001a)	MATE small plant (2001b)	MATE all plant (2002)	MEDD non-compliant plant (2002)	MEDD all plant (2003)
plant or municipality name	yes	Yes	Yes	yes	Yes	Yes	Yes	Yes
'département'	yes	Yes	Yes	yes	Yes	Yes	Yes	Yes
opening year	yes	Yes	Yes	yes	Yes	Yes	Yes	Yes
capacity in t/h	mostly	Yes	No	yes (one exception)	Yes	Yes	Yes	Yes
capacity in t/year (actual data for a specific year)	mostly	No	yes (mostly)	No	No	No	No	No
capacity in t/day	some cases	No	some cases	1 case	No	No	No	No
plant owner	yes	No	yes	No	No	No	No	No
plant operator	yes	No	yes	No	No	No	No	No
replacement of plant	No	Yes	Yes (some)	No	Yes	Yes	Yes	Yes
expected closure	No	Yes	yes	yes	Yes	Yes	No (but might not apply)	Yes
compliance information	No	Yes	no	yes (but incomplete)	Yes (but incomplete)	Yes (but incomplete)	Yes (but incomplete)	Yes
information on energy recovery	yes	No	yes	no	No	No	No	says that energy recovery has been generalised
population whose waste is treated in the plant	yes	No	no	no	No	No	No	No
number of ovens	yes	Yes	some	some	yes	yes	yes	Yes

How the database was established

When setting up the inventory of ‘existing’ plants for the early 1990s, first of all, ‘existing’ plants had to be distinguished from ‘new’ plants. Existing incinerators according to the Directive are those plants for which the operation permission was granted before 1 December 1990. This information was not included in either of the inventories. Instead, the date of the operation start was indicated. Given that several years can pass between a plant’s authorisation and its operation start, but that on the other hand upcoming regulations are frequently anticipated when operating licences are granted, the date of operation start does not with security allow to distinguish between the two plant types. As a rule of thumb, and in accordance with experts' opinions, all those plants, which had started operation by 1991, were considered as ‘existing’ plant, except for cases where it was obvious that they were directly constructed to the Directive's abatement requirements. Where plants have started operation before 1991 but were obviously built to the Directive's abatement requirements this is mentioned in the tables in annex 6.B and in the first two tables of chapter 5.

Secondly, the MATE and MEDD inventories, being the most recent inventories and the only ones indicating compliance information, were taken as reference inventories. This means that information given in these inventories was generally considered as correct, even if unexplained contradictions with the other inventories occurred. Information additionally needed was added from the other two inventories. For this, firstly the ADEME information was used, as it contained the relatively more recent data. Only information then still missing was added from the ITOM inventory. Furthermore, all ‘existing’ plants mentioned in ITOM and/or ADEME but not included in the MATE and MEDD inventories were added to the data base in order to establish the plant park in the early 1990s in a comprehensive way.

Various plants appeared to have been included in different inventories under different names. This information was merged, again following the hierarchy of inventories described above. In the tables in annex 6.B. the alternative names are given in brackets.

As can be seen in Table 6.C.1, information on plant compliance was dealt with exclusively in the MATE and MEDD inventories. However, taking these as reference inventories, it is sensible to assume that plants included in ITOM and ADEME inventories but not in the MATE inventories had been closed down before the year the respective MATE inventory was established (1998 for large and 2000 for small incinerators). More specifically, one can assume that plants only included in the ITOM inventory were closed down before 1998, the year covered by the ADEME inventory. Likewise, plants included in the ADEME but not in the MATE inventories can be assumed to having been shut down in between the establishment of these inventories.

Annex 6.D: Emission data of 10 NRW incinerators

The following tables indicate the results of emission measurements for 10 NRW incinerators and compare the results to the EU Directive's and German regulation's requirements.

Table 6.D. 1: Emissions of incinerator A in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	0.03	0.05	0.1	0.2	0.3	0.5	+0.02	+66.7
CO	10.5	6.5	10.5	6.5	21.0	13.0	-4.0	-38.1
HCl	0.3	0.09	0.6	0.2	3.0	0.9	-0.2	-66.7
SO_2	1.8	0.4	0.6	0.1	3.6	1.0	-1.4	-77.8
Cd + Hg	0.02	0.009	10.0	4.5	41.8[C]	18.0[C]	-0.01	-50.0
HF	0.04	0.025	2.0	1.3	4.0	2.5	-0.015	-37.5
Dioxins + furans (ng TE)	2.5	0.4			2,500.0	400.0	-2.1	-84.0

a) Emission limits of 89/369/EEC (all 10 plants have a nominal capacity of more than 6t/h) to be met as of 1 December 1996 – b) Emission limits of 17. BImSchV (daily average) to be met as of 1 December 1996 – c) Hg emissions compared to the Hg emission limit. Source: Bültmann and Wätzold, 2000

Table 6.D. 2: Emissions of incinerator B in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	2.02	1.2	6.7	4.0	20.2	12.0	-0.82	-40.6
CO	12.8	13.0	12.8	13.0	25.6	26.0	+0.2	+1.6
HCl	2.0	3.0	4.0	6.0	20.0	30.0	+1.0	+50.0
SO_2	1.1	1.7	0.4	0.6	2.2	3.4	+0.6	+35.3
Cd + Hg	0.01	0.002	5.0	1.0	20.0[C]	4.0[C]	-0.008	-80.0
HF	0.1	0.002 (1997)	5.0	0.1 (1997)	10.0	0.2 (1997)	-0.098	-98.0
Dioxins + furans (ng TE)	0.005	0.01			5.0	10.0	+0.005	+100.0

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 3: Emissions of incinerator C in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	0.2	2.9	0.7	9.7	2.0	29.0	+2.7	+1,350.0
CO	40.9	29.9	40.9	29.9	81.8	59.8	-11.0	-26.9
HCl	2.4	2.6	4.8	5.2	24.0	26.0	+0.2	+8.3
SO_2	78.8	2.6	26.3	0.9	157.6	5.2	-76.2	-96.7
Cd + Hg	0.08	—	40.0		145.2[C]			
HF	0.04	—	2.0		4.0			
Dioxins + furans (ng TE)	7.8	0.004			7,800.0	4.0	-7.796	-99.9

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 4: Emissions of incinerator D in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	5.6	1.5	18.7	5.0	56.0	15.0	-4.1	-73.2
CO	12.5	9.6	12.5	9.6	25.0	19.2	-2.9	-23.2
HCl	11.9	4.5	23.8	9.0	119.0	45.0	-7.4	-63.2
SO_2	49.4	41.6	16.5	13.9	98.8	83.2	-7.8	-15.8
Cd + Hg	0.07	0.02	35.0	10.0	120.0[C]	40.0[C]	-0.05	-71.4
HF	0.6	0.5	30.0	25.0	60.0	50.0	-0.1	-16.7
Dioxins + furans (ng TE)	0.9	0.06			900.0	60.0	-0.84	-93.3

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 5: Emissions of incinerator E in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	41.7	—	139.0		417.0			
CO	36.3	14.6	36.3	14.6	72.6	29.2	-21.7	-59.8
HCl	5.5	3.8	11.0	7.6	55.0	38.0	-1.7	-30.9
SO_2	50.8	25.2	16.9	8.4	101.6	50.4	-25.6	-50.4
Cd + Hg	0.014	0.017	7.0	8.5	23.2[C]	33.6[C]	+0.003	+21.4
HF	0.1	0.4	5.0	10.0	10.0	40.0	+0.3	+300.0
Dioxins + furans (ng TE)	0.023	0.025			23.0	25.0	+0.002	+8.7

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 6: Emissions of incinerator F in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	1.2	0.6	4.0	2.0	12.0	6.0	-0.6	-50.0
CO	34.4	21.1	34.4	21.1	68.8	42.2	-13.3	-38.7
HCl	0.8	0.3	1.6	0.6	8.0	3.0	-0.5	-62.5
SO_2	1.31	1.06	0.44	0.35	2.6	2.1	-0.25	-19.1
Cd + Hg	—	—						
HF	0.07	0.01	3.5	0.5	7.0	1.0	-0.06	-85.7
Dioxins + furans (ng TE)	0.007	0.006			7.0	6.0	-0.001	-14.3

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 7: Emissions of incinerator G in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	11.1	1.0	37.0	3.3	111.0	10.0	-10.1	-90.9
CO	74.9	17.6	74.9	17.6	149.8	35.2	-57.3	-76.5
HCl	13.7	0.96	27.4	1.9	137.0	9.6	-12.74	-92.9
SO_2	38.6	17.9	12.9	5.9	77.2	35.8	-20.7	-53.6
Cd + Hg	—	0.014		7.0		28.0[C]		
HF	0.26	—	13.0		26.0			
Dioxins + furans (ng TE)	5.8	0.009			5,800.0	9.0	-5.791	-99.8

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 8: Emissions of incinerator H in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	2.2	3.8	7.3	12.7	22.0	38.0	+1.6	+72.3
CO	19.9	23.3	19.9	23.3	39.8	46.6	+3.4	+17.1
HCl	18.2	18.8	36.4	37.6	182.0	188.0	+0.6	+3.3
SO_2	52.3	49.6	17.4	16.5	104.6	99.2	-2.7	-5.2
Cd + Hg	0.16	0.03	80.0	15.0	320.6[C]	60.0[C]	-0.13	-81.3
HF	—	—						
Dioxins + furans (ng TE)	1.336	0.079			1,336.0	79.0	-1.257	-94.1

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 9: Emissions of incinerator I in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	7.9	1.2	26.3	4.0	79.0	12.0	-6.7	-84.8
CO	30.2	16.3	30.2	16.3	60.4	32.6	-13.9	-46.0
HCl	6.7	0.3	13.4	0.6	67.0	3.0	-6.4	-95.5
SO_2	22.4	3.8	7.5	1.3	44.8	7.6	-18.6	-83.0
Cd + Hg	—	—						
HF	0.24	0.14	12.0	7.0	24.0	14.0	-0.1	-41.7
Dioxins + furans (ng TE)	—	0.009				9.0		

For the source and notes cf. Table 6.D.1 above.

Table 6.D. 10: Emissions of incinerator J in mg/m³

Pollutants	1994	1996	% of EU limits[a]		% of German limits[b]		Change	
			1994	1996	1994	1996	mg/m³	in %
Dust	4.0	1.6	13.3	5.3	40.0	16.0	-2.4	-60.0
CO	32.1	22.4	32.1	22.4	64.2	44.8	-9.7	-30.2
HCl	23.3	1.0	46.6	2.0	233.0	10.0	-22.3	-95.7
SO_2	21.7	65.7	7.2	21.9	43.4	131.4	+44.0	+202.7
Cd + Hg	0.061	0.007	30.5	3.5	121.0 [C]	13.8[C]	-0.054	-88.5
HF	0.015	—	0.75		1.5			
Dioxins + furans (ng TE)	11.1	0.07			11,100	70.0	-11.03	-99.4

For the source and notes cf. Table 6.D.1 above.

Annex 6.E: Emission data of 6 Dutch incinerators

The following tables present results from emission measurements for 6 Dutch incinerators in 1990 and 1995 in mg/m3 and as percentage of the EU Directive's emission limit values.

Table 6.E. 1: Emissions of incinerator VVI Alkmaar

Pollutant	1990 mg/m3 (% of EU requirements)	1995 mg/m3 (% of EU requirements)	Change mg/m3	Change %
Dust	37.4 (127)	10.7 (36)	-26.7	-71.4
CO	29 (29)	19.0 (19)	-10	-34.5
HCL	662.8 (1326)	7.14 (14)	-665.7	-98.9
SO2	167 (56)	50 (17)	-117	-70.1
Cd+Hg	0.09 (45)	0.009 (5)	-0.081	-0.9
HF	1.6 (80)	0.05 (3)	-1.55	-96.9

Source: Lulofs, 2000

Table 6.E. 2: Emissions of incinerator ARN Weurt

Pollutant	1990 mg/m3 (% of EU requirements)	1995 mg/m3 (% of EU requirements)	Change mg/m3	Change %
Dust	13 (43)	0.56 (2)	-12.44	-95.7
CO	17 (17)	33.5 (34)	16.5	97.1
HCL	2.9 (6)	4.2 (8)	1.3	44.8
SO2	111 (37)	14 (5)	-97	-87.4
Cd+Hg	0.019 (10)	0.07 (35)	0.051	263.2
HF	1.3 (65)	0.56 (28)	-0.74	-56.9

Source: Lulofs, 2000

Table 6.E. 3: Emissions of incinerator Avira Arnhem

Pollutant	1990 mg/m3 (% of EU requirements)	1995 mg/m3 (% of EU requirements)	Change mg/m3	Change %
Dust	1 (3)	1.5 (5)	0.5	50
CO	125 (125)	45 (45)	-80	-64
HCL	0.82 (1.6)	2.6 (5.2)	1.78	217
SO2	226 (75)	12.0 (4)	-214	-94.7
Cd+Hg	0.0162 (8)	0.035 (18)	0.0188	116
HF	0.12 (6)	0.26 (13)	0.14	116.7

Source: Lulofs, 2000

Table 6.E. 4: Emissions of incinerator AVR Rijnmond

Pollutant	1990 mg/m3 (% of EU requirements)	1995 mg/m3 (% of EU requirements)	Change Mg/m3	Change %
Dust	95 (317)	1.1 (4)	-93.9	-98.8
CO	183 (183)	37.3 (37)	-145.7	-79.6
HCL	597 (1194)	4.3 (9)	-592.7	-0.99
SO2	350 (117)	12.8 (4)	-337.2	-0.96
Cd+Hg	0.21 (105)	0.006 (3)	-0.204	-97.1
HF	3.8 (190)	0.33 (17)	3.47	-33

Source: Lulofs, 2000

Table 6.E. 5: Emissions of incinerator Gevudo Dordrecht

Pollutant	1990 mg/m3 (% of EU requirements)	1995 mg/m3 (% of EU requirements)	Change Mg/m3	Change %
Dust	19 (63)	23.7 (79)	4.7	24.7
CO	773 (773)	34.8 (35)	-738.2	-95.5
HCL	7.3 (15)	7.4 (15)	0.1	1.4
SO2	254 (85)	7.4 (2)	-246.6	-97.1
Cd+Hg	0.26 (130)	0.08 (40)	-0.18	-0.69
HF	0.13 (7)	0.1 (5)	-0.03	-23.1

Source: Lulofs, 2000

Table 6.E. 6: Emissions of incinerator Roteb Rotterdam

Pollutant	1990 mg/m3 (% of EU requirements)	1995 mg/m3 (% of EU requirements)	Change mg/m3	Change %
Dust	18 (60)	1.11 (4)	-16.9	-93.9
CO	148 (148)	25.7 (26)	-122.3	-82.6
HCL	694 (1388)	15.6 (31)	-678.4	-97.6
SO2	158 (53)	3.4 (1)	-154.6	-97.8
Cd+Hg	0.11 (55)	0.01 (5)	-0.1	-90.9
HF	3.5 (175)	0.4 (20)	-3.1	-88.6

Source: Lulofs, 2000

Chapter 7 Driving Forces behind National Compliance Paths

1 Introduction

In the previous chapter it was shown that France, Germany, the Netherlands and the United Kingdom followed distinct compliance paths in the implementation of the Council Directive on the reduction of air pollution from 'existing' municipal waste-incineration plants (89/429/EEC), of which the properties were summarised in table 6.9. The findings to a certain extent back those indicated by the Commission data presented earlier: the fact that Member States are far from equally effective in implementing EU environmental policy. In this context the political science literature has made suggestions about possible reasons as to why the effectiveness of implementation may vary across Member States. Do these apply to our cases? What has determined the discrepancies in the implementation of the MWI Directive in France, Germany, the Netherlands and the United Kingdom?

It is the objective of this chapter to identify the reasons for the implementation outcomes, i.e. the driving factors behind the incinerators' compliance paths and the authorities' enforcement behaviour, by telling the story of the implementation process in the four countries. The ultimate goal is to assess whether the national decision-makers took into account implementation costs and intentionally tried to reduce the inefficiencies in the Directive during implementation. In analysing these driving factors, the chapter adopts a broad qualitative, cross-country approach to a study of which factors influenced the implementation of the EU Directive.[59] On the basis of this analysis, relevant country-specific variables are identified and factors supporting or impeding effective implementation deduced.

The outline of this chapter is as follows: In sections 2 to 5 the implementation history is presented country by country and structured around driving forces. Section 6 presents the major results in a cross-country comparison, and section 7 discusses the findings in light of political science-based explanations of the effectiveness of implementation.

2 France

There are two striking features of the French implementation path that need to be explained: Firstly, the fact that France upgraded 76% and thus the majority of its 'existing' large municipal waste incinerators (50 out of 66 plants) while closing down approximately 82% of the small incinerators (207 out of 254 plants) between the early 1990s and 2003. Secondly, not only did plants comply late, also enforcement measures were taken with a delay compared to the official compliance deadline. As shown in the following, the major factors behind this compliance path have to do with the comparatively high costs the upgrading in particular of the huge number of small plants would have implied, the unavailability, at least in the short term, of alternatives to treat the waste of the small municipalities, several interactions with other environmental policy that created uncertainty for investment and compliance decisions, and competing interests of, and the power relationship between, the major actors responsible for implementation of the Directive on a local level.

2.1 Lacking public awareness with respect to environmental and health hazards from incineration clears the way for late compliance and enforcement

Until the mid-1990s, there was little public and political awareness of the environmental and health hazards associated with emissions, in particular dioxins, from waste incinerators in France. Indeed, incineration was generally considered to be clean, while landfill was regarded as rather problematic. With this absence of a public discussion up until the second half of the 1990s, France was in clear opposition to a number of other

59 Chapter 10 below adopts an econometric approach to quantitatively analyse the political and economic factors behind the French compliance path.

European countries where environmental and health risks related to waste incineration have been a major public concern since the end of the 1970s. This absence of concern may explain why France identically transposed the minimum requirements defined by the Directive without including additional limit values, for example for dioxins and furans.

The general public's trust in the relative cleanness of municipal waste incineration was also reflected in a policy that only in the second half of the 1980s started to develop regulation directed at atmospheric emissions from municipal waste incineration other than aimed at basic dedusting, and also in the fact that up until the late 1990s not much political effort was taken to ensure the enforcement of related environmental regulations. It was only in the second half of the 1990s that public concern about environmental and health hazards related to pollution from waste incineration started to address issues that had been central in political discussions of neighbouring countries for up to almost 20 years. In fact, around 1997, concern about dioxin pollution and its effects on health became a public issue. The fact that the dioxin topic got on the political agenda and became a public concern in France is partly explained by a Greenpeace campaign of 1996 and subsequent campaigns of the CNIID (the National Centre for Independent Information on Waste). The Greenpeace campaign dealt with municipal waste incineration in general, and with related dioxin pollution in particular.[60]

Subsequently, in early 1997, France started its official policy towards dioxin pollution from municipal waste incineration by issuing a circular (of 24th February 1997) which demanded new municipal waste incinerators to meet the emission standard of 0.1 Ng/m3.[61] France also required dioxin measurements at large 'existing' incinerators.[62] The measurement campaign following the circular and the generally rising concern about dioxin led to a media crisis: in January 1998 measurements had revealed a heavy contamination of cow milk with this pollutant in the vicinity of the incineration plant in Lille. In May 1998 all results of the dioxin measurements were available, identifying the incinerators that caused problems.[63] All this 'woke up' the French public and the public estimation of waste incineration deteriorated, by this putting the French Ministry of the Environment -which clearly stressed that compliance with the Directives allowed a first reduction of dioxins- in a stronger position to finally implement this legislation.[64] A development facilitated by the replacement of a weak Environment Minister, with a politically stronger minister, in 1997. Subsequently, action was taken to assess whether incinerators identified as causing problems were complying with the *arrêté* of 1991 or not. And as a consequence, in 1998, the Environmental Ministry finally instructed the prefects to apply enforcement measures to bring MWIs into compliance with the *arrêté* of 1991, which transposed the EU Directive into French law. Summing this up, while public concern about pollution related to waste incineration eventually resulted in the enforcement of the EU Directive and the corresponding French ministerial order, on a larger scale this did not start before 1998 (only in one case formal enforcement started in 1997) and hence almost two years after the Directive's compliance deadline with respect to large plants.

60 Greenpeace demanded, amongst other things, a 5 years' moratorium on the construction and extension of incinerators and scenarios on how to phase-out waste incineration.

61 An emission standard fixed for hazardous waste incinerators in an *arrêté* of 10 October 1996.

62 Based on a circular of 30th May 1997 (circulaire du 30 mai 1997 relative aux dioxines et furans) the Ministry for the Environment asked the prefects to set up balances of dioxin emissions related to big emitters, including the municipal waste incineration plants of a capacity of at least 6 t/h. The prefects have to prescribe annual measurements of dioxins to incinerators of that capacity group.

63 A complementary study measuring dioxin emissions from small incinerators was carried out by the ADEME, the environmental agency. These results were published in 2000.

64 Cf. chapter 4: Although dioxins are not subject to emission limit values in the 1989 Directive and neither in the 1991 ministerial order, the prescribed combustion temperature aims at reducing dioxin emissions. Furthermore, the installation of cleaning technology necessary to reach compliance with the prescribed emission limit values, as a side effect, also reduces dioxins.

2.2 Political considerations of plant owners affect compliance decisions

Late public concern about pollution from waste incineration is only one factor making unpunished non-compliance possible. Other factors have to do with the organisation both of the waste incineration sector and the administrative structure of policy implementation in France. In fact, a mix between competing political and between political and economic stakes characterises the interests of the major actors involved in implementation and enforcement. To show this, it is in order to firstly take a closer look at the interests and stakes of the municipal waste incineration sector throughout the following two sub-sections before outlining the general administrative structure of environmental policy implementation in sub-section 2.4.

In France, municipalities (collectivités locales), i.e. the smallest administrative units, have the responsibility for municipal waste collection and treatment. They consist of a mayor and a parliamentary assembly whose members are directly elected at the local level. The municipalities organise the collection and treatment of municipal waste. Although most incineration takes place at large plants around big cities, the fact that even small municipalities often have incinerators explains why France used to have a high number of ‘existing’ plants. Recall from the preceding chapters that in the early 1990s there were approximately 320 ‘existing’ municipal waste incinerators, with 66 ‘large’ plants (capacity > 6 t/h), and about 254 smaller plants (capacity ≤ 6 t/h), many of which were very small.

Municipal waste incinerators are local monopolies and during the 1990s were generally owned by the municipalities, i.e. by a political actor, although there are various ways of organising their construction and operation between private and public organisations (AMORCE, 1999).[65] According to French law, whoever applied for authorisation -be it a private company or the municipality- is liable for environmental impacts of the plant. Nevertheless, irrespective of any private sector involvement in their construction or operation, the incinerators were financed and ultimately owned by the municipalities. Hence it is the municipalities who were ultimately responsible for the decision about abatement investment and the installation of abatement equipment. The mayor, as head of the local government, was therefore the key actor with respect to abatement decisions.

The fact that France held municipal elections in 1995 in part explains at least the late compliance of large incinerators, which had to meet the Directive's requirements by December 1996. In general, decisions about costly investment are said to be stopped about a year before elections. And afterwards, it generally takes another year until the political business is re-started. The elections had an impact on compliance decisions because waste collection, treatment and related investments are financed by local taxes (AMORCE, 1999). Therefore, taxes generally have to be increased to finance additional investment in abatement equipment. As a locally elected political actor the mayor is dependent on political support in order to stay in power. Given that abatement investment is costly one can assume that the mayor, when taking compliance decisions, considers the voters' preferences for reduced emissions (and their willingness to pay higher taxes), on the one hand, against the overall fiscal pressure in his municipality and the voters' preferences for lower taxes, on the other hand. Indeed, interviews with experts indicated that in the implementation of the EU Directive mayors were likely to be in a situation of conflicting interest. On the one hand, as part of the political system, they were in charge of applying national policy. As political local actors, they had the interest to gain public support and to stay in power. Given the still low level of public awareness with respect to the risks associated with emissions from waste incineration during a large part of the 1990s, mayors were reported to have frequently found it in their interest to keep taxes low

65 As one option, the municipality is responsible for construction and a) operates the plant itself (‘régie’) or b) delegates the operation to a private firm (‘marché d'exploitation’). Alternatively, the municipality delegates both construction and operation to a private firm (‘délégation du service publique’).

(and hence emissions high), and to delay investments, at least shortly before elections. Of course, as liable plant owners mayors should have feared enforcement and related penalties, but such measures did not start before the end of the 1990s.

It was pointed out earlier that France was relatively disadvantaged in terms of compliance costs. This has, on the one hand, to do with the fact that only basic abatement was applied in the majority of plants prior to the Directive's implementation. On the other hand, it is explained by France's large number of small MWI plants. Firstly due to economies of scale, retrofitting with abatement equipment is disproportionately expensive for small plants. Secondly, small municipalities often did not have the financial means to bear these costs (without considerable tax increases). This means that the mayors of the municipalities which owned small incinerators were subject to the same potential trade-off between local political support and tax increases, on one side, and making compliance investment, on the other side. Furthermore, a viable alternative to the small incinerators was to regroup waste treatment between municipalities and to build larger waste incineration plants treating the municipal waste of several small municipalities together. Such co-ordination, however, requires a difficult political and economic co-ordination process which was only possible in the medium- to long-term. Therefore, small municipalities aimed to delay compliance of their 'existing' plants as long as possible. Indeed expert interviews suggested that they attempted to keep non-compliant small plants running, supposedly until the end of their operational lives. This also indicates that the Directive, despite not having incorporated the suggestions for stricter requirements made by the European Parliament and the Social and Economic Committee, was not undemanding for France.

2.3 New French waste law affects compliance choices by creating policy interactions and uncertainty

Next to the factors outlined before, the specific compliance decisions made have been affected by policy interactions, while high policy uncertainty additionally explains the poor compliance results. In fact, uncertainty related to waste policy has been one of the decisive factors explaining late compliance at least of the large French incinerators. To comply with the EU Directive, plant owners (municipalities) could choose between closing down their plants and retrofitting them with expensive abatement equipment. During this decision-making process, a number of new regulations were being prepared. These introduced uncertainty with respect to viable investment decisions.

As a central factor, a new French waste law (Loi n° 92-646, directed at the elimination of waste) which clearly changed the French waste policy was issued on 13th July 1992. This law sought ambitious reductions in the landfilling of waste by essentially banning the landfilling of municipal waste from 2002, and compensating for this with material and energy recycling (MarketLine, 1994).[66] By this it changed the hierarchy of waste treatment options and required a re-consideration of overall waste management and waste treatment strategies in each region. Investment decisions and with this the start of projects to bring the plants into compliance with the Directive were thus slowed down. Between 1991 and 1992/93, uncertainty also existed with respect to planned requirements for the re-use and recycling of slags from MWI plants and for the efficiency of the treatment of off-gas cleaning residues from waste incineration that were under negotiation. Especially the latter hampered abatement investment decisions as the treatment requirements had an impact on the choice between the alternative abatement technologies available. Interaction with another law (arrêté du 18 décembre 1992), which was primarily addressed at industrial hazardous waste landfills and which defined requirements for waste to be disposed off in these landfills, stopped the discussion about slags re-use. Shortly afterwards, in 1993/1994, discussions about a new EU Waste Incineration

[66] In that it reflects the French assessment of waste treatment alternatives, and the risks related to them, as outlined in sub-section 2.1.

Directive with revised emission limits began. These introduced uncertainty about future emission limit values and, hence, about viable investment decisions and again slowed down abatement decisions. Finally, new uncertainty was introduced when in 1998 the French Environmental Minister demanded a revision of the regional waste plans. These plans, required by the French waste law of 1992 and two decrees of February 1993, are established at the level of 'départements'. They co-ordinate local waste management, define the overall waste treatment strategy and stipulate how municipal waste is treated. When, in 1998, the Environmental Ministry found that the waste plans foresaw a large share of waste incineration and only a small share of recycling, it demanded their revision in order to reduce the share of waste incineration relative to organic and material recycling.[67]

Next to introducing policy uncertainty, the 1992 French waste law had a second effect. By banning the landfilling of municipal waste from 2002 and compensating for this with material and energy recycling, it indirectly promoted incineration with energy recovery and, thus, a retrofit of such plant relative to municipal waste incinerators without energy recovery. This introduced a bias in the decision of which plants were to be upgraded and which were to be closed. As far as large municipal waste incinerators are concerned, approximately 54 out of 66 plants operated with energy recovery, and the majority of those plants which did not recover energy were actually closed down or replaced. Amongst small municipal waste incinerators, the share of plants operating with energy recovery was much lower. Additionally to general compliance cost differences between large and small MWIs owing to economies of scale this policy interaction therefore is a further reason for why the majority of large incinerators were retrofitted whereas the majority of small plants, eventually, shut down operations (cf. chapter 6). Today, energy recovery is generalised across the French plant park (MEDD, 2003).

2.4 Conflicting interests and political pressure hinder enforcement

A further point to highlight are the factors that can explain why enforcers let the regulated agents have their way until the deadline for compliance set by the EU Directive was passed. It should be mentioned that France, in general, has the reputation of not preparing and timing enforcement measures so that compliance will be reached on time, but rather starting enforcement once a regulatory deadline is passed. However, this is not the only explanatory factor for late enforcement in the MWI case. An important aspect lies in the role played by the mayor (municipality) outlined above together with the power relationship he has towards the enforcement agents who in turn are influenced by conflicting policy interests themselves. Some information about the administrative structure of the French implementation system is necessary to make this obvious (cf. Box 7.1).

Box 7. 1: The French administrative system of environmental policy implementation

In France, the competence for environmental policy and its implementation is allocated to the central state where the Environmental Ministry -and here especially its department 'Directorate of the prevention of pollution and risks' (Direction de la Prévention des Pollutions et des Risques - DPPR)- is the leading organisation. With this it is also responsible for the transposition of European environmental regulation, as well as for the setting of technical standards and emission limit values on a national level. Nevertheless, there is a rather high scope of decision-making and action at a local level. The system of environmental policy directed at industrial pollution can, therefore, be considered as consisting of two levels, a national and a regional level, where the important point to note is that the discretionary power at a local level is allocated to direct representatives of the central state who carry out the practical implementation tasks.

[67] In 1998 35% of the French municipal waste was incinerated. Source: http://www.ademe.fr/collectivites/Dechets/chiffres/dec01.htm, 23/11/2000.

Box 7.1 continued

At a decentralised level, firstly, the prefect (préfet) -the governor of the Prime Minister and of all ministers of an administrative district or region- is the highest administrator of a 'département'. He constitutes the major connecting link between the central and local administration. Formally, he is the central actor of environmental policy execution on a local level, but his responsibilities are broad and also cover the execution of other political targets (social, cultural, security related issues, etc.) on a local level as well as the development of his 'département' or region. In this context, based on the municipalities' suggestions and strategies, the prefect also sets up the waste plans (plans départementaux d'élimination des déchets). By this the department controls and co-ordinates several municipalities which otherwise are responsible for their own waste management.

Secondly, the leading organisations further specifying the national environmental law on a regional level are the DRIRE (Direction Régionale de l'Industrie, de la Recherche et de l'Environnement). They are outposts of the central government and consist of central government representatives of the Environment Ministry, the Research Ministry (previously Ministère de la Recherche, now Ministère délégué à la Recherche et aux Nouvelles Technologies) and the Ministry of Economic and Financial Affairs and Industry (Ministère de l'Economie, des Finances et de l'Industrie). The DRIRE, hence, is a larger institution, not only dealing with environmental issues but also with industrial and economic policy and research. They are supervised by the three ministries, although as far as environmental policy is concerned, the Environment Ministry is the supervisory authority.

Plants subject to authorisation are specified and regulated by the 'law directed at classified installations for the protection of the environment'[68] of 1976, which constitutes the core of French environmental policy directed at industrial pollution. Various EU directives aimed at the prevention of air pollution come under its area of application. According to their environmental harmfulness plants are classified into those subject to authorisation or declaration requirements (art. 2 and 3). Plants considered as potentially highly harmful to the environment (installations classées à autorisation), have to apply for an operation licence (arrêté d'autorisation) to be granted by the prefect. These are the so-called 'installations classées à autorisation'. Plants which do not come under this category have to be registered only. These are the so-called 'installations classées à déclaration'. Municipal waste incinerators belong to the group of industrial processes that require operation permits. On a national level, the Environment Ministry is responsible for plants subject to authorisation, while on a regional level, the classified installations are in the DRIRE's responsibility. Here, the 'inspectorate of classified installations' (inspection des installations classées), a sub-division within the DRIRE, which is supervised by the Environment Ministry, is in charge of the system of authorisation and registration of industrial plants.[69] Where new legal emission limits are formulated, they have to be incorporated into the plants' operation permits. The conditions to be fulfilled by a plant in order to receive the operation licence are developed by the inspectors and the DRIRE, which thus specify, on a district or regional level, the 'arrêtés' issued on a national level. They act as consultants of the prefect in environmental questions who, formally, has the final decision.

The inspectorate is also responsible for control tasks (monitoring and enforcement), having to ensure that plants operate in accordance with the rules defined by the law on the 'installations classées' and that they meet standards laid down on a case by case basis in the plant specific operation permits. The inspectors can be considered as policemen of the 'installations classées'. With respect to both permitting and enforcement measures the DRIRE makes recommendations, which must be formally approved by the prefect. Where new legal limits are formulated, the DRIRE and prefect are responsible for their implementation.[70]

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[68] Loi relative aux installations classées pour la protection de l'environnement; Loi n° 76-633 du 19 juillet 1976. This law was further specified by décret n° 77-1133 du 21septembre 1977, dealing with authorisation procedures and the duties of plant operators.

[69] Some smaller municipal waste incinerators, however, are authorised by other institutions, such as the DDAF (department of agriculture and forestry) or by the DDASS (department of sanitary and social activity) at the level of a 'département'.

[70] As neither the DRIRE nor the prefect are institutions specialised only on the environment, and as the prefects have to approve the DRIREs' recommendations, in the following analysis, the DRIRE and the prefect are mostly considered as if they were one actor.

It is important to note that the general responsibility for the implementation of environmental legislation lies with the Environment Ministry, but that the major executing public authorities for environmental policy directed at industrial pollution and responsible for authorisation tasks as well as monitoring and enforcement of a plant's compliance are placed on a decentralised level. These decentralised institutions are the DRIRE (and here namely the inspectors) together with the prefect. In accordance with the prefects the DRIRE, thus, decide about the specific adaptation of national or European environmental legislation in their administrative district (département) and in this have a rather high scope of decision at their disposal, although their discretionary power in principle only allows them to apply stricter standards than required on a national level, and not to weaken standards.

The fact that both the prefects and the DRIRE are not only responsible for environmental policy enforcement but are more generally in charge of the economic development of the region, and furthermore that the DRIRE are supervised by several ministries with potentially diverging interests, turned out to be one factor that influenced the Directive's implementation. Such multiple objectives and interests of the institutions in charge of practical implementation can lead to problems. Especially the DRIRE with their double-responsibility had to weigh up between environmental and industrial-political interests. This is one reason why the implementation of the EU municipal waste incineration Directive was not always their primary interest or priority.

Furthermore, expert interviews indicated that the balance of power between local enforcers and politicians played a decisive role for enforcement, where the DRIRE/prefects found themselves opposite to strong locally elected representatives (mayors), who put pressure on the enforcers not to take action against non-compliant incinerators. This was made possible by the French administrative structure, where enforcement agents are representatives of the central government. With the mayors having a double role as plant owners and as political actors, this constellation put strong mayors in a position which allowed them to exert pressure, via the central government, on the enforcers. Therefore, the fact that until 1998 the DRIRE/prefect took hardly any enforcement measures can partly be explained by their general function in the administrative system but also by pressure exerted on them from the side of strong mayors. As indicated in sub-section 2.1 this situation began to change by the end of the 1990s when rising public and political awareness put the Environmental Ministry in a stronger position to enforce the Directive's implementation.

2.5 Enforcement finally results in compliance

Once the Environmental Ministry had instructed the prefects in 1998 to actually enforce the Directive's implementation with respect to large municipal waste incinerators, formal enforcement steps (outlined in Box 7.2) were applied to all 14 cases of then still non-compliant plants. In 7 cases only the first formal enforcement step, the 'mise en demeure', was applied, while in the 7 remaining cases also the second, stricter enforcement measure, the 'procédure de consignation' was used. The timing of the application of enforcement measures is presented in Table 7.1.

Table 7. 1: Timing of officially recorded formal enforcement steps towards large MWIs

Enforcement step	1997	1998	NV
'Mise en demeure'	-	6	1
'Procédure de consignation'	1	6	-

Source: http://www.environnement.gouv.fr, information published in January 1999, December 1999 and June 2000.

Compliance of the majority of the concerned plants was reached within 2 years after formal enforcement had started, while one plant took two more years before reaching compliance in 2002. Compliance of large plants, thus, was reached with up to 6 years' delay.

Box 7. 2: Formal enforcement procedure in France

Enforcement measures in France consist of administrative sanctions. Penalty sanctions (sanctions pénales) are only taken towards plants operating without prior authorisation. The prefects have a two-step formal enforcement procedure at hand. In the first step (called "mise en demeure") the prefect fixes a date by which the plant must comply. If this date is exceeded, the plant should either be closed or production suspended or the second step of the enforcement procedure (called "procédure de consignation") applied. In this step the operator has to pay the monies necessary to bring the plant into compliance into the public treasury. When compliance is achieved or the plant closed the money is repaid to the operator. The prefect can also commission the necessary upgrading work at the expenses of the plant.

With respect to small incinerators the lax policy continued. There are no indications that any formal measures have been applied to enforce the small incinerator's compliance requirements with respect to the transitional 1995 deadline. Although an interview with the Environment Ministry revealed that some enforcement measures were taken in reaction to the finding, in 1998, that 3/4 of the 190 then remaining smaller incinerators were non-compliant, there were no indications of the second, stricter formal enforcement step having been applied at that time. Forecasts expected enforcement to result in the closure of approximately 90 plants by 2000. The majority of the small plants remaining in operation in 2000, however, were still non-compliant with respect to the Directive's 2000 requirements[71] (cf. chapter 6). The point that the stricter enforcement step was not applied until much later is confirmed by an official statement of the new Environment Minister in 2002 reminding the prefects of the fact that the primary objective of the first enforcement step is to recall the requirements to be met, and that it should, therefore, be followed by the second enforcement step as soon as possible. Formal enforcement steps recorded by the Environmental Ministry (MEDD, 2002) concern 20 small incinerators. Table 7.2 presents the repartition over time of the number of plants to which the two enforcement steps were applied. All in all, by 2003, enforcement towards small incinerators had led to the closure of a majority of the previously non-compliant installations.

Table 7. 2: Timing of officially recorded formal enforcement steps towards small MWIs

Enforcement step	1998	1999	2000	2001	2002
'Mise en demeure'	3	4	-	5	5
'Procédure de consignation'	-	-	-	1	2

Apparently, also the measurement requirements defined by the Directive had not always been enforced, a fact considered as particularly widespread amongst the plants of a capacity below 3 t/h. In fact, next to inspections carried out by the inspectors, incinerators are required to self-monitor their emissions and to transfer the results to the DRIRE.[72] A study set up by the French Environmental Ministry in 1999 showed that large incinerators measured their emissions regularly, whereas not all small plants fulfilled their measurement requirements. Measuring equipment is generally installed together with the abatement equipment. Non-compliance or late compliance in France, consequently, not only existed with respect to abatement requirements imposed by the Directive but also with respect to monitoring requirements. Given the information provided by the Environmental Ministry that all small incinerators were expected to comply with the Directive by 2003, one can assume that these plants now also have their monitoring equipment installed. With all plants eventually having been forced to comply, and despite the fact that some non-compliant small incinerators could stretch their operations over the compliance deadline, it is unlikely that all these plants have managed to continue

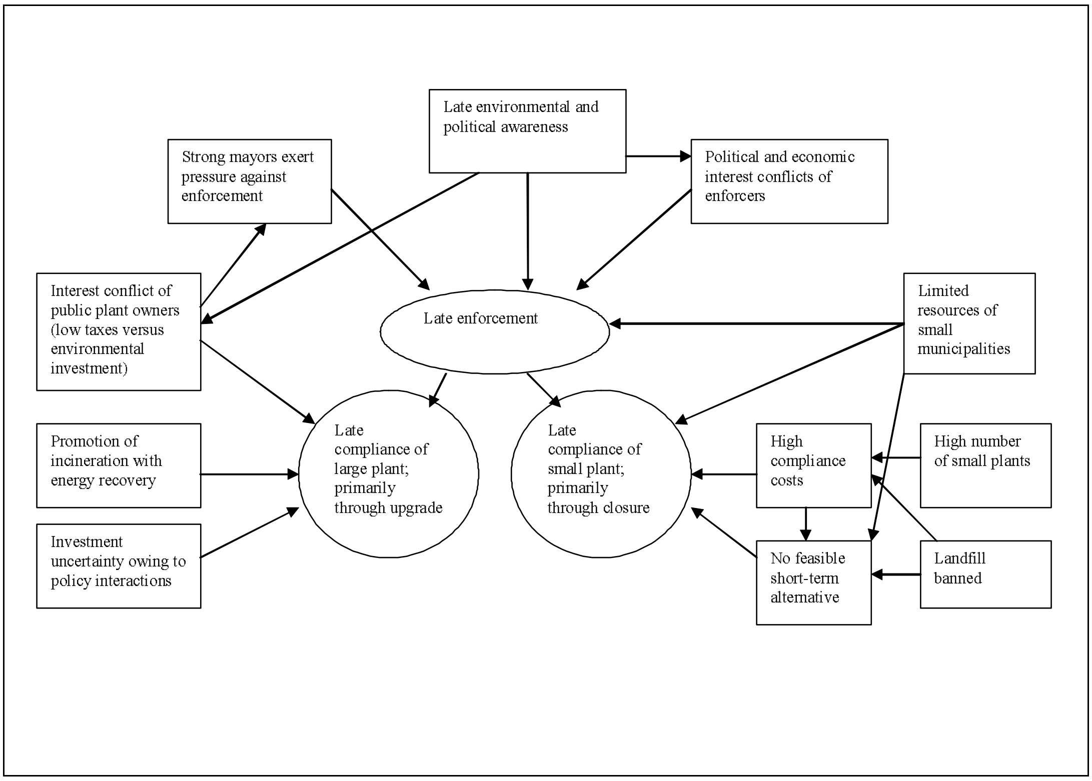
[71] Source: http://www.environnement.gouv.fr/dossiers/dechets/incineration/010122-incinerateurs-petits.htm; (18/5/2001).

[72] Additionally, recognised technical measuring organisations come to measure emissions manually. Enforcement measures are to be decided based on all these measurement results.

operations until their expected end of life without being upgraded to the Directive's requirements.

The key driving factors behind the French implementation path are summarized in Figure 7.1.

Figure 7. 1: Key explanatory variables of the French compliance path



3 Germany (North Rhine-Westphalia)

Germany[73] is part of the group of countries that implemented stricter requirements towards atmospheric pollution than those defined in the Directive. As furthermore shown in the previous chapter, German waste incineration plants rarely exceeded their emission limits, and remained normally well below these limits. This implies that they normally largely over-comply with the EU emission limit values. What is more, the majority of North Rhine-Westphalian incinerators reached compliance before the date defined in the German Ordinance and the EU Directive. Factors that can explain these outcomes are the high public awareness about health and environmental hazards related to waste incineration, the fact that municipal waste incinerators encountered no problems in passing on the costs of abatement investment to households, a voluntary agreement negotiating early compliance in NRW and the fact that monitoring was quasi perfect and non-compliance not tolerated by enforcement agencies.

3.1 High public interest in emissions from waste incineration furthers strict laws and compliance

It was back in the mid-1980s that the German public became highly concerned over emissions, particularly dioxins, from waste incinerators. These emissions were believed to have serious impacts on eco-systems and human health. At that time more than 20% of Germany's municipal waste was incinerated and this percentage was increasing.[74]

73 For Germany, the empirical analysis focuses on the Federal State *(Land)* of North Rhine-Westphalia (NRW), which, together with Bavaria, operates the highest number of waste incineration plants of all German States. Furthermore, experts were of the opinion that outcomes in the other States were comparable.

74 In 1980 West-Germany disposed off 78.5% of municipal waste in landfills, while incinerating 19.7%, the latter representing 6.3 million tonnes. In 1993 the shares for West- and East-Germany were 69.6% and 21.4%

Citizens' groups and environmental organisations pressed for tighter emission limits for existing incinerators and to prevent the building of additional plants because environmental regulation at that time (TA Luft 1986) only demanded from plant operators to decrease dioxin emissions as far as possible without setting a specific emission target. This was considered as insufficient.

By extensively using their right to object and take legal action against the authorisation and construction of new incinerators, citizens' groups and environmental organisations literally blocked all authorisation procedures. This strategy delayed the construction of new incinerators by several years and forced politicians and plant operators to take the concerns of citizens and environmental organisations seriously. This led, on the one hand, to the adoption of the German Ordinance on Incineration Plants for Waste and Similar Combustible Substances of 1990 which introduced emission limit values for dioxins. On the other hand, operators realised that if they wished waste incineration to remain a viable waste treatment alternative, they had to make efforts to satisfy the public concern. The public pressure therefore did not only lead to stricter emission limits, it also helped bring about compliance.

On a decentralised level, in reaction to the public concern about emissions from waste incineration, the North Rhine-Westphalian state government had decided to reduce these airborne emissions without waiting for the adoption of the 1990 German Ordinance. This was also driven by recent technological progress in abatement technologies. And finally, the dioxin issue was particularly important for NRW because of the relatively high number incineration plants in this German state. The NRW Environmental Ministry subsequently, in 1998, issued a study to investigate the technical potential to reduce dioxin emissions and to assess the costs such technologies would impose on operators. Once the results showed that the limit value of 0.1 ng TE/m^3 could be reached and that the necessary abatement technology did not entail excessive costs, the state government started negotiations with the plant operators. The government's aim was to establish a voluntary agreement about emission reductions, their timing and a coordination of the upgrading work across the NRW plants. While first being sceptical, operators accepted the government's plans once the providers of abatement technology guaranteed meeting the emission value for dioxins; after the authorities had agreed to grant the operators additional time to meet the requirements in the case of delays for which they were not responsible; and after the permitting authorities had promised to process the necessary permitting procedures swiftly. The voluntary agreement 'EMDA' (cf. chapter 6) was thus agreed upon in February 1990, several months before the German Ordinance came into force.

3.2 Co-operation from the operator side also furthered by the possibility to pass on compliance costs

As in the French case, also the German municipal waste sector is organised locally: district governments and independent municipalities (Kreise und kreisfreie Städte) are responsible for both the collection and disposal of municipal waste.[75] The district governments and municipalities are depending upon the state, subordinate to the state government, but have a comprehensive discretionary power. Although the exact extent to which alternative waste treatment options are used is stipulated in waste management plans, which are formulated by the state governments in co-operation with the authorities responsible for waste collection and disposal (cf. §29 KrW-/ AbfG)[76], district governments and municipalities largely decide whether waste is landfilled, incinerated or

respectively, equivalent to 8.6 million tonnes (Statistisches Bundesamt, 1996). While therefore in Germany municipal waste is mainly disposed off in landfills, the landfill share is decreasing.

[75] Some district governments and independent municipalities delegate their responsibility for the collection and/or disposal of municipal waste to community authorities or municipal service companies (Stadtwerke).

[76] In most states the waste management plans are converted into plans for individual districts and municipalities.

composted. Households are obliged to let the authorities take care of their waste and to pay waste collection fees.

Waste collection and disposal facilities de facto hold regional monopolies. They are either run by district governments and municipalities or by private companies commissioned by these local governments. Until the end of the 1980s the overwhelming majority of incinerators was in the hands of district or municipal authorities. This began to change in the early 1990s such that by 1994 only 38% of the incineration plants were still owned and run by public bodies.[77] Nevertheless, irrespective of their private or public status, the companies are obliged to burn the district's or municipality's municipal waste, and the responsibility for waste disposal as well as the competence to charge waste collection fees always remains with the district government or municipality. The fees the authorities charge equal their costs if they run waste disposal facilities themselves, or the payments they make to the private companies.

In contrast to the French case, problems with respect to the financing of abatement equipment necessary to comply with stricter regulations were not reported or seem at least not to have hindered compliance. While operators were initially opposed to dioxin emission limit values, fearing that the available technology could not reliably ensure compliance, and although clearly not being happy about tighter emission limits, they eventually adopted a rather co-operative strategy when abatement requirements were tightened. Next to their realising that stricter emission regulation would improve the image of waste incineration amongst the general public, co-operative behaviour was also brought about by the fact that abatement technology providers ensured that their technologies were able to meet the emission limit values. A further driving factor, which clearly facilitated upgrading investment, was the fact that operators were able to pass on the costs to households. In pressing for tighter emission limits, citizens' groups and environmental organisations obviously accepted that waste collection fees might rise. Although it can not be ruled out that not all households completely realised that a reduction in emissions from waste incinerators would lead to an increase in waste fees, there was no considerable resistance when fees were increased as a consequence of pollution abatement activities of municipal waste incinerators. Having autonomy over their pollution abatement decisions, as long as these meet the emission limit values defined, the (predominantly private) operators therefore have negotiated with district governments/independent municipalities to raise their payments where abatement costs have increased. Given the public's concern over emissions, the authorities accepted higher disposal charges to finance investment in pollution abatement.

3.3 Automatic monitoring and reporting ensures transparency

In general, special environmental agencies (Umweltfachbehörden) are in charge of monitoring and enforcing compliance with emission limits (cf. Box 7.3 for a short presentation of the allocation of competences in implementation of environmental legislation in Germany). But monitoring activity of the environmental agencies is supported by operators' self-monitoring. With respect to most pollutants, the regulation requires operators of municipal waste incinerators to install continuous monitoring and recording equipment. In Germany, the emission data are submitted to the supervisory authority in a yearly emission report or by telemetric transfer. North Rhine-Westphalia actually has connected the MWIs to a system of telemetric transfer of emissions (Emissions-Fernüberwachung). Here, the emission values are automatically transmitted to the supervisory authority once a day. For other pollutants, not required to be continuously measured, individual measurements are performed. Operators are obliged to commission (and pay) private authorised institutes to carry out these measurements at least once a year and forward the results to the environmental agencies. On the basis of the emission

[77] The 62% of incinerators which had been privatised usually functioned as limited liability companies, in which district governments and municipalities often held a majority of shares.

data provided by plant operators and authorised institutes, environmental agencies control compliance with legal limits.

Box 7. 3: Allocation of competences in implementation of environmental legislation in Germany

Germany consists of a three-level federal structure: federal – state (*Land*) - local. While federal and state governments have a comparatively extensive legislative power, the local governments mainly serve as executive bodies, carrying out tasks that serve to implement federal or state law. The responsibility for air pollution prevention measures has been placed with the federal government, as is mostly the case with respect to environmental legislation, and the responsible Ministry is the Ministry for the Environment, Conservation and Nuclear Safety (Ministerium für Umwelt, Naturschutz und Reaktorsicherheit). It was therefore the German government that, in 1990, enacted the Ordinance on Incineration Plants for Waste and Similar Combustible Substances, which was implemented in parallel with the EU Directive.

The implementation of federal law (acts and ordinances) is incumbent on the states. Most German States have ministries bearing 'environment' in their title, but only few States have founded ministries exclusively dedicated to environmental issues. The states are responsible for establishing or appointing permitting and supervisory authorities. In most states permitting authorities are part of the general administration, i.e. of district governments/independent municipalities. In NRW and Bavaria, the two states with the highest number of MWIs, the authorisation of waste incinerators and their abatement equipment is incumbent upon regional governments.

Supervisory authorities, in charge of monitoring and enforcing compliance with emission limits are in most states special environmental agencies (Umweltfachbehörden). This is the case in NRW and Bavaria.[78] Neither being elected bodies nor an integral part of the general administration, they have a relatively high degree of autonomy and direct decision-making powers. As authorities specialised on environmental issues they can be assumed as being less subject to interest conflicts than their French counterparts.

Particularly in North Rhine-Westphalia, the automatic emission recording, processing and telemetric transfer of data to the authorities implies that non-compliance can hardly remain unobserved. The telemetric system alerts the authorities when emission limits have been exceeded. Monitoring is further facilitated by the relatively small number of incinerators, also in Germany overall.[79] This, together with the fact that authorised private institutes perform on-site controls to check the equipment and to measure emissions, also means that monitoring places relatively few demands upon the environmental agencies' resources. The NRW environmental agencies appear to have carried out their monitoring and enforcement work carefully. In cases of non-compliance the agencies have a range of formal sanctions at hand, which reach from fines to a temporary or permanent closure of plants. While the agencies generally tried to resolve problems informally they did apply severe sanctions when they thought this necessary to obtain compliance. As example can serve the case of a plant which repeatedly encountered problems in meeting the dioxin emission limit value and which was forced to reduce waste input until the problems with the abatement technology were solved (cf. Bültmann and Wätzold, 2002). In most cases, however, operators managed to resolve breaches of emission limit values owing to a malfunctioning of abatement equipment within the time foreseen by the legislation. Therefore, the implementation gap that some authors (Lübbe-Wolff, 1993) see for German environmental policy is relatively small with respect to MWI emissions.

78 The Bavarian Environmental Agency (Bayerisches Landesamt für Umweltschutz) and the 12 State Environmental Agencies (Staatliche Umweltämter) of NRW (one for each district).

79 In the early 1990s Germany ran 48 'existing' incinerators, 14 of which were situated in North Rhine-Westphalia.

3.4 A voluntary agreement furthers early compliance in NRW

The fact that municipal waste incinerators did not only comply with environmental requirements but actually did so before the official deadline set in the Directive and the German Ordinance was brought about by the EMDA, the 'Emission Reduction Plan for Dioxins in North Rhine-Westphalia'. As pointed out in the previous chapter this voluntary agreement required all waste incineration plants to meet emission limit values for dioxins by 1 December 1995 and it turned out that it was technically not sensible to install the abatement equipment for dioxins separately from that for other pollutants covered by the 17. BImSchV. Therefore, the EMDA's retrofitting deadline effectively applied to the entire abatement technology necessary to comply with the German Ordinance and this voluntary agreement hence obliged NRW municipal waste incinerators to meet the emission limits a year before the deadline set in the ordinance.

Only few plants received exemptions from the deadline because they encountered specific site-related problems. Furthermore, it had been negotiated in the EMDA that all plants were to be given additional 6 months to optimise their abatement equipment after it had been installed. As a result, most NRW incinerators had their abatement equipment in place on time for the EMDA and consequently before the deadline fixed by the German ordinance and the EU Directive.

3.5 Technical factors explain over-compliance also with stricter German standards

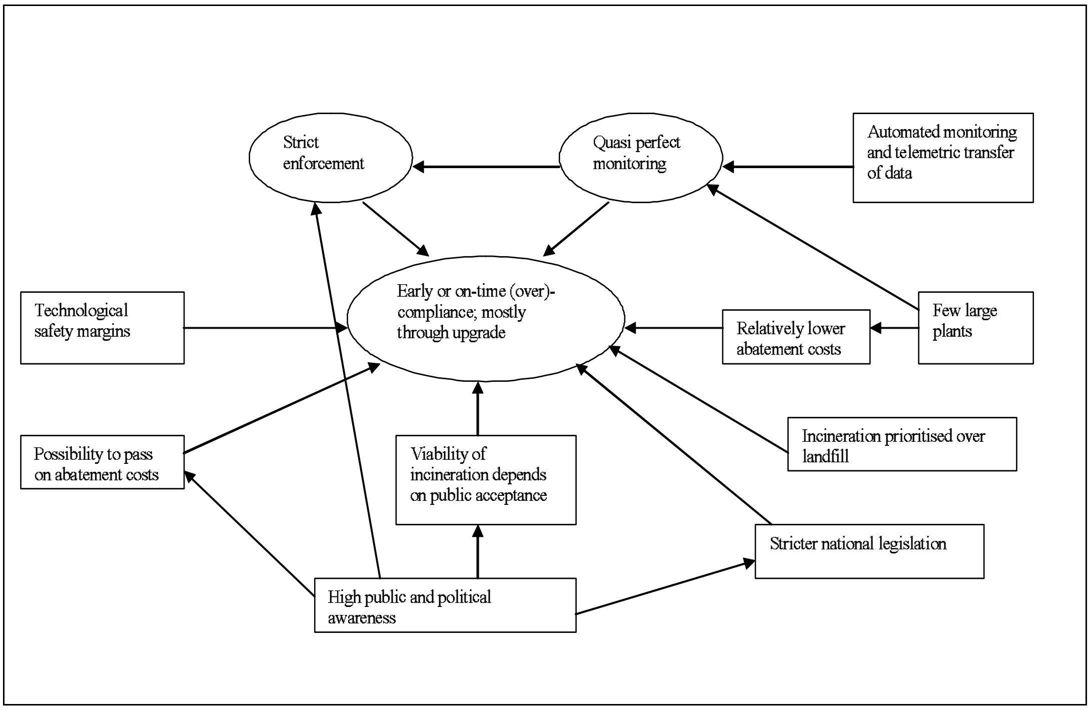
The factors stated so far explain why it was made impossible for operators not to comply with the requirements defined in the national regulation and, therefore, in the EU Directive. The fact that the plants, on average, over-complied with the stricter domestic regulation, however, is explained by further technical and political factors.

There are two technical factors that contributed to the incinerator's low emissions. Firstly, the refuse burnt in municipal waste incinerators is not homogenous and thus makes the emission values of MWIs fluctuating. In order to make sure that emission limits are always met (i.e. that peak emissions do not exceed their limits), operators needed a safety margin and on average were below the official limits. Furthermore, because suppliers of abatement equipment had to guarantee MWI operators that their equipment met certain emission limit values, they usually added another safety margin, and the limits they guaranteed were slightly tighter than those they aimed to attain. Secondly, the abatement technology introduced to control dioxin emissions in order to meet the limit set in the German ordinance also further reduces emissions of many other pollutants. While the first factor can be assumed to be true for complying plants in all four countries, the second applies only to Germany and the Netherlands, and to a lesser extent the UK, which has a weaker national limit for dioxins (see chapter 6).

Next to such technical factors, the high public awareness of environmental and health hazards related to municipal waste incineration can be assumed to be a further factor that contributed to (over-)compliance. Not only was the implementation of stricter domestic legislation in Germany clearly driven by the high degree of public and political awareness of the environmental and health impacts of incineration, particularly those associated with dioxins. This high degree of awareness also meant that (over-)compliance was a matter of self-interest for MWI operators who feared for the continued public and political acceptance of incineration as a waste treatment alternative and wished to lessen the citizens' and environmental groups' pressure.

The key driving factors behind the German implementation path are presented in Figure 7.2.

Figure 7. 2: Key explanatory variables of the German compliance path



4 The Netherlands

The Dutch case in many aspects resembles the German one. In the Netherlands, compliance with the Directive was reached well in time and municipal waste incinerators on average over-complied with the Directive as well as with the stricter domestic transposition. But in contrast to NRW, with 8 out of 13 plants the majority of 'existing' municipal waste incinerators was closed or replaced, while only 5 'existing' plants were upgraded. As will be shown, driving factors behind the Dutch compliance case are a high public and political awareness with respect to emissions from municipal waste incineration, quasi perfect monitoring and the possibility to pass on costs related to abatement investment to households. A specific factor of the Dutch case is the creation of an interest community between the government, plant operators and their sector organisation, which resulted in compliance decisions being co-ordinated across the country. Early and over-compliance were driven by reasons similar to those in Germany: a stricter domestic regulation requiring incinerators to comply almost 2 years before the deadline defined in the Directive and technical and political factors which favoured security margins in the abatement equipment's efficiency to reduce emissions.

4.1 Increasing public and political concern over emissions from municipal waste incineration furthers strict legislation

Also in the Netherlands, air emissions from waste incineration were a public and political concern long before the EU Directive was enacted. In 1979 the dioxin issue was addressed by the Dutch parliament after research had indicated considerable emissions from dioxins and heavy metals. Six years later emission limits were imposed on 'new' incinerators for a number of pollutants under the 1985 Guideline Combustion. As the government feared that high retrofitting costs for 'existing' plants would increase the costs of living too much, the decision to require emission limits for existing plants in operation permits was left to the discretion of the Provinces (regional level) (see Box 7.4 for a short presentation of the organisation of municipal waste management in the Netherlands).

But the Guideline failed to lessen public concern over the hazards of emissions from waste incinerators. On the contrary, public concern increased, reaching a peak in 1989 when dioxins were found in dairy products produced from the milk of cows grazed near to a waste incinerator in the Lickebaertpolder. The dioxin contamination was so high that the milk and dairy products were immediately removed from sale. As a result of this incident, and related to the fact that Germany and Sweden had issued stricter emission limits, the Dutch government hastened the introduction of stricter regulations to control emissions from waste incinerators, issued as the 1989 Guideline Combustion. This was also related to the government preference for incineration over (the also problematic alternative) landfill, stated in the National Environmental Policy Plan (NEPP) and eventually laid down in the Environmental Management Act of March 1993. This preference resulted from measurements undertaken at the end of the 1980s, which had shown that large areas and groundwater were polluted by landfills. Landfilling was thus abandoned as a preferable waste treatment option. Although, since the First Dutch NEPP published in 1989, the emphasis had been on prevention and recycling of waste, national policy aimed at an increase in incineration capacity in order to be able to decrease landfill capacity. To increase incineration capacity, the Dutch government sought to make waste incineration more acceptable through strict regulation. With increasing public environmental concern, expressed by public action, media coverage and administrative delaying tactics, also the perception of costs had changed. The willingness to pay for the environment increased.

Box 7. 4: The organisation of municipal waste management in the Netherlands

Also the Dutch waste markets are co-ordinated by public authorities. During the period studied here, in principle, each *Province* managed the collection and disposal of its waste, drawing up a waste plan estimating the amount of waste to be treated and outlining the disposal methods to be used. The decision about waste treatment facilities was thus with the Provinces. The Provinces' waste plans had to take into account the central government's preference for waste incineration over landfill. In the Netherlands, major decisions with respect to municipal waste treatment are therefore taken by the regional level and this in line with national environmental policy. The regions are furthermore supervised by the national level.

Municipalities are responsible for municipal waste collection and disposal. Both waste collection and disposal are financed through a waste tax raised by the municipalities. Tax revenues are used to pay waste incinerators and other waste disposal plants and to cover the municipalities' waste collection costs. As in Germany, the general responsibility of the municipalities for municipal waste collection and disposal does not imply that treatment plants are necessarily public plants. In the Netherlands, waste incinerators have been run for years as limited liability companies. But only very recently, ownership of some plants has been really privatised. While in the early 1990s governments almost exclusively owned the plants, the configuration of shareholders is now different for every incinerator. However, in most cases Provincial and/or Municipal public authorities still hold majority shares. The plant management autonomously takes day-to-day decisions, while a management board is responsible for important decisions, e.g. about investment in expensive pollution abatement equipment. Shareholders, and therefore the public authorities, are represented on the management board. Irrespective of the composition of its shareholders, the limited liability company is held responsible for its environmental impacts and compliance with environmental regulations.

4.2 Search for political acceptance of incineration and possibility to pass on costs create and interest community

The public concern, together with the government preference for incineration over landfill resulting in the need for additional incineration capacity and the fact that governments were involved in waste incineration plants, led to the interests of the government and the incineration sector becoming interwoven. This created an interest community between public authorities (at national and provincial level) and the waste incineration sector which is co-ordinated by the sector organisation 'Association of Dutch Waste Incinerators' (VEABRIN, 'Vereniging van Exploitanten van

Afvalverbrandingsinstallaties in Nederland')[80], which has been their interest organisation since 1985 (Eberg, 1997). This sector association wanted to protect and expand its markets. Therefore, both public authorities and plant operators had an interest to enhance public acceptance of waste incineration and, thus, to reach compliance with emission limits perceived as sufficiently strict.

This was further facilitated by the fact that, as in Germany, the plants' status as regional monopolies together with the high public concern over emissions allowed plant owners to pass on abatement costs to households. To cover abatement costs, plant operators were able to negotiate higher payments from the authorities, financed through the waste tax. This can be assumed to also have increased the operators' acceptance of strict regulation.

The Guideline Combustion 1989[81] was issued after an intensive consultation phase. While not opposing to stricter emission limits in general, operators first doubted that the suggested limits, in particular those for dioxins, dust, SO_2, heavy metals and NO_x, were accomplishable. Discussions were rather about technological feasibility of abatement and the exact timing of retrofit than about the stricter regulation in general. A study and advisory group was formed, the 'Stuurgroep Uitvoering RV '89', involving representatives of the Environmental Ministry, the Provinces and the incineration sector. Together they assessed and co-ordinated the implementation of the 1989 Combustion Guideline. This group also dealt with questions about which incinerators were to be retrofitted, which were to be replaced and where new incineration locations were to be set up. Therefore, contrary to other countries, such as France, there was a country-wide co-ordination of the compliance strategy of 'existing' municipal waste incinerators in the Netherlands. This is likely to explain the closure primarily of the relatively older plants (cf. chapter 6).

4.3 Monitoring is quasi perfect and non-compliance informally resolved

In the Netherlands, as in North Rhine-Westphalia, monitoring of incinerators appears as quasi perfect, so that undetected non-compliance is basically impossible (see Box 7.5 for a short presentation of institutional responsibilities with respect to the implementation, monitoring and enforcement of environmental policy addressed at emissions from municipal waste incinerators). Inspections were performed frequently (often monthly) and include emission data, composition and storage of waste, functioning of abatement equipment, etc. It can be estimated that more than 200 hours per year are spent on monitoring and enforcement of a municipal waste incinerator (this, however, refers to all aspects, not only to air emissions). Monitoring is clearly facilitated by the low number of large plants: in the early 1990s, the Dutch incineration sector comprised 13 'existing' plants, all of which had a capacity greater than 6 t/h.

The work of the Province's supervising civil servants is supported by self-controls of the incineration plants. Furthermore, owing to the 1989 Combustion guideline and the EU Directive, the requirements in operation permits had to be revised and the plants' measuring system is part of the licensing procedure. Here, just as in NRW, incineration plants are equipped with sensors and computers automatically recording and processing emission data on a number of substances. The plant's measuring system is checked by supervising civil servants when they inspect the plants. Those pollutants that only require periodical monitoring are measured regularly.

When in1995 a comprehensive inventory of the waste incineration sector was carried out for the Waste Board of the Environmental Ministry (cf. chapter 6), 206 of the 212 controls taken were within the limits set by the Dutch regulation and none exceeded the EU Directive's limits. The Provincial authorities knew about the six breaches of the Dutch

[80] Nowadays VVAV, 'Waste Processing Association' - 'Vereniging van Afvalverwerkers'.

[81] The guideline eventually withdrawn on 7 January 1993 and replaced by the Air Pollution from Waste Incineration Decree, largely identical to the Guideline Combustion 1989.

emission limits and were working with the plants to solve the problems. There was one exception: for the incinerator Gevudo Dordrecht the Dutch authorities tolerated the failure to meet the new Dutch regulations on 1 January 1995 and granted it an extension until 1 December 1996. Short-term breaches of emission limits were sometimes due to malfunction of technology. However, these were resolved informally and until the date this case study was completed, formal sanctions had not been employed against any municipal waste incinerators.

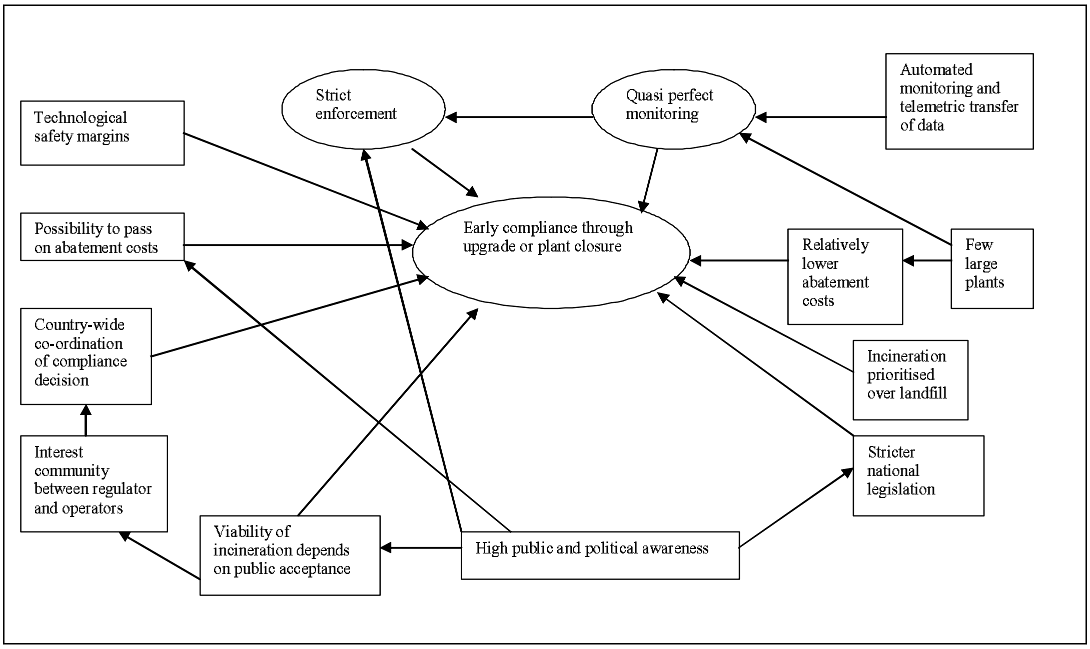
Box 7. 5: A multi-level monitoring and enforcement system

In the Netherlands it was the Ministry of Housing, Spatial Planning and Environment that issued the 1993 Air Pollution from Waste Incineration Decree serving as transposition of the EU Directive. The Ministry was also formally involved in implementation tasks. For the implementation of a regulation, the Environmental Ministry generally appoints civil servants to perform monitoring tasks, many of which are situated in the Inspectorate of the Environmental Hygiene, which is part of the Environmental Ministry, and pursues advisory and monitoring tasks. The Inspectorate works towards provinces, the level in between the central government and the local level, as well as towards municipalities, and only indirectly towards companies, thus controlling the quality of permits as well as of the system of monitoring and enforcement of the provinces. The Inspectorate does not issue sanctions itself, but can force provinces and municipalities to act, for instance, by employing administrative procedures instructing a province to renew or withdraw an operation permit or to impose a financial sanction. As a second institution, the Ministry of the Environment's Waste Board also participates in monitoring. It monitors the waste incineration sector as a whole with respect to the implementation of national waste and air quality policy.

In principal, the 12 Dutch Provinces are responsible for permitting tasks as well as for monitoring and enforcement towards municipal waste incinerators. Each provincial government, amongst other things, sets up a commission out of its members (the 'Gedeputeerde Staten'), which not only pursues the daily management of the provinces and reports to the Provincial government, but also appoints supervising civil servants with the task of monitoring and, if necessary, enforcing compliance with environmental regulation. However, when formal sanctions are required it is the Provincial government that takes the lead. In the Netherlands, therefore, the central and regional levels, rather than individual municipalities, play the major role in monitoring and enforcement.

The key driving factors behind the Dutch implementation path are presented in Figure 7.3.

Figure 7. 3: Key explanatory variables of the Dutch compliance path



5 The United Kingdom (England and Wales)

While the UK met the Directive's deadline for compliance, it followed a distinct compliance path, closing down 33 out of 37 'existing' plants and with this the quasi totality of its 'existing' municipal waste incinerator plant park. As will be shown in this section, next to cost factors and the age of plants, various policy interactions were the major driving forces behind this compliance strategy.

5.1 Outdated state of plants would have implied huge retrofitting costs

The UK incinerators in operation at the time the EU Directive was adopted were generally older generation mass burn plants, built in the 1960 and 1970s, with only basic pollution abatement in the form of electrostatic precipitators or cyclones. Furthermore, they were built by UK furnace and boiler-making firms with little specialist knowledge of incinerator technology and were, consequently, poorly designed. One important driving factor behind the closure of the majority of UK municipal waste incinerators is therefore the poor standards of these plants whose upgrading to the domestic and European environmental requirements would have involved high costs. Two factors explain the poor standards of the UK MWI plant park.

Firstly, unlike many mainland European countries, municipal waste incineration historically played a relatively minor role in the UK, with only around 7% to 10% of the total waste arising burnt in incinerators in the early 1990s. The fact that landfill sites were fairly abundant and landfill costs comparatively low explains the relative importance of landfill. Where incineration was used, it was principally in urban areas, which had limited access to landfill sites (without incurring significant transport costs). Furthermore, the UK's historical abundance of fossil fuels had reduced incentives for energy recovery from waste incineration. In fact, only four 'existing' incinerators operated with energy-from-waste recovery.

Secondly, the poor standard of municipal waste incinerators in the UK is linked to the fact that throughout the 1970s and 1980s waste policy was poorly articulated and implemented in the UK, and up to the mid-1990s there existed no coherent national waste policy or strategy. Under the Control of Pollution Act 1974 the management and disposal of municipal waste was the responsibility of the local Waste Disposal Authority, usually the Metropolitan or County Council, for the relevant area. The Waste Disposal Authority was free to choose the most economical disposal option available. Within central government, the Department of the Environment (DOE)[82] was responsible for environmental protection issues, and responsibility for negotiation and subsequent implementation of the EU Directive, therefore, rested with the DOE. But prior to the mid-1990s the DOE played little role in waste planning and it restricted its role to the provision of information to Waste Disposal Authorities on different disposal options (House of Lords, 1989). It was only in 1995 that the DOE published a national waste strategy, which established a 40% recovery target for municipal waste by 2005 and which classified incineration as waste recovery alongside recycling and composting.

Concluding, it is worth mentioning that a 1989 survey suggested that 12 of the 37 UK plants would have reached the end of their operating lives by 1994 in any case. As will be demonstrated below, the exact compliance path, however, was driven by further policy interactions as outlined in the following two sections. These led to a reconsideration of how many plants would stay in operation, so that by 1992 a survey of MWI operators undertaken by Environmental Data Services Ltd. predicted that only 5 'existing' UK incinerators would continue in operation after the December 1996 deadline. This estimation eventually proved almost correct.

[82] In 1997 the DOE merged with the Department of Transport to form the Department for the Environment, Transport and the Regions.

5.2 Slow reform of the regulatory system of industrial pollution leads to late transposition

As long ago as 1976, the Royal Commission on Environmental Pollution had responded to criticism of the existing regulatory system of industrial pollution in the UK and called for a more effective and transparent integrated approach, but it lacked the political support for implementation of its proposals. It was only in the mid-1980s that a number of factors, such as increased public awareness, tensions between the formal requirements of EC legislation and the traditional flexible British approach and internal conflicts within the British government over the control of the pollution Inspectorate led to the issue being reopened. A review of industrial pollution and safety ordered by the Cabinet Office in 1985 and published in 1986 recommended the creation of an integrated pollution Inspectorate, and eventually led to the creation of Her Majesty's Inspectorate of Pollution (HMIP) in 1987, under control of the Department of the Environment (DOE). The DOE, however, was still faced with the task of reconciling the British regulatory approach with the EU legislation system.

A DOE consultation paper of 1986 proposed to bring municipal waste incinerators, amongst other industrial processes, under the control of HMIP. The DOE also developed a proposition of a Clean Air Bill, but failed to obtain parliamentary time for this proposal in the 1988/1989 parliamentary sessions. To satisfy EC pressure for implementation, it was forced to issue 'stop-gap' regulations in 1989. This was the 'Health and Safety (Emissions into the Atmosphere) (Amendment) Regulations 1989' which brought all incineration processes with a capacity greater than 1 t/h under HMIP control (cf. chapter 5). In 1990, eventually, the adoption of the Environmental Protection Act introduced a more integrated and centralised approach to the control of industrial pollution in the UK, thus radically overhauling the prior regulatory system. It also provided the legislative framework for implementation of the EU Directive, formally reached by issuing the Municipal Waste Incineration Directions in November 1991.

Unlike in Germany and the Netherlands, there is no evidence that UK environmental NGOs played an influential role in the development and implementation of the Municipal Waste Incineration Directions. Although dioxins have been an issue of concern for environmental and public health policy-makers in the UK since at least the mid-1980s, the UK government has consistently taken a more relaxed view of the risks associated with dioxins than the Dutch and German governments. The UK's 'first past the post' electoral system has meant that the Green Party has achieved little representation at either local or national levels. However, over the last decade environmental NGOs and 'grass roots' citizens' organisations campaigning against waste incinerators have attracted increasing public support and political influence. The hazards associated with dioxin emissions from incinerators now feature as a prominent public concern. This is reflected by the dioxin emission standard defined in the UK transposition of the EU Directive.

5.3 Restrictions on local government expenditure render the financing of abatement equipment largely unfeasible

Within central government, the Department of the Environment was not only responsible for environmental protection issues but also for local government finance and with this for the implementation of related government policy, as formulated by the UK Treasury. The objective of this policy was to restrain local government expenditure. Throughout the 1980s and most of the 1990s the DOE consequently maintained tight control over local government expenditure, in line with Treasury policy to reduce the UK's public sector borrowing requirement.

At this time Metropolitan Councils were not only responsible for the collection and disposal of municipal waste, they also owned and operated the waste incinerators and were therefore responsible for pollution abatement installed in these plants.[83] Waste management, including pollution abatement at incinerators, had to be financed from the Council's general budget, made up of a combination of local taxation and an annual block grant from the central government, administered by the Department of the Environment. As a consequence of the DOE's tight control over local government expenditure, Councils had limited discretion over local taxation and were effectively required to set budgets within limits specified by central government. Moreover, the Department of the Environment refused to subsidise the up-grading costs associated with the EU Directive, subsidies the Association of Metropolitan Authorities had lobbied for.

In parallel, in 1987, Her Majesty's Inspectorate of Pollution (HMIP) was established as a Directorate of the DOE to provide an integrated national pollution regulator[84]. After the Inspectorate, in 1989, had assumed responsibility for the regulation (authorisation, monitoring and enforcement) of all incinerators with a capacity greater than 1 t/h (cf. chapter 6), regulatory responsibility was not only significantly tightened, but also passed from local authorities to a centralised national pollution regulator. It was therefore the Inspectorate that undertook the practical implementation of the Directive. Having lost regulatory responsibility for incineration processes, Councils operating incinerators were in a weak position to resist implementation of the Directive. They had little influence over the pollution Inspectorate, which required them to apply for authorisations for their plants by December 1992. These authorisations not only set out detailed monitoring and reporting requirements, they also required that plants be upgraded to BATNEEC standards by December 1996 or closed. Failure to comply with these requirements would have made them liable to formal legal sanctions under the 1990 Environmental Protection Act.

In sum, given that the Department of the Environment refused to subsidise the incinerators‘ upgrading costs and at the same time placed tight constraints on local authority expenditure and taxation, only a few Councils could afford the investment required to upgrade their plants. 21 more plants than originally expected closed as a result of the co-implementation of both the EU Directive and the national emission limits.

5.4 Further policy interactions explain the selective upgrade of plants

Policy interactions -or rather the absence of a coherent waste policy- also explain the relative ease with which numerous 'existing' municipal waste incinerators were closed down. Unlike France, where in particular small municipalities were also subject to problems of financing the necessary abatement investment to meet the Directive's requirements, UK Councils had a relatively cheaper alternative to comply with the Directive: landfill. An estimation of the Association of County Councils indicated that in 1987/88 the average operational costs per tonne of waste were £2.25 for landfill as against £12 for incineration. The relative abundance and low cost of landfill, coupled in the early 1990s with the absence of national policy to promote more sustainable alternatives, therefore meant that all but four of the UK's existing incinerators were closed.

But also the specific upgrade of four incinerators was driven by policy interactions. Since the end of the 1980s UK electricity policy, established under the Department of Trade and Industry's (DTI) responsibility for promoting renewable energy, had a significant impact on the development of municipal waste incineration in the UK. The Electricity Act 1989 required, by order, public electricity suppliers to purchase specified capacity generated

[83] More recently, a number of forms of public-private-sector partnership arrangements have been adopted. These are, for example, joint ventures, contracting-out of services to private firms, and the Private Finance Initiative, bringing private-sector capital and expertise into the UK waste sector, and financing the development of a number of new large waste-to-energy plants.

[84] HMIP was subsumed within the Environment Agency with its establishment as a fully independent Agency on 1 April 1996.

from non-fossil fuel sources, including energy from waste, on long-term contracts in order to support renewable energy and subsidise nuclear power. This was known as the Non-Fossil Fuel Obligation (NFFO). Electricity companies must pay a premium price for electricity generated under the NFFO, determined by a competitive bidding process involving potential generators. As a result, the NFFO significantly improved the financial viability of electricity generation from MWI. Largely because of this opportunity to sell electricity on long-term contracts at premium prices to electricity supply companies, those 4 existing plants equipped to recover energy were upgraded and continued in operation.

Taking all the previous arguments together, the closure of all but four 'existing' incinerators was driven by their relative age and poor technical standard, the fact that incineration played only a minor role because of the relative abundance and low cost of landfill, and by interaction with Treasury, while interaction with energy policy explains the upgrading of the four waste-to-energy plants. Policy interactions hence made it possible to avoid otherwise high retrofitting costs and facilitated waste-to-energy plant retrofit.

5.5 Non-compliance is penalised

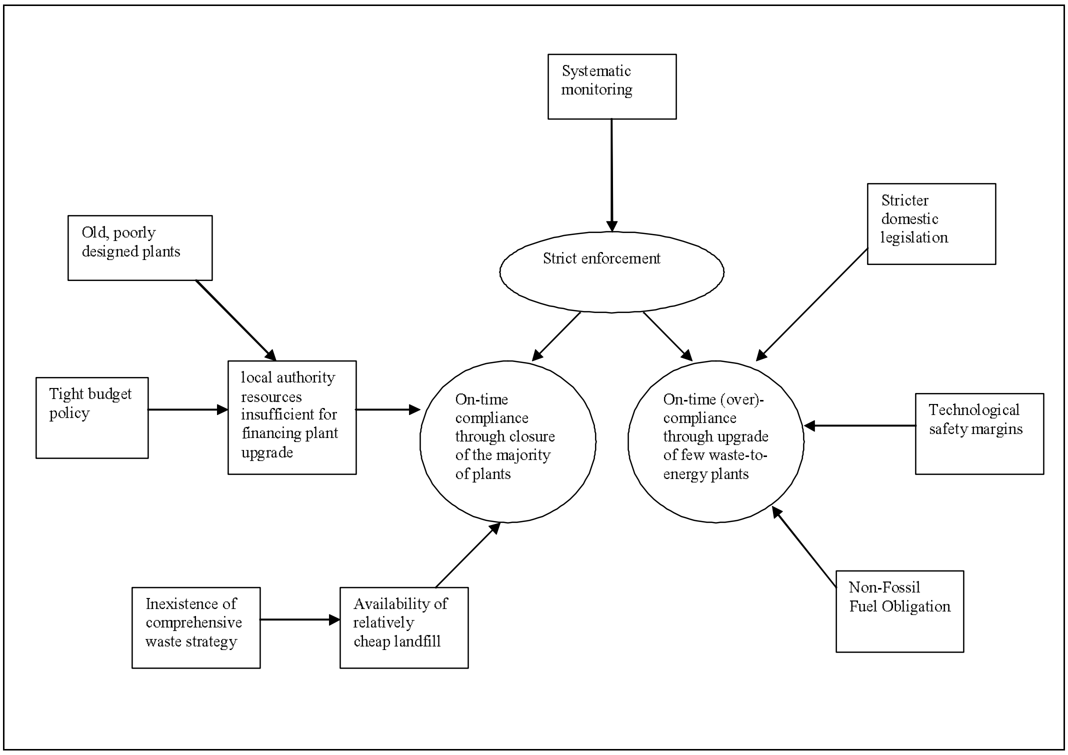
Finally, frequent monitoring and the application of enforcement measures forced the UK incinerators into on time compliance with the Directive although not all plants managed to timely install their abatement equipment. In fact, the Environmental Agency's annual inspection programme for a typical incinerator consists of four to six planned visits and up to six additional visits to investigate complaints and operational problems. The number of visits can however significantly rise where major problems arise, especially where public interest is generated. Moreover, as in the other countries, operators in the UK are required to self-monitor their emissions and to report any breaches of their authorisations to the Environmental Agency, HMIP's successor. Breaches of an authorisation condition render an operator liable to prosecution under the 1990 Environmental Protection Act: as maximum penalty a £ 20,000 fine on summary conviction, or an unlimited fine and up to two years imprisonment on conviction in a higher court. Failure to comply with a court order could result in action for contempt of court, rendering the operator liable to sequestration of assets and an unlimited term of imprisonment. However, the regulator has considerable discretion in deciding whether to prosecute a particular case.

Monitoring by plant operators showed indeed that breaches have occurred at both 'new' and 'existing' plants. The Environment Agency has used both informal (negotiation) and formal sanctions (enforcement notices and prosecutions) to solve these problems, requiring plant operators to ensure better mixing of the refuse burnt, and undertake improvements to the acid gas abatement equipment. According to interviews with the Environmental Agency undertaken at the end of the 1990s, for the majority of plants these problems were largely solved. As stated in the previous chapter, in one upgraded plant problems have continued throughout the 1990s and the plant operator -the Sheffield Council- in 1999, was successfully prosecuted under the Environmental Protection Act and fined £ 18,000, close to the maximum fine available under the Act for summary conviction in magistrates court. This example shows that the UK enforcer did not tolerate sustained non-compliance.

Apart from this example, UK plants can be assumed to be on average compliant with domestic regulation. Just as in the case of Germany and the Netherlands, technological safety margins and the -additionally to the Directive- established dioxin emission limit value explain why, on average, UK incinerators over-comply with national and the EU Directive's standards. The stricter domestic emission standards further add to over-compliance with the Directive.

The key driving factors behind the UK's implementation path are summarised in Figure 7.4.

Figure 7. 4: Key explanatory variables of the UK compliance path



6 Cross country comparison

The aim of this chapter was to identify the key factors influencing the implementation of the 1989 Municipal Waste Incineration Directive that explain the distinct compliance paths of the countries studied: (over-)compliance through retrofit or replacement of plants in Germany and the Netherlands; (over-)compliance through the closure of the majority of plants and the upgrade of few plants in the UK; and delayed compliance in France. Furthermore, we wanted to know whether there are indications that implementation decisions were driven by considerations of cost-effectiveness, aimed at a reduction of inefficiencies included in the Directive.

As an interesting and important result, the research points to the impact of multiple policy interactions on implementation outcomes. Interactions with both environmental and non-environmental policies, and associated regulatory uncertainty and anticipation, played an important role in the outcomes observed. In part, these factors were important as they affected compliance costs. They explain the strikingly different compliance paths taken by France and the UK - the two countries where plants faced significant upgrading costs to meet the Directive's standards. In the UK the absence, until recently, of a coherent national waste policy contributed to the availability of landfill as an alternative to incineration, whilst policy to promote renewable energy (the NFFO) facilitated the upgrading of those plants with energy recovery. In France, interactions with the national (1992) waste law closed off landfill as an alternative for domestic waste, whilst promoting incineration with energy recovery (in both Germany and the Netherlands a policy presumption against landfill also favoured investment in incineration). Hence in the UK, unlike in France, it was possible to reduce compliance costs by transferring waste to landfill. Indeed, the availability of an alternative low-cost waste treatment option together with limited resources available to upgrade plant at the local level can be assumed to be the major driver behind the UK's particular compliance path. And in the French case, regulatory anticipation and uncertainty contributed to the pattern of delayed compliance for large incinerators; an implementation path tolerated for several years by enforcement agencies and the public.

Compliance cost considerations, therefore, clearly drove compliance decisions in France and the UK. But while in the UK the major cost-saving potential -transferring waste to landfill- was not in opposition to the Directive's requirements, an increase of cost-effectiveness in France relative to the Directive's requirements was only achieved by introducing unforeseen discretion. In the other two countries the question of whether implementation increased or decreased the Directive's inefficiencies is less applicable as the national implementation processes were much less shaped by the European policy.

As a further and expected result, we found that monitoring and enforcement were generally in line with the overall compliance outcomes observed and have clearly been an important factor in reaching environmental effectiveness. However, being endogenous variables, they were not sufficient to explain the outcomes in isolation, but were shown to be influenced by the degree of autonomy and scope of regulatory authorities, and by public and political awareness of environmental and health risks.

Organisational administrative structures did have an impact on the level of enforcement in France where delayed compliance was partly made possible by non-specialised local regulators and enforcers, trading off environmental protection against local economic concerns. Furthermore, while the involvement of local governments in the financing, management and regulation of waste disposal in all four countries introduces the potential for conflicts of interest, particularly where local authorities own disposal facilities, in practice this has only led to implementation difficulties in France. In France, the mayors, as both head of the local government and owner of incineration plants, were frequently able to exert pressure on the prefects to delay enforcement. In the other three countries the institutions responsible for, or overseeing, monitoring and enforcement were specialist environmental bodies, which, moreover, seem to have enjoyed greater independence from local political pressure. One can assume that these could use their relatively greater autonomy to favour the environment and that, in general, independent specialist institutions may enforce policies more strictly and thus contribute to successful implementation.

A high degree of environmental awareness clearly had a positive impact on enforcement and operators' compliance decisions in Germany and the Netherlands. In these countries, the implementation of stricter domestic legislation was clearly driven by the high degree of public and political awareness of the environmental and health impacts of incineration, particularly those associated with dioxins. Furthermore, this high degree of environmental awareness meant that (over-)compliance was a matter of self-interest for MWI operators who feared for the continued public and political acceptance of incineration as a waste treatment alternative. In these cases environmental goal attainment was also facilitated by the combination of monopolies funded through local fees or taxes and high public awareness (assuming this is positively correlated with individual households' willingness to pay for abatement measures). It encouraged German and Dutch plant operators to reduce their emissions below their legal limits as it enabled them to pass on their abatement costs. This argument is also consistent with the pattern of late enforcement and compliance found in France, where low public and political awareness of these environmental and health hazards was correlated with the absence of stricter emission legislation and an assumed low willingness to pay for abatement measures. Here, incinerators -despite also being local monopolies- could not easily pass on their investment costs, which was particularly important as France found itself in a relatively disadvantageous position with respect to compliance costs. Only after dioxin pollution became a media issue in France and public and political awareness rose were enforcement measures taken. This also underlines the important role the disclosure of information can play to support compliance.

7 The findings in the light of the political science based explanations of effective implementation

Putting these results into perspective with suggestions made by the political science based 'fit' approach one can notice that a comparison between the pre-existing general policy approach and the Directive's requirements does not allow us to explain the implementation outcomes across all countries studied here. Germany, the Netherlands, and to a lesser extent France, had formulated an emission standard based policy before the EU Directive was adopted, whereas this was not the case in the UK. Furthermore, the UK's IPC approach differs from the MWI Directive's media emission limit value-based approach. Despite the fact that the adaptation pressure should thus have been higher in the UK due to policy misfit, this country implemented the Directive quite effectively, and more effectively than France. Also the suggestion that a misfit with respect to style and structure should be more decisive for implementation effectiveness than policy contents can not be supported by this case study: it does not hold for the UK where practical implementation was rather effective despite the before-mentioned misfit in the policy approach. And it does not hold for France either, which is the only country amongst the four studied which encountered continuous problems with the implementation of the Directive, despite the 'misfit' primarily relating to the strictness of limit values and hence to the policy contents and not style and structure. From an economic point of view this is not astonishing, given that it was the policy contents which implied high compliance costs in France. The role various policy interactions played for the outcomes of the Directive's implementation, furthermore, stresses the importance of not restricting the view to purely administrative fit aspects but of taking into account the wider, also non-environmental, context in which a policy is implemented. Taking all these findings together, the empirical study makes a point for applying a broad approach when studying implementation processes.

The results are more in line with Haigh (1997/98), who, contrary to the 'fit model', suggests that empirically, Member States, which have long carried out practices required by a Directive, may have more difficulty in implementing the policy correctly than other Member States, which start freshly on the subject. He explained this by the fact that a country, having already developed its traditions, may not be convinced of the necessity to make changes to existing practices in order to implement fully a legislation. Evidence for this is partly found in this study. In France, legislation of the same type but with slightly less strict emission limit values (French ministerial order of 1986) existed before the adoption of the Directive. Interviews undertaken in this country suggested that the public authorities tolerated for a considerable time the application of abatement technology meeting the less strict emission limit values for those plants that had been upgraded under the 1986 legislation. Contrary to France, the UK, starting freshly with regulation of atmospheric emissions from municipal waste incinerators, enforced emission limit values that met the Directive's requirements. Like France, Germany and the Netherlands also had domestic legislation of the same tradition at the time the Directive came into force, but in these countries the domestic standards were clearly stricter than the EU policy's. The problem of not making necessary changes to the domestic standards, therefore, did not occur. However, these countries did encounter problems in the transposition of the Directive, but only with respect to the choice of the formally correct policy instrument.