Institute for Environmental Strategies

Economic evaluation of dust abatement techniques in the European Cement Industry

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1 Background and Scope of this Study

To an increasing extent, cement kilns are burning waste as a secondary fuel. Until today, they are not covered by European legislation which has set standards for pollution to air from plants for municipal waste incineration, in particular Directives 89/429/EEC and 89/369/EEC. In October 1998, the Commission has adopted Proposal COM (1998) 558 on the incineration of waste. A part of the proposal is the extension of the scope of these Directives to cover cement kilns that burn waste as a fuel, and in particular to set standards for dust emissions from such plants.

It is the objective of this study to consider the costs and benefits of extending the scope of the Incineration Directive to specify emission limit values from cement kilns burning waste as fuel.

Starting from the present situation of actual dust emission levels from existing cement kilns, the various technical options for a reduction of dust emissions are to be presented and discussed. These technical alternatives shall be examined for

- the emission reduction that can be achieved by such techniques and
- the investment and operational costs that these techniques will eventually cause for the operator of the cement kiln.

At the macro-economic level, the costs of the dust abatement measures are to be compared to the damage costs of unreduced dust emissions from cement kilns burning waste as fuel.

For this cost-benefit-analysis at the macro-economic level, the results from the methodology developed by ETSU (1996) for their examination of the costs and benefits of the proposed new emission limits for municipal waste incineration plants, are taken as a basis. Additionally, more recent work on the subject by the same and other authors is evaluated in order to check what uncertainties maybe associated with the assumed damage costs, and to discuss whether the characteristics of different types of dust can be taken into consideration in the assessment of damage costs.

On an individual plant level, the costs for dust reduction are compared to the economic advantage that is given for the operator of a cement kiln when he uses waste as a secondary fuel

2 Fine particulates as an air pollutant of prime concern

The term particulates refers to a large variety of small particles which can originate both from natural sources (e.g. sea spray, airborne natural dust, plant pollen) and from anthropogenic sources (fossil fuel burning, motorized traffic, industrial activities, etc.).

A general distinction is made between

- primary particulates which are generated during burning of fuels and by all sorts of industrial and human activities that can lead to material abrasion and
- secondary particulates which are formed by atmospheric reaction of other components such as sulphur dioxide (SO2), nitrogen oxides (NO2), volatile organics and ammonia, which themselves have originally been emitted e.g. from traffic, industry, fuel burning or agriculture.

In the past, total suspended matter (TSP) was mostly the basis for setting national air quality standards in a number of countries, and TSP was also the most common parameter to describe the immission situation in a given region

Of prime concern for human health, however, are those fractions of TSP whose diameter is small enough so that they can enter into the pulmonary system. Today it is generally accepted that this is the case for particulates with an aerodynamic diameter of less than 10 µm, commonly addressed as the PM10 fraction of TSP.[1]

For Germany, e.g., it has been estimated that approximately 60 per cent of the total suspended matter (TSP) immissions fall into the PM10 category [TA Luft, 1998]. Other sources point out that PM10 typically makes up even 70-80% of TSP both in municipal and rural areas of Germany [e.g. LfU, 1998].

Health effects which are ascribed to small particulates include increased mortality as well as increased acute and chronic morbidity, such as e.g. respiratory infections, chronic pulmonary diseases, bronchitis, asthma, and childhood croup, and it is a wide-spread current convention to use PM10 as the most relevant index of ambient particulate concentrations [ETSU, 1996].

Because of the specific importance of particulates with a small diameter, new sampling methods have been developed that allow for a distinction between particulates of different sizes.[2] Nowadays, numerous immission measurements and a lot of epidemiological evaluations are available for the PM10 fraction.

Still smaller particulates with an aerodynamic diameter < 2,5 µm (PM2,5) or even smaller (e.g. PM1, PM0,1) are thought to be of the greatest significance for health since they penetrate more deeply into the respiratory tract (e.g. [STATISTICS NORWAY, 1998]). For these fractions, however, the analytical problems become even more complex, and many existing data have primarily orientating character [LfU, 1998] and should be taken with some care.

With decreasing diameter of particulates, the relative importance of secondary particulates increases over the contribution from primary particulates. Secondary particulates which are formed by chemical reaction between e.g. sulphates, nitrates and ammonia mostly have an aerodynamic diameter smaller than 2,5 µm. They can contribute as much as 50% of all PM2,5, while less than 20% of all PM10 are considered to consist of secondary particulates [TA Luft, 1998].

Concerning the relative contributions of particulates from the various sources, some first estimates have been made for specific regions in Europe:

For Switzerland, road traffic is assumed to contribute at least 50% of the total PM10 immission concentrations [BUWAL 1996].

For the South German Federal State of Baden-Württemberg, estimated contributions to the total anthropogenic dust emissions are as follows: 43% from traffic, 46% from industry and trade, and 11% from small combustion sources [LfU, 1998].

For the more heavily industrialised region of Northrhine-Westphalia, industrial point sources are estimated to contribute 61% of all anthropogenic PM10, while road traffic adds some 30%, and the remaining 9% originating from minor sources [BÖKER, 1999].

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[1] For a recent summary of the epidemiological literature see e.g. [LfU, 1998].

[2] Sampling techniques have a strong influence on the results of fractionated determination of particulates. Standardized sampling equipment is being developed in international cooperation that will allow comparison of national data from different countries.

As has been stated already, national air quality targets were normally dealing with TSP in general in the past. At the same time, it was generally assumed that below certain threshold levels, health effects of air pollution on a population would be negligible. However, more recent investigations show that threshold values cannot be generalised but rather are specific to individuals [STATISTICS NORWAY, 1998].
Consequently, the European Office of the World Health Organization have decided to no longer recommend guidelines for particulates [WHO, 1996]

Until recently, Directive 80/779/EEC has set the following limits to total suspended matter (depending on the analytical method used) for the European Union

Table 1 TSP limit values (Dir. 80/779/EEC)

Statistical value	Analytical method	
	Gravimetry	Black Smoke
Arithmetical mean of all daily averages / 1 year	150 µg/m³	
95-percentil of all daily averages / 1 year	300 µg/m³	
50-percentil of all daily averages / 1 year		80 µg/m³
50-percentil during winter (01.10.-31.03)		130 µg/m³
98-percentil of all daily averages / 1 year		250 µg/m³
Guideline arithm. mean of all daily averages / 1 year		40 - 60 µg/m³
Guideline arithm. mean of all measurements / 1 day		100 - 150 µg/m³

In contrast to these former limits and guidelines, the new Framework Directive on Air Quality 96/62/EEC is setting much stricter standards. At present, the Commission Proposal for a Daughter Directive is being discussed that will bring about the following changes:

According to the present state of discussion, in Stage 1 of the implementation a 24-hour-limit of 50 µg/m³ (with a certain number of allowed transgressions) and an annual limit of 40 µg/m³ will have to be met by 1 January 2005. In Stage 2 of implementation, it is intended to lower the annual limit to 20 µg/m³ by the year of 2010.

Meeting the targets of the new EU Daughter Directive will require significant efforts from the involved actors, as it is a widespread phenomenon that the proposed limit values are presently clearly exceeded in many regions of Europe (e.g. [BÖKER, 1999], [BUWAL, 1997], [ISRAËL et al., 1992], [LfU, 1998])

It will only be possible to improve the European PM10 immission situation by joint efforts at, among others, all relevant industrial point sources, as stack emissions of PM10 do hardly affect the local immission situation, but rather are transported over large distances [SCHNEIDER & KUHLMANN, 1996] and thus add to the general immission situation of a huge area.

As long range transport is more relevant for PM10 than for other pollutants such as NOx or SO2, PM10 immissions show a much more homogeneous distribution over wide areas[3], resulting in surpassed air quality limits even in rural areas (BUWAL, 1997). The Swiss limit value for the annual PM10 immission concentration of 20 µg/m³ (which is equivalent to the EU Stage 2

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[3] The effect is enhanced by formation of secondary particulates in some distance from the emission sources.

target) is safely met only in mountainous regions above 1000 m altitude (ibd).

Detailed studies of the immission situation in Berlin, for instance, have shown that the city itself contributes only about 20% to the total PM10 immissions, while approximately 80% of the municipal immission situation is caused by long range transport [ISRAËL et al., 1992]. By applying atmospheric trajectory models on the results of multielemental analyses, the same group of scientists found that PM10 immissions in a village near the North German city of Uelzen could be traced back as far as the metallurgical smelting industry in Lille (North France) and the oil refinieries in Le Havre and Rotterdam.

Long range transport also occurs with fine particulates from natural sources and may even span over several thousand kilometers, as transport of Atlantic seaspray and Sahara dust has been observed in Baden-Württemberg episodically [LfU, 1998].

3 Dust emissions from the European cement industry

In 1995, 252 cement plants with 437 kilns were operating in the European Union. Circa 78% of the total production capacity are working in dry processes, 16% in semi-dry or semi-wet processes, and 6% in wet processes [CEMBUREAU, 1997].

The typical capacity of a new kiln is 3.000 Mg clinker per day. Older kilns with a capacity of less than 500 Mg/d are scarce (ibd.), and are often not operating throughout the year

Traditionally the emission of dust, particularly from kiln stacks, has been the main environmental concern in relation to cement manufacture [IPTS, 1999]. It is beyond any doubt that the European cement industry has taken tremendous steps over the last few decades to reduce their dust emissions.

According to SCHNEIDER and KUHLMANN [1996], the specific dust emissions have decreased from typical values around 35 kg per metric tonne of product to levels below 0,1 kg/Mg. This is reflected in improved dust abatement techniques that have allowed to reduce the emissions from levels around several hundred milligrams per norm cubic meter down to levels between 5 and 15 mg/Nm³ [IPTS, 1999] at modern kilns.

However, this emission reduction does not necessarily mean that the risk potential which is linked to the dust emissions has decreased by the same order of magnitude, as many abatement measures in the past were mainly directed at an improved retention of coarse particles, while they were less effective with respect to fine particulates [BUWAL, 1996].

In spite of the improvements of the past, cement kilns are, besides other large industrial installations, still considered as one relevant source of PM10 particulate emissions [e.g. TA Luft, 1998]

Main sources of dust emissions from the cement industry are the kiln exhaust gas, raw mills, clinker coolers and cement mills, as in all these processes large volumes of gases are flowing through dusty materials [IPTS, 1999]. Depending on the specific flue gas system, several large emission sources can exist.[4]

The finely ground raw materials and fuels have a typical grain size of less than 10 µm. The same is the case for particulate emissions from cement grinding, packaging and loading operations [SCHNEIDER & KUHLMANN, 1996]. Also, it is generally assumed by regulating authorities that 95% of the dust emitted with the stack gas has a diameter of less than 10 µm

In summary, it can be assumed that the majority of all dust emissions from cement kilns belong to the PM10 category.

As mainly the oven exhaust will be influenced when wastes are co-incinerated in cement kilns, the following sections will be focused on the dust emissions in oven exhaust gas.

Table 2 Emission limit values in EU Member States and Switzerland

Country	Limit value [mg dust/Nm3]	Reference / explanatory remark
Germany	50	daily average
Austria	50	daily / half hourly average
France	50 (100)	daily / monthly average for new kilns; (figure in brackets for clinker cooler)
	<150	(daily / monthly average) for old kilns after 2001
Belgium	50	new kilns
	50 - 150	old kilns
Greece	100	new kilns
	150	old kilns
Italy	50	
Portugal	50	new kilns
	100	old kilns
Spain	250	present limit for new or retrofitted kilns
	400	present limit for old kilns
	100	new limit (under discussion)
Sweden	50	90-day-average for new kilns
	150 - 500	old kilns
Netherlands	15	one kiln only
Luxembourg		one kiln only
Finland		two kilns only
Denmark		local permit (one kiln only)
Ireland		two kilns only
UK	50	daily average for wet exhaust gas
Switzerland	50	half hourly / daily average for new and old kilns; additionally, annual emission loads are limited at some sites
EU Proposal	30	daily average for cement kilns which burn waste as a fuel [EU Proposal COM (1998) 558].

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[4] Additional dust emissions can originate e.g. from outdoor storage piles, from storage of input materials, fuels or product in silos or bunkers, from mechanical and pneumatic transport devices, from product packaging, and from loading and unloading operations. In total, these minor sources often add up to one hundred or more diffuse sources on the site of one rotary kiln.

Emission limit values and actual emission levels

Until today, dust emissions from the European cement industry have been regulated by individual Member States (and other countries like e.g. Switzerland). Emission limit values for the kiln exhaust gas in the EU range from 15 mg/Nm3 up to 250 mg/Nm3. At some older installations, even dust emission concentrations up to 500 mg/Nm3 are tolerated. Commission Proposal COM (1998) 558 on the incineration of waste contains an emission limit value of 30 mg dust per Nm³ (measured as a daily average).

Table 2 presents a compilation of the national emission limit values of the EU Member States and Switzerland for dust in oven exhaust gas from cement kilns.

It is important to note that the national emission limit values have to be taken in conjunction with the corresponding evaluation method:

E.g. for Germany, the Technical Guideline for Clean Air Protection [TA Luft, 1986] regulates that an emission limit value is observed when all daily average values are below the limit, 97% of all half hourly average values are below 6/5 of the limit value, and all half hourly average values are lower than the 2-fold of the emission limit value (Table 3).[5].

Table 3 Emission limits according to the German TA Luft

Reference	Emission limit value
daily average	50 mg/m3
97% of all half hourly values	60 mg/m3
all half hourly values	100 mg/m3

According to the Swiss Clean Air Act, the same evaluation criteria are applicable for Switzerland (Art. 15, Abs. 4, Luftreinhalteverordnung).

For practical reasons, all emission limit values and suppliers guarantees discussed in this study are to be understood along the lines of the evaluation method of Table 3.

The actual average emission level of a cement kiln can significantly differ from the permit for a number of reasons. A European survey of the actual emission levels of all European cement kilns does not exist. According to [CEMBUREAU, 1997], the actual dust emissions of European cement kilns lie between 20 and 200 mg/Nm3.

As large variations exist between the flue gas systems of individual kilns and thus the associated exhaust gas volumina may vary widely, for a comparison between different kilns types it is useful to standardise the dust emissions with respect to kiln capacity. For such a standardisation, certain assumptions with respect to specific exhaust volumina etc. (as e. g. published recently by [UBA, 1998]) have to be made

Table 4 Specific exhaust volumina and dust concentrations in raw gas for various kiln systems and clinker coolers [UBA, 1998]

Emission source	**Specific exhaust volumina per capacity [Nm3/kg]**	**Dust concentration in raw gas [g/Nm3]**
Rotary kiln with cyclonic	2,1 -2,5	35 –150

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[5] Continuous monitoring is a fundamental prerequisite for such a statistical evaluation of dust emissions. At present, continuous monitoring of dust emissions is a standard practice in some but not all EU Member States.

preheater (exhaust gas used in raw material grinding)		(up to 900 before pre-treatment)
Rotary kiln with cyclonic preheater (exhaust gas used in raw material grinding) with H2O injection	1,7 - 2	20 - 50
Rotary kiln with grate preheater	1,8 -2,2	1,5 - 12
Dry rotary kiln (exhaust gas used in raw material grinding)	2,5 - 3,0	35 -150 (up to 900 before pre-treatment)
Dry rotary kiln with H2O injection (exhaust gas used in raw material grinding)	2,2 - 3,0	15 - 25
shaft kiln	2,0 - 2,8	2,0 - 7
clinker cooler	0,7 - 1,8	0,7 - 15

Detailed calculations of emission factors have been made for the Austrian cement industry by [HACKL & MAUSCHITZ, 1997]. Based on continuous dust measurements on all 12 Austrian kilns, they determined the emission factor for oven exhaust gas at 50,68 g dust per Mg clinker for the year of 1996. According to the same study, this corresponds to an average emission concentration of 21,83 milligrams of dust per norm cubic meter, with a minimum concentration of 3,70 mg/Nm3 and a maximum emission concentration of 47,16 mg/Nm3.

As the Austrian average emission concentration is close to the lower end (20 mg/Nm^3) of the typical European emission range given by [CEMBUREAU, 1997], it can be assumed that on a European scale, at present the average dust emission factor of a typical cement kiln is significantly higher than 50 g dust per Mg clinker.

These emissions are relevant not only because they mainly belong to the PM10 fraction, but at least potentially also because of heavy metals, other elements and persistent organics which can be absorbed onto the dust [UBA, 1998].

4 Reduction techniques for dust emissions

Although some primary techniques[6] for the reduction of dust emissions should not be neglected [UBA, 1998], the principal method to reduce the emissions from the main emission sources is reduction of the dust emission level by either electrostatic precipitators or fabric filters [IPTS, 1999].

Electrostatic precipitators

Electrostatic precipitators are the standard method for dust abatement in the oven exhaust gas of European cement kilns today.

In an electrostatic precipitator (EP), dust particles in the raw exhaust gas become negatively charged in an electrostatic field which causes them to migrate towards the positively charged collection plates. These collection plates are rapped or vibrated periodically, dislodging the material and allowing it to fall into collection hoppers below. Principally, EPs are able to operate under temperatures of up to 400 °C and under conditions of high

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[6] Including process control optimisation, steady feed rate of fuels and raw materials, good maintenance of all installations, etc.

humidity [IPTS, 1999]. Increased humidity often makes precipitation more efficient as it prevents formation of isolating alkali chloride layers on the surface of the electrodes [UBA, 1998]. At an operating temperature around 120 °C, the electric resistance of the dust is at an optimum for efficient precipitation, while at the same time the contents of volatile compounds (including heavy metal compounds) in the oven exhaust can be minimised.

EP efficiency depends on the number of electrostatic fields (normally between two and four), on the field dimensions, and on conditioning of the exhaust gas with respect to temperature, humidity, and particle resistance.

If a cement rotary kiln with cyclonic preheater is operating in compound operation, the oven exhaust gases are cooled down and take up moisture by raw material contact in the raw mill, which renders them in a good state for EP treatment. In direct operation, evaporation cooling with water injection is necessary for exhaust conditioning before the electrostatic precipitator can operate effectively. At cement kilns with grate preheaters, the oven exhaust gas has a lower temperature and higher humidity due to the process characteristics without special conditioning.

Electrostatic precipitators are very reliable in normal operation but have certain shortcomings during special conditions such as kiln start up, switching from compound operation to direct operation, and particularly during so-called CO trips when the voltage is switched off for a short period of time in order to avoid an explosion risk when CO concentrations in the oven exhaust become too high.

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The performance optimum of EPs lies at 1-2 mg/Nm^3 in routine kiln operation, with the above-mentioned exceptions under special conditions.

In many cases, it is advantageous to treat the clinker cooling air jointly with the oven exhaust: Part of the cooling air is often used as pre-heated combustion air in the kiln burners, another part can be used for drying of raw materials in the raw mill. Where such configurations are possible, joint treatment of clinker cooler air and oven exhaust gas in one EP is economically favourable compared to separate treatment in two EPs.

Existing EP installations can often be upgraded by fitting more modern electrodes or by installing automatic voltage control on older installations [IPTS, 1999]. Addition of an extra electrostatic field for improved precipitation maybe an option on some but not all existing installations. Improved conditioning of oven exhaust gas may also improve the performance of an existing EP.

Fabric filters

In the European cement industry, fabric filters are routinely used for gas cleaning at the cement mill, at the coal mill, at storage silos and bunkers, and at cement loading into bulk road or rail tankers. In many cases, clinker cooling air is also cleaned by fabric filters. While in North America cleaning of oven exhaust gas by fabric filters is quite common, this is a fairly new development in Europe. In early 1999, one newly built fabric filter for oven exhaust gas is being taken into routine operation at an existing rotary kiln (capacity: 3000 Mg per day) in North Germany; a second one has been recently taken into operation in Eclepens, Switzerland.

Fabric filters consist of a membrane which is permeable to gas but which will retain the dust. Initially, dust is deposited both on the surface fibres and within the depth of the fabric, but as the surface layer builds up it itself becomes the dominating filter medium.[7] As the dust cake thickens, the resistance to gas flow increases, making periodic cleaning of the filter medium necessary in order to control the pressure drop over the filter [IPTS, 1999]. The pressure drop over a new fabric filter is approximately 10 times higher than for an electrostatic precipitator [SCHOBESBERGER, 1998].

Periodic cleaning of fabric filters can be done either in off-line or in on-line mode. The most common cleaning methods include reverse air flow, mechanical shaking, vibration and compressed air pulsing [IPTS, 1999]. The duration of a cleaning pulse is about a few hundreths of a second. In order to optimize filter performance and reduce mechanical stress, cleaning is done in such a way that approximately 10% of the dust cake remain on the fabric.

Permeability of the fabric for dust is highest during the short cleaning pulse when it typically lies between 20 mg/Nm3 and 45 mg/Nm3, depending on the pressure of the cleaning pulse. If the basic permeability of a fabric filter is 1,5 mg/Nm3, the average dust passage over time will therefore lie between 1,8 mg/Nm^3 and 2,4 mg/Nm^3 [VDI, 1997].

Standard fabric materials include polyester fibres and sometimes also polyacryl nitrile fibres. Required material specifications include permeability to air, mechanical stability, and resistance towards temperature and chemicals. Fabric filters are fairly sensitive to temperature drops below the dew point as well as to temperature peaks.

For oven exhaust gas, polyester fibres do not provide sufficient durability at high temperatures. Instead, special membrane fibres (which are more expensive than polyester) are needed in this case [SCHOBESBERGER, 1998].

Lifetime of the fabric material is crucial for operating costs of the filters. Depending on the quality of material, gas temperature and volume, dust quantity and chemical composition, and maintenance of steady operating conditions, lifetime of the material typically lies between one and three years.

For new fabric filters, the suppliers routinely guarantee at least 30 mg/Nm^3 as the achievable emission level. Without technical changes, this guaranteed emission level can be lowered to 20 mg/Nm^3 by a more frequent exchange of fabric material, leading to increased operating costs. For a guaranteed emission level below 10 mg/Nm^3, the filter dimensions would have to be increased in order to reduce the load level per area of filter medium. This would result in higher investment costs.

To summarize, today there is wide consensus that average dust emission levels in oven exhaust from cement kilns below 10 mg/Nm^3 are achievable with either electrostatic precipitators or fabric filters [IPTS, 1999]. This is confirmed by numerous European cement kilns which are routinely running at that level.

There are, however, certain operating conditions kiln during which this emission level will be occasionally exceeded. Such situations have to be taken into account when an emission limit value for a cement kiln is fixed in a permit.

5 Costs of dust abatement

It is the task of this section to calculate the additional investment and operating costs which will arise when dust emissions from the cement industry are to be reduced by either improved EPs or by fabric filters. In the

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[7] This effect is utilized when new filter material is pre-coated with lime or cement before its first use [SCHOBESBERGER, 1998].

following sections 6 and 7, the costs for improved dust abatement will be compared with the social costs of unabated dust emissions.

In a first step, total costs for dust abatement at new cement kilns are calculated on the basis of specific information received in expert interviews with kiln operators and suppliers of dust abatement equipment. These abatement costs are presented not only for standard equipment, but also for improved abatement techniques which are able to comply with even stricter emission limit values than the EU Proposal COM (1998) 558.

In a second step, the upgrading costs that will allow existing kilns to meet the new requirements of Commission Proposal are estimated. As each plant is a singularity, it will only be possible to estimate these upgrading costs within a certain range.

Preliminary assumptions

It has already been stated above that the actual emission level of a kiln normally differs from the emission limit value in the permit. Additionally, most kiln operators require a guaranteed emission level from their supplier which is lower than the limit value in their permit.

In effect, the costs for filter equipment must refer to another type of emission level than the social costs of unabated particulate emissions, as the latter have to be based on average emissions while the former must refer to peak emissions.

To be able to compare the two aspects, Table 5 presents some assumptions with respect to emission limit values, suppliers guarantees, and the average emission levels over longer periods of time (e.g. one year):

Table 5 Relations between emission limit values, guaranteed emission levels, and average emission levels assumed for this study

Type of emission level	mg/Nm3			
Emission limit value (daily average TA Luft) - for calculation of investment and operating costs -	50	30	20	15
Emission level (guaranteed by supplier)	30	20	10-15	<10
Average emission level - for calculation of social costs and benefits -	<20	<15	<10	<5

The assumptions of Table 5 are based on an evaluation of emission limit values and regular performance data of more than 10 cement kilns; they were confirmed in expert interviews with several kiln operators and with suppliers of dust abatement equipment.

Traditionally, the standard guarantee which was given by the supplier of an electrostatic precipitator to a European kiln operator was 30 mg/Nm³ under all operating conditions. More recently, when new kilns are planned and built, lower guarantees of typically 15 or 20 mg/Nm³ are agreed upon between the kiln operator and the supplier. Some kiln operators demand for an even lower guarantee of 10 mg/Nm³ as they consider the slightly higher investment costs as a premature modernisation investment. There are suppliers who routinely guarantee 10 mg/Nm3 when a new kiln is fitted with dust abatement equipment.

Nevertheless, there are older cement kilns which have more problems to control the peaks of their dust emissions. In individual cases, it has proved problematic to safely keep an emission limit value of 50 mg/Nm³ (according to TA Luft - Table 3) even although the average emission level of the kiln was around 20 mg/Nm³.

Overall costs for dust abatement for new kilns

In our calculations of the overall costs for dust abatement, we have determined the annual equivalent costs of dust abatement at a discount rate of 8% p.a. and an assumed equipment lifetime of 10 years.[8] The investment costs cover the suppliers costs for construction, material etc. (price "flange to flange") plus the estimated extra costs for infrastructure such as tubes, stack, ventilator with motor, cables, integration into the process control system, and foundation.

Besides the investment costs, operating costs for energy, maintenance and repair, and personnel, were taken into account. Assumptions on costs are based on information received in expert interviews with kiln operators and suppliers of equipment.

The costs for dust abatement with electrostatic precipitators at three different kiln sizes of 1000, 2500 and 5000 Mg clinker production per day (kilns A, B and C, respectively) were then calculated for two variants:[9]

Variant (i) is characterised by joint treatment of oven exhaust and clinker cooler air in one electrostatic precipitator, while variant (ii) is characterised by two separate EPs for oven exhaust and clinker cooler air.

Details of example calculations are given in the Annex to this study. The results are summarised in Table 6

Table 6 Annual Equivalent Costs for dust abatement

[Standard situation: electrostatic precipitator; emission limit value 30 mg/Nm³]

[€ / Mg clinker]	Variant (i)	Variant (ii)
Kiln A	1,01	1,22
Kiln B	0,67	0,74
Kiln C	0,51	0,52

The costs estimates in Table 6 depend on the discount rate used to convert capital expenditure into annualised costs and the expected lifetime of the equipment. This study assumes a discount rate of 8% and a lifetime of 10 years. If 4 per cent was used as the discount rate, the costs would be around 86% of the costs given in Table 6.

Extending the lifetime would further reduce the annualised costs. For example, a lifetime of 15 years instead of 10 years would reduce the costs estimates by around 20 per cent.

As has been shown in Table 2, today the legal situation in most parts of

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[8] The actual overall lifetime of an EP is approximately 20-25 years, while exchange of electrodes or other internals is necessary after between 8 and 20 years.

[9] The standard kiln type here was a preheater/precalciner kiln. For existing Lepol kilns, the following statements can be made.

At Lepol kilns, joint treatment of oven exhaust and clinker cooler air is possible at relatively low costs. Where this is not possible, separate treatment of oven exhaust will be extremely expensive because all materials would have to be resistant against the highly corrosive atmosphere. In such cases, substitution of an existing Lepol system by a new preheater/ precalciner system will be favourable for economic reasons.

Europe would only require an emission limit value of 50 mg/Nm³ or even higher. However, the costs of meeting this value will be similar to the costs of meeting the 30 mg/Nm³ limit, as more or less the same technique will be installed anyhow. The possible savings would be less than 10% (for the small Kiln A) or even 5% (for the large Kiln C). In addition, no operator would normally install equipment that only complies with 50 mg/Nm³, as the risk of later incurring higher upgrading costs due to evolving legislation would be much higher than the short-term economic advantage.

Additional costs caused by stricter emission limit requirements for new kilns
According to suppliers, control of dust emissions at an emission limit value of 15 mg/Nm³ is technically feasible for new kilns today if required in specific cases. Still lower limits are technically more difficult to be safely observed under all operating conditions.

If an emission limit value of 15 mg/Nm³ is to be safely controlled by an electrostatic precipitator, the investment costs are typically between 15% (for large kilns) and 30% (for small kilns) higher than at 30 mg/Nm³. This has been confirmed both by kiln operators who decided to add a fourth electrostatic field to an EP which was originally designed for three fields, and for cases where the kiln operator decided to increase the EP dimensions while keeping the number of electrostatic fields constant.

Concerning the operating costs, electricity consumption increases with the number of fields in a degressive rather than in a linear way, as electricity consumption is mainly determined by the quantity of precipitated dust which is highest in the first field.

In summary, for an EP the additional investment plus operating costs for control of an emission limit value of 15 mg/Nm³, expressed as annual equivalent costs, are between 0,07 € per Mg clinker for large kilns and 0,34 € / Mg clinker for small kilns (Table 7).

At this cost level, fabric filters become an economic alternative to EPs. Besides the investment and electricity costs, their costs are determined by the frequency at which the exchange of fabric material will be necessary. According to CEMBUREAU [1997], the bags must be replaced every 2 - 4 years. To be on the conservative side, the costs given in Table 7 were calculated on the basis of a filter exchange after every two years.[10]

Table 7 Annual Equivalent Costs for improved dust abatement
[Electrostatic precipitator (EP) or fabric filter; emission limit value 15 mg/Nm³]

	Variant (i)		Variant (ii)	
	Total costs [€ / Mg clinke]	Marginal costs* [€ / Mg clinker]	Total costs [€ / Mg clinke]	Marginal costs* [€ / Mg clinker]
EP:				
Kiln A	1,30	0,29	1,56	0,34
Kiln B	0,82	0,14	0,89	0,15
Kiln C	0,59	0,08	0,60	0,07
Fabric filter:				
Kiln B	0,74	0,07	0,85	0,11
Kiln C	0,60	0,09	0,64	0,12

* Marginal costs refer to the increase in cost relative to the standard situation (Table 6)

At still lower emission limit values (e.g. 10 mg/Nm³), increased filter

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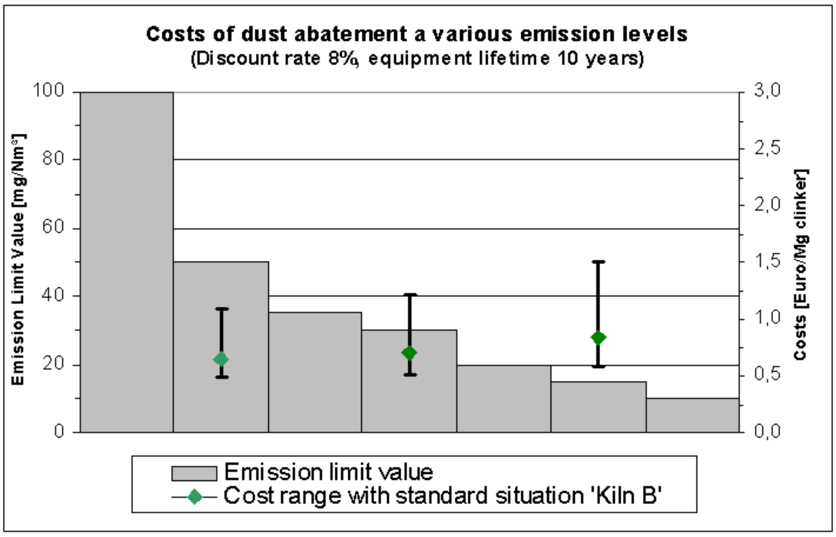
[10] Detailed example calculations are presented in the Annex to this study.

dimensions will be required which will result in significantly higher investment costs.

An illustration of the relationship between emission limit value and dust abatement costs is given in Figure 1.



For comparison, Table 8 provides information about the costs of dust abatement as published recently by [CEMBUREAU, 1997][11] and by [UBA Austria, 1998]:

Table 8: Cost of dust abatement according to other sources

all costs in [€ / Mg clinker]	Electrostatic precipitator	Fabric filter	Reference
Oven exhaust	0,4 - 0,8	0,5 - 0,9	Cembureau, 1997
Clinker cooler	0,22 - 0,38	0,26 - 0,38	Cembureau, 1997
Cement mill	0,22 - 0,38	0,08 - 0,12	Cembureau, 1997
Not specified	0,4 - 0,5	0,8 - 0,9	UBA Austria, 1998

Concerning the cost comparison between EPs and fabric filters, CEMBUREAU [1997] states: "The total cost per tonne of clinker caused by the dedusting is usually in favour of EPs if clean gas dust contents above about 30 mg/Nm³ are required. Below 20 mg/Nm³ the cost for dust filters is often lowest for pulse jet bag filters. This is only a general statement and can be different for specific applications. The reasons for the lower costs of EPs above 30 mg/Nm³ are mainly the low pressure drop over the filter and the reduced maintenance costs."

According to UBA Austria [1998], specific costs of fabric filters are circa 0,35 € / Mg clinker higher than the costs for EPs. Their study states that the investment costs for EP's and fabric filters are rather similar; the cost difference is caused by the higher operating costs which are mainly determined by the necessary exchange of fabric material, and also by the higher electricity consumption due to the larger pressure drop over the

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[11] CEMBUREAU costs are total costs including investment and operating costs. They are standardised for a preheater/precalciner kiln with a capacity of 3000 Mg per day. Amortisation was 10 years at an interest rate of 10%.

fabric filter.

Additional costs caused by stricter emission limits for existing kilns

As has already been stated, each existing kiln is a singularity and it is therefore difficult to make generalised assumptions about the additional costs that will arise when the kiln operator is required to install a better dust abatement equipment.

In some cases, upgrading of an existing EP by fitting new electrodes with a larger collecting area maybe an option. In other cases, it may be feasible to add another electrostatic field to an existing EP. Under favourable conditions, both operations could be successfully completed during the annual routine maintenance shutdown of the kiln, while under unfavourable circumstances a longer shutdown time might be needed. In the latter case, it may be more economic to build a completely new EP or fabric filter besides the old one and exchange it during the annual routine shutdown phase of the kiln.

Consequently, the upgrading costs of an existing EP to comply with an emission limit value of 30 mg/Nm³ may lie somewhere in the range between 30% and 100% of the cost of a completely new EP or fabric filter. As the costs for a new standard EP have been calculated with 0,51 - 1,22 € /Mg clinker (Table 6), the upgrading costs are estimated in the range between 0,15 and 1,2 € /Mg clinker. If the upgrading is aimed to comply with a stricter emission limit value of 15 mg/Nm³, the additional marginal costs are estimated between 0,1 and 0,34 € / Mg clinker

There will certainly be cases for which such an upgrading can be a serious option only after the depreciation period of the existing EP has long expired. Experience shows that such an investment is made by kiln operators preferably not as an isolated action, but in the context of extensive modernisation investments, which may often include the concentration of several small kiln capacities in one larger unit.

Summary of dust abatement costs for new and existing kilns

t o summarize this section, the marginal costs for improved dust abatement will be as shown in Table 9. Marginal costs to achieve 30 mg/Nm³ at new kilns are derived from the discussion of Table 6 (p. 15), costs to achieve 15 mg/Nm³ are taken from Table 7:

Table 9 Marginal costs for improved dust abatement
(in € / Mg clinker)

Emission limit value:	50 mg/Nm³	30 mg/Nm³	15 mg/Nm³
New kilns low value	0	0,025	0,07
high value		0,1	0,34
Existing kilns low value	0	0,15	0,1
high value	(?)	1,2	0,34

Note: Costs for improved dust abatement at existing kilns which presently do not comply with an emission limit value of 50 mg/Nm³ will presumably lie at the high end of the range.

In Table 9, marginal costs are given for moving from one emission limit value to the next lower one (e.g. from 50 to 30 mg/Nm³). Therefore the total costs for moving from 50 mg/Nm³ to 15 mg/Nm³ will lie between 0,1 and 0,44 € / Mg clinker for new kilns, and between 0,25 and 1,54 € / Mg clinker for existing kilns.

6 External Costs of unreduced particulate emissions

As has already been mentioned in Section 2 of this study, emissions of fine particulates have manifold harmful effects to human health because they intrude deeply into the bronchi and even reach the pulmonary alveolus, and they weaken the self-cleaning mechanism of the lungs.

While in the upper parts of the respiratory tract, particles are removed within few hours, the clearance of non-toxic, unsoluble particulates from the alveolus is a very slow process [LfU, 1998].

Because the average half-life of unsoluble particulates is approximately 500 days, they can accumulate in the lungs and become a focus of inflammation. Although the exact biological and medical mechanisms are not yet fully understood, a high correlation between PM10 immission concentrations and and increased prevalence of cough, expectoration, respiratory infections and shortness of breath is clearly observable [BUWAL, 1996; LfU, 1998].

Additionally, respiratory malfunctions and deterioration of other diseases result in increased numbers of hospital admissions and emergency room visits, as well as increased mortality [BUWAL 1996].

In their examination of the costs and benefits of the proposed new emission limits for waste incineration, ETSU [1996] have assessed the external costs which are caused by particulate emissions. In their assessment, they have taken into account increased mortality, changes in hospital admissions (for respiratory infections and for chronic obstructive pulmonary disease), changes in emergency room visits (for COPD and asthma), hospital visits for childhood croup, bronchitis, restricted activity days, shortness of breath days (for asthmatics) and symptom days in general.

ETSU [1996] point out that these associations have been found at normal background levels in a wide range of locations. They state that not only there is no threshold level of pollution for these effects, but rather there is "some evidence, though not compelling, that the health effects per unit incremental exposure are higher when background pollution levels are lower".

Swiss observations confirm that the dose-effect relationships are valid at relatively low immission levels, which is in agreement with the assumption that no threshold level exists [BUWAL, 1997]. Norwegian scientists also state that in the case of particulates the dose-response functions seem to apply at very low concentrations [STATISTICS NORWAY, 1998].

One might argue whether external cost calculations for particulate emissions from waste incinerators are transferable to particulate emissions from another industrial sector such as the cement industry. At this point, it must be clearly stated that the existing assessments of external damage caused by PM10 mostly have not put a special emphasis on the chemical composition of the particulates: "Size is what is considered most important in a normal pollution situation" (STATISTICS NORWAY, 1998]. There is wide agreement that other pollutants, particularly SO2 and NOx, may contribute to the health effect of air pollution, but in "several studies, SO2 effects apparently disappear when particulates are measured correctly" (ibd).

Certainly, other particulate constituents such as e.g. adsorbed heavy metals or persistent organics may contribute to the overall health impact but are even less well examined.

In their most recent work, the authors of ETSU [1996] still consider their assumption that all fractions of PM10 are equally aggressive to human health as an uncertainty [AEA, January 1999].

However, this uncertainty cannot overthrow the scientific finding that fine particulates are an air pollutant of prime concern, and at present the selection of PM10 as the target variable to describe dose-effect relationships appears to be the best choice for practical reasons.

By transferring the ETSU data for external costs of PM10 particulates in general to dust emissions from cement kilns, the external costs of the latter could be overestimated because

- raw materials and cement dust might be less toxic than average PM10 because of their chemical composition of mainly inert materials;
- within the range of all PM10 particulates, particulates emitted from the cement industry are still relatively coarse, while the particulates of greatest significance for human health rather woud be the finer particulate fractions (PM2,5, PM1 or PM0,1).

On the other hand, external costs of particulate emissions from the cement industry could be underestimated by this approach, because the particulates may contain

- carcinogenic constituents such as hexavalent chromium, arsenic, nickel;
- toxic constituents such as lead, zinc, cadmium, manganese or vanadium;
- etching constituents such as lime or chromium
- and / or allergenic constituents such as hexavalent chromium or nickel.[12]

ETSU [1996] have calculated the external costs of PM10 emissions for three different locations in Europe, and for three different stack heights, as shown in Table 10. In their calculation, only acute health effects were taken into account.

Table 10 External costs of unreduced dust emissions [ETSU 1996]
(in ECU / Mg of PM10) - Acute health effects only -

Location \ Stack height	50 m	90 m	100 m
Paris	120.431	68.818	57.348
Stuttgart	49.442	36.698	28.674
Birmingham	74.694	39.530	30.680

With respect to site dependence of the external costs, ETSU [1996] refer to previous studies by ARMINES which "have shown that the health damage from air pllution varies by about a factor of ten for a hypothetical plant located in Paris, compared with one at a rural site on the Atlantic Coast, and that the average damage for sites in France can be estimated within a factor of three". According to them, variation with stack height amounts roughly to a factor of two, depending on the vicinity of a large population centre [ETSU, 1996, p. 7-4 f.].

The assumed large dependence of the external costs on the location of the emission source seems to be somewhat in contrast with the growing knowledge about long-range transport of fine particulates (see Section 2 of this study). If fine particulates are being transported over hundreds of kilometers, and if even a large city such as Berlin with all its big and small emission sources adds only 20 per cent to the widespread immission level which is already existent, it seems to be justified to assume that the variation between external costs of PM10 emissions emitted in a densely populated compared to a remote rural area will be significantly smaller than a factor of ten.

In order to determine the cost-benefit-ratio between improved dust abatement in the cement industry and external costs of particulate emissions, we therefore suggest to leave out the very high external costs of the Paris example, and to work with a cost range between 30.000 and 75.000 € / Mg dust (as calculated by ETSU for Stuttgart and Birmingham) for more densely populated areas. With a variation factor of 3 for an emission source which is located in a remote rural area, the lower end of the

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[12] . Concentrations of toxic chemical elements in particulate emissions from the cement industry may increase when wastes containing these elements are co-incinerated.

cost range will then be between 10.000 and 25.000 € per Mg of PM10 emissions.[13]

For the subsequent comparison of costs and benefits of improved dust abatement in the cement industry (Section 7), the cost range between 10.000 € and 75.000 € per Mg of PM10 will be taken as a basis for the external costs. It should be kept in mind, though, that these external cost estimates are very sensitive to the method chosen to value mortality effects, as well as which health effects are included and excluded, and also concerning the issues of how location affects the population exposed.

For reasons of consistency with earlier work, estimates in this study are based on the figures of the earlier ETSU [1996] report on waste incineration. In that study, the method chosen to value mortality effects was a "value of a statistical life" (VOSL) approach with a figure of 3 million ECU per life. At the same time, only the acute health effects were quantified by ETSU.
Some analysts propose an alternative method to value mortality effects that uses the "value of life years lost" (VOLY) approach instead of VOSL. This approach would reduce the estimated external costs, while on the other hand including the possible chronic effects would increase the external costs.

A more detailed discussion of these valuation issues has been elaborated recently in the context of discussions of emission ceilings for atmospheric pollutants.[14]

7. Comparison of costs and benefits of improved dust abatement

In Section 3 of this study, literature data about specific exhaust gas volumina [UBA, 1998] and the corresponding emission factors for dust emissions [HACKL & MAUSCHITZ, 1997] have been presented.

For calculation of the external benefit of avoided particulate emissions, the following relations between average emission concentrations and emission factors (standardised to kiln capacity) are used:

Table 11 Average emission concentrations [mg / Nm³] **and emission factors** [g PM10 / Mg clinker]

average emission concentration [mg dust / Nm³]	30	20	15	10	5
Emission factor [g PM10 / Mg clinker]	75	50	37,5	25	12,5

On the basis of the assumptions concerning the relation between emission limit values and average emission levels (Table 5), and with a range of external costs between 10.000 € and 75.000 € per Mg of PM10 emission, the benefits of improved dust abatement in the cement industry are estimated as shown in Table 12.

Table 12 External benefits of improved dust abatement in the European cement industry

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[13] A more detailed analysis would have to be based on a detailed assessment of one or several specific sites, which would have been clearly beyond the scope of this study.

[14] "Economic Evaluation of Proposals for Emission ceilings for Atmospheric pollutants" by IIASA / AEA Technology (for DG XI of the European Commission, January 1999).

Emission limit value [mg/Nm³]	50	50	30	20	15
Average emission concentration [mg/Nm³]	30	20	15	10	5
PM10 emissions [g per Mg clinker]	75	50	37,5	25	12,5
Avoided PM10 emissions	[-25]	0	+12,5	+12,5	+12,5
Marginal benefit [€/Mg clinker] - high estimate -	[-1,88]	0	0,94	0,94	0,94
Marginal benefit [€/Mg clinker] - low estimate -	[-0,25]	0	0,125	0,125	0,125

Estimates of the benefit are conservative inasmuch as the avoided emissions are probably underestimated. This is especially valid for existing kilns whose specific PM10 emissions at present maybe significantly higher than 50 g per Mg clinker.

If data from Table 9 and Table 12 are combined, the cost-benefit ratios between costs of improved dust abatement and external benefit due to avoided PM10 emissions are calculated as shown in Table 13.

As can be seen from Table 13, lowering the dust emission limit for new cement kilns from 50 mg/Nm³ to an emission limit value of 30 mg/Nm³ will bring about a cost-benefit ratio > 1 even if the marginal costs are assumed to be at maximum while the marginal benefit is estimated at the low end of the range. Depending on the specific situation of the kiln and on the uncertainties about the exact external costs, the cost-benefit ratio will lie between 1:1,25 and 1:37.

If the emission limit value is lowered further down to 15 mg/Nm³, the costs will increase to a certain extent while at the same time the marginal benefit will be higher. In this case, the cost-benefit ratio will lie between 1:0,74 and 1:28, depending on the specific case.

The situation is more complex for existing kilns, as the cost-benefit ratio depends strongly on the initial emission level:

Table 13 Comparison of costs and benefits of improved dust abatement in the European cement industry

Initial emission level [g PM10 per Mg clinker]	20 g / Mg			30 g / Mg	
Reduction of Emission limit value [in mg/Nm³]	50 30	30 15	50 15	50 30	50 15
Emission reduction [g PM10 per Mg clinker]	5	10	15	15	25
Marginal costs for low new kilns [€/Mg] high	0,025 0,1	0,07 0,34	0,1 0,44	-	-
Marginal costs for low existing kilns [€/Mg] high	0,15 1,2	0,1 0,34	0,25 1,54	0,15 1,2	0,25 1,54
Marginal benefit low [€/Mg] high	0,125 0,94	0,25 1,88	0,375 2,81	0,375 2,81	0,625 4,69
Cost-benefit ratio lowest for new kilns highest	1,25 37,6	0,74 26,9	0,85 28,1	-	-
Cost-benefit ratio lowest for existing kilns highest	0,1 6,3	0,74 18,8	0,24 11,24	0,31 18,7	0,41 18,8

If lowering the emission limit value from 50 to 30 mg/Nm³ will lead to a reduction of the average emission concentration from 20 to 15 mg/Nm³, the additional dust abatement costs for the kiln operator are roughly in the same order of magnitude as the external benefit (cost-benefit ratio between 1:0,1 and 1:6,2). Whether an upgrading of the dust abatement equipment will yield an external net benefit or not, is determined by the technical upgrading options at the specific kiln, and by the assumed marginal benefit (which maybe site-specific, depending on the location of the kiln).

If, however, the average emission level of the existing kiln is initially higher, the external benefit of an improved dust abatement and thus the cost-benefit ratio will be much higher, too. With an initial emission level of e.g. 30 mg/Nm³ on average, the cost-benefit ratio will lie between 1:0,3 and 1:19. With an even higher initial emission level, the cost-benefit ratio of improved dust abatement at an existing kiln maybe >1 under all circumstances.

If an existing kiln with a high initial dust emission level is equipped with a better dust abatement equipment, in many cases the cost-benefit ratio will be more favourable when the actual emission level is immediately lowered to < 5 mg/Nm³, rather than just bringing it down to an average emission level of < 15 mg/Nm³.

8 Commercial advantages from using wastes as fuel substitutes

By taking in wastes as secondary fuel, an economic advantage is given for the operator of a cement kiln for two reasons: On the one hand, he can save expenses for regular fuel, on the other hand he can charge a disposal fee for the wastes.

Based on data about fuel costs and disposal fees that were obtained from various industries, the economic net revenue from the substitution of 5% of regular fuel by wastes as secondary fuel is estimated at circa 0,7 € per Mg of clinker produced.

In practice, many European kiln operators are presently substituting between 25% and 50% of their energy demand by secondary fuels, which is corresponding to an economic advantage of often more than 5 € / Mg clinker.

Depending on the individual situation, the proportion of wastes whose co-incineration can cover the additional dust abatement costs to comply with an emission limit value of 30 mg/Nm³ will lie between 0,2% (for large new kilns) and 8,5% (for small existing kilns). If an emission limit value of 15 mg/Nm³ is to be complied with, the respective costs can be covered by substitution of 0,5% of regular fuel at large new kilns as the lower extreme, and by substitution of 11% of regular fuel by small existing kilns as the upper extreme.

9 Summary

In the existing legislation of individual EU Member States, emission limit values for dust from the cement industry vary between 15 mg/Nm³ for individual new kilns and 400 mg/Nm³ for some older existing kilns. In their Proposal [COM (1998) 558] on the incineration of wastes, the European Commission has suggested an emission limit value of 30 mg dust/Nm³ for cement kilns that burn waste as a fuel.

The majority of dust emissions from the cement industry belong to the fine particulate fraction with an aerodynamic diameter below 10 µm (PM10). This PM10 fraction is of prime concern for human health, because the fine particulates intrude deeply into the human respiratory tract. Recently, the international debate on air quality is focusing on particulates because they seem to correlate best with severe health effects such as respiratory infections, chronic obstructive pulmonary disease, childhood croup, bronchitis, and asthma.

Estimates of the external cost associated with these health effects are very

sensitive to the valuation methodology which is still subject to scientific debate. Based on the existing literature, for the purpose of this study the external costs of dust emissions are assumed to lie between 10.000 € and 75.000 € per Mg of PM10.

Abatement techniques for reduction of dust emissions which allow cement kilns to comply with an emission limit value of 30 mg/Nm³ or even 15 mg/Nm³ are readily available on the market. The two principal types of equipment are electrostatic precipitators and fabric filters. Their investment and operating costs were investigated in expert interviews with kiln operators and suppliers of equipment who have experience in the installation and routine operation of these filters.

Depending on kiln configuration, kiln size and quality of equipment, the overall costs of dust abatement range from 0,5 € to 1,5 € per metric tonne of clinker produced. The marginal costs of improved dust abatement that enable new kilns to comply with the Commission Proposal are between 0,025 € and 0,1 € per Mg clinker produced, which is equivalent to a cost-benefit ratio between 1:1,25 and 1:37. In other words, for new kilns even under unfavourable assumptions the external benefit of improved dust abatement is always greater than the costs.

For existing kilns, the situation is more complex: Depending on the specific case, the cost-benefit ratio will lie in a range between 1:0,1 and 1:19 (or even higher). In many cases, the reduction of dust emissions by better dust filters will yield a net benefit. This is particularly true for older kilns which at present may have an average emission level significantly above 20 mg dust per Nm³. In individual cases, however, the upgrading of an existing kiln may not result in a net benefit because the initial emission level of the kiln is already relatively low (although the kiln is unable to comply with Proposal [COM (1998) 558]), while the costs for upgrading the existing filter equipment are at the high end of the range, and while at the same time the marginal benefit of the improvement is assumed to lie at the low end of the spectrum due to site-specific factors.

Although slightly more expensive for the operator, a more ambitious reduction of dust emissions, i.e. to comply with an emission limit value of 15 mg/Nm³, may yield similar or even more favourable cost-benefit ratios than the investment which is required to just comply with the Commission Proposal. When planning a new kiln, some kiln operators install such improved equipment voluntarily as they consider the slightly higher investment costs as a premature modernisation investment.

By taking in wastes as a secondary fuel, an economic advantage is given for the kiln operator which often amounts to more than 5 € per Mg clinker. Depending on the individual situation, the additional costs for dust abatement to comply with an emission limit value of 30 mg/Nm³ can be covered by co-incineration of an amount of wastes equivalent to between 0,2 % (for large new kilns) and 8,5 % (for small existing kilns) of the regular fuel demand. If an emission limit value of 15 mg/Nm³ is to be complied with, the respective costs can be covered by substitution of 0,5 % of regular fuel as the lower extreme (at large new kilns), and by substitution of 11% of regular fuel as the upper extreme (at small existing kilns).

Experience shows that upgrading investments at existing kilns are preferably not made as an isolated action, but in the context of extensive modernisation investments which may include the concentration of several small kiln capacities in one larger unit.

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