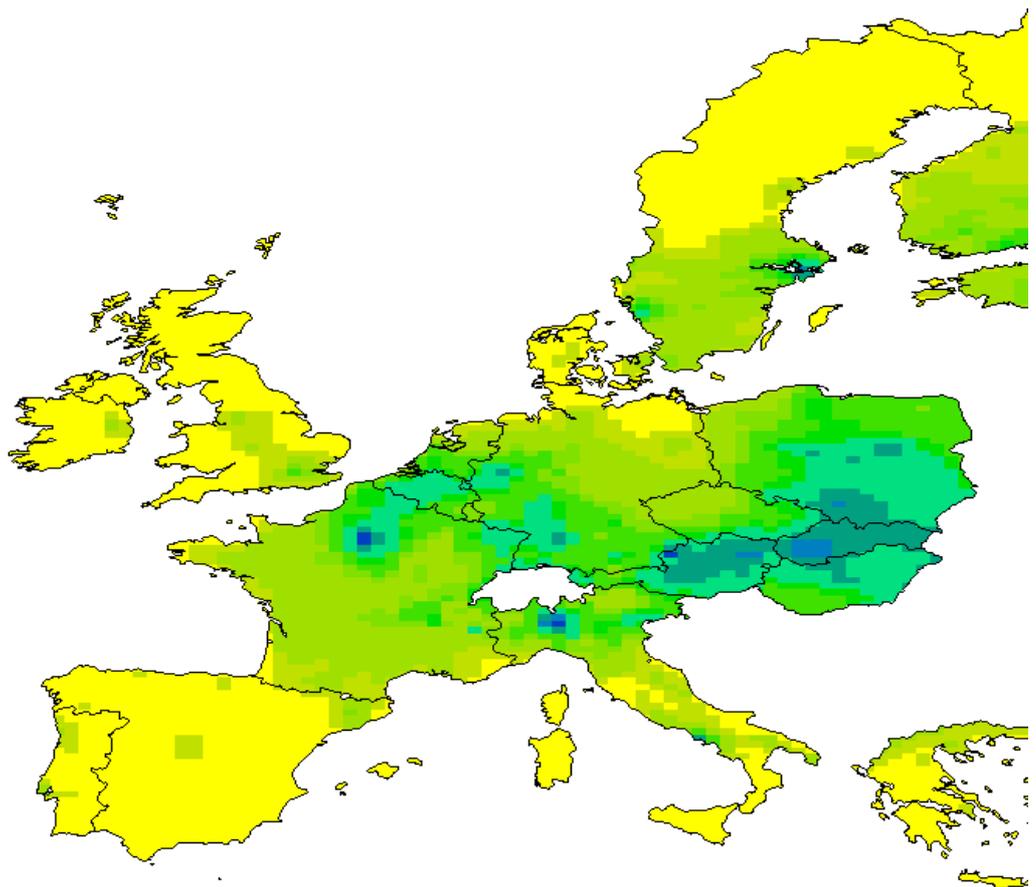
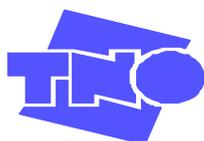


# Economic Evaluation of Air Quality Targets for PAHs

Final report for  
European Commission DG Environment



March 2001





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# Summary

## ***Background to the study***

AEA Technology and TNO were contracted by the European Commission (DG Environment) to undertake an economic evaluation of possible air quality limits for PAHs that are being considered under the Framework Directive on Ambient Air Quality in January 2000. This is the Final report on the study.

Concern over PAHs arises because they have been linked to lung and other cancers, through epidemiological assessment of workers in the aluminium and some other industries. The assessment made here considers conditions in the EU15 and the first group of Accession Countries (Cyprus, Czech Republic, Estonia, Hungary, Poland and Slovenia).

The EC's Working Group on PAHs and the Steering Group on Ambient Air Quality requested consideration of five possible limit values: 0.01, 0.05, 0.5, 1.0 and 5.0 ng/m<sup>3</sup> of benzo[a]pyrene (B[a]P), to be achieved by 2010. The figure of 5 ng/m<sup>3</sup> was introduced because of concern over the ability of some industries to meet the other limits suggested.

The Working Group concluded that it is appropriate for B[a]P to be used as a marker for overall PAH concentration, based on analysis that suggests that it make a consistent contribution (in the order of 40%) to the overall carcinogenicity of other species. Further analysis suggests that the role of B[a]P may be less important than previously estimated (representing between 5 and 41% of overall carcinogenicity amongst the group of PAH compounds for which data are available, depending on alternative assumptions regarding the potency of individual species). The role of the dibenzopyrenes in particular, may be more significant than has generally been considered. However, the study team regard it still to be appropriate to focus on B[a]P given limited data on other PAH compounds, and the dominance of particle-bound PAHs in assessments of total carcinogenicity of PAH mixtures. The only source category where this seems likely to cause problems relates to wood preservation, for which emissions of B[a]P are relatively low compared to emissions of the lighter compounds such as fluoranthene.

## ***Methods***

The analysis proceeds through the following stages:

1. Development of emission inventories for the EU15 and first 6 Accession Countries (together referred to as the EU21).
2. Estimation of the concentration of PAHs in the ambient air
3. Quantification of the costs of reducing PAH concentrations to the possible limit values identified by the European Commission and its Working Group on PAHs
4. Review of the effects of PAHs
5. Quantification of the benefits of reducing PAH concentrations to the limit values under investigation, in terms of impacts avoided and economic value
6. Comparison of costs and benefits, taking account of the uncertainties that are present.

A theme in work on PAHs is that a high level of uncertainty is present in many areas – quantification of emissions, assessment of concentrations, impact assessment and so on. Here,

the study team has taken the decision to quantify to the extent possible and to report the level of uncertainty that is present. Others have taken a different view. Notable among these is the air quality assessment team working on the second Auto-Oil Programme (AOP-II), for which analysis of PAHs, alongside PM<sub>2.5</sub> and 1,3 butadiene was not attempted because of a lack of data on emissions and concentrations that was considered to be of 'sufficient quality'. It is important that this level of uncertainty is borne in mind when considering the results of this study. There is a clear need for a more systematic collection of data on PAHs, though this will, in part, be facilitated through the monitoring requirements of any new directive.

### ***Emission sources***

The following sources of PAHs have been considered in the study, with those estimated to pose the most severe problems in the EU21 in 2010 identified in bold:

- Industry
  - **cokeries**
  - **aluminium works (specifically plant using the Soderberg-process)**
  - wood preservation using carbolineum and creosote
  - road building
  - roofing
- Domestic combustion
  - **wood**
  - **coal**
  - other fuels
- Traffic
  - **diesel vehicles**
  - gasoline vehicles
  - other

The European emission inventory for 2010 is dominated by emissions from domestic combustion of wood. Total emissions from the industries traditionally associated with PAHs (aluminium works, cokeries and wood preservers), are forecast to be much lower. However, emissions from these sources are far more concentrated, and hence can give rise locally to much higher concentrations, than the domestic or traffic sources. The higher total emission from domestic sources and from transport causes a higher level of exposure across the population, even though peak concentrations are significantly lower, by around an order of magnitude, than those in the immediate vicinity of some industrial plant.

### *Background concentrations*

Rural and urban background concentrations are estimated to be highest in central Europe, at around 0.4 ng.m<sup>-3</sup> and 1 ng.m<sup>-3</sup> respectively. These levels clearly far exceed the lowest standards under investigation (0.01 and 0.05 ng.m<sup>-3</sup>). The required level of abatement against these standards in some background locations in 2010 is therefore estimated to be well in excess of 90%, although industrial plant will be operating at BAT, EURO IV vehicle standards will be in the process of implementation, and higher efficiency domestic solid-fuel heating devices will have made a significant impact in the market in some countries. The study team does not consider that these lowest limit values are attainable on a reasonable time scale if at all. Attention has therefore focused on the higher suggested limit values ( $\geq 0.5$  ng.m<sup>-3</sup>).

## ***Industrial sources***

Problems associated with industrial and commercial activity are predicted to be much reduced by 2010. It is anticipated that the older style Soderberg process plant will have been phased out in the European aluminium industry, most having been closed or adapted to the use of pre-baked anodes in the 1990s. Analysis of emissions and effects of Soderberg plant has been carried out, however, to demonstrate the extent of problems that would be present should any remain by 2010. Changes in demand and the type of feedstock used in other areas, such as wood preservation, roofing and road-surfacing mean that they too are forecast to generate an insignificant level of emissions. The only exception by 2010 is likely to concern cokerries.

Even under best available techniques (BAT), B[a]P levels around a number of coking plants would exceed  $5 \text{ ng.m}^{-3}$ . It seems unrealistic to assume that significant reductions in emissions from these plant are possible beyond BAT, unless closure is considered. However, the effect of closure may well be to shift production out of the EU21 to other parts of eastern Europe or to Asia, in other words, to parts of the world that currently have a lower level of environmental regulation. This is clearly against the spirit of the air quality directives.

The extent to which emissions from any cokery will lead to exceedence of the standards considered here, in locations where people who are not employed at the plant spend a significant amount of time (and hence be relevant to an air quality Directive), is not clear. The uncertainty arises largely because (in the absence of plant-specific analysis) we do not know the distance from each cokery to nearby housing. This distance may be considerable, with the two being separated by (for example) other industrial buildings, railway sidings, storage facilities, etc. The analytical problem is compounded by the fact that concentrations around cokerries fall away rapidly with distance. For a plant producing 1 million tonnes of coke per year, it is estimated that the concentration at the source would be in the region of  $5 \text{ ng.m}^{-3}$ , falling to around  $1 \text{ ng.m}^{-3}$  at a distance of  $1 \text{ km}^1$ . However, significantly higher concentrations have been recorded at some plant. To improve the analysis to assess conditions at the worst affected site relevant under a directive would require a series of much more detailed assessments around individual coking facilities. This was outside the scope of this study.

Despite the estimated high concentrations around cokerries, the affected population close to cokerries is small, with the number of cancers estimated ranging from 0.03 to 2.3 cases per year, with a best estimate of 0.3 cases *across all plant in the EU21*. It is estimated that the effects on the population further from the plant will be greater than those close to the plant: in other words, population exposed typically increases with distance from a plant at a faster rate than the concentration decreases. This increases the best estimate of the annual cancer rate attributable to cokerries to 4.4/year.

Very large exceedences are also possible around Soderberg-process aluminium plant, should any remain in the EU21 by 2010. However, concentrations attributable to these plant fall rapidly with distance, declining to about  $1 \text{ ng.m}^{-3}$  within 2 km of the source. Results suggest a similar level of effect to those estimated for cokerries – very few people being exposed to the high concentrations. Overall, a best estimate of only 1 cancer per year has been calculated as a result of PAH emissions from European Soderberg plant in 2010.

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<sup>1</sup> Concentrations roughly scale with production for larger and smaller plant. Much higher average concentrations have been reported around some cokerries.

## ***Traffic***

The situation with respect to the control of emissions from traffic is far more complex. Many measures are available for abatement of emissions, including:

- Technical measures (e.g. particle traps, catalysts, alternative fuels and drive-trains),
- Economic measures imposed at the national level (e.g. fuel taxes, scrappage subsidies), and
- Local 'non-technical' measures (e.g. encouraging public transport and cycle use, improving traffic flows, road pricing).

However, many of the measures that could be introduced to reduce PAH levels will follow from legislation on other pollutants, and measures to reduce congestion and noise in towns and cities. Definition of baseline concentrations in 2010 is clearly problematic given uncertainty over the extent to which these measures will be implemented. Attainment of a  $1 \text{ ng.m}^{-3}$  limit value in areas subject to traffic hot-spots would reduce the number of cancers by an estimated 10/year for the EU21, going to  $0.5 \text{ ng.m}^{-3}$  would reduce the number of cancers by 21/year. The limited extent of the problem linked to traffic means that it is unlikely that additional measures specifically directed to PAH abatement from traffic could in general be justified on cost-benefit grounds. The situation may well be different were the analysis widened to include the numerous other impacts related to traffic, such as emissions of other pollutants ( $\text{PM}_{10}$ , VOCs,  $\text{NO}_x$ , etc.), noise, congestion and accidents. Widening of the analysis in this way has not been done as we do not regard it appropriate to justify limit values for one pollutant in terms of other environmental impacts. There may, after all, be more cost-effective options for controlling these other burdens than the measures that are most appropriate to PAH control. The need to control traffic emissions to meet either a  $1.0$  or  $0.5 \text{ ng.m}^{-3}$  limit could be significantly reduced by action on domestic emissions, given the contribution of these sources to background levels.

## ***Domestic sources***

Solid fuel consumption in the home is forecast to cause 86% of the lung cancers attributable to PAHs in 2010 (352 cases out of a total of 408). Although a number of options have been identified for abatement there is a lack of cost-effectiveness data. These options include, but are not restricted to:

- Fuel switching,
- Use of optimised stoves,
- Information campaigns to disseminate best practice in using solid fuel appliances and fireplaces, and
- Prohibition of solid fuel use in problem areas.

Comparison of the benefits of reducing emissions to meet a limit of  $1 \text{ ng.m}^{-3}$  with the costs of stove optimisation suggests that the two are broadly comparable. Other options, such as the use of alternative fuels need to be explored, though feasibility and costs of, e.g., the provision of natural gas to areas currently not served by pipelines are likely to be very site specific. Not surprisingly there are many other factors relevant to this debate, including emission of other pollutants and potential effects on the rural economy of restricting wood burning.

### ***Recommendations for further research***

The most fundamental research needs relate to emission factors from domestic and traffic sources. The problems here are clear from the general absence of agreement seen in national inventories. The following recommendations for research are made:

#### ***Industrial facilities:***

1. Individual plant studies to identify the extent and type of situations where exceedences of any proposed limit value may be expected.
2. Clarification of the potential for *new* cokerries to meet limit values.

#### ***Domestic combustion:***

1. Refinement of emission factors to take proper account of the type of wood or coal burned, condition of wood, type of fireplace/stove, effectiveness of existing controls, and so on.
2. Measurement of PAHs in a variety of locations where domestic emissions may be expected to be significant.

#### ***Traffic:***

1. Emission factors need to be derived from a representative fleet of vehicles over the driving cycles usually considered in Europe, and for which extrapolation to estimate total emission from the sector is now done routinely.
2. Measurements of PAHs need to be made at roadside locations to demonstrate the reliability of the data presented here.
3. National and local studies on the costs of meeting air quality standards for other pollutants across Europe need to be collated. These should identify the extent of exceedences in individual cities, the variation in the extent to which different options are expected to lead to reduction in exceedences, and costs.

#### ***Analysis of impacts:***

1. Refinement of risk factors for individual PAHs.
2. Further investigation of risk factors for effects other than lung cancer, particularly bladder cancer.

### ***Final remarks***

Despite a high level of uncertainty in input parameters, this study has traced through the pathway of emission, to concentration, impact and abatement for PAHs. Together with the report of the Working Group this work provides a much better basis for discussion of the problems of exposure to PAHs in the ambient air than has existed previously. Results indicate that, from a cost-benefit perspective, actions to reduce PAH emissions should focus on domestic burning of wood and coal. Action on specific industries and traffic appears less important, and may in any case be addressed through other legislation, such as IPPC and other daughter Directives on air quality.

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The content of the report remains (of course) the responsibility of the authors, and should not be interpreted as reflecting the official position of any of the above.

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# 1 Introduction

## 1.1 OBJECTIVES

The objective of this study is to estimate the costs and benefits of meeting possible ambient air quality standards for polycyclic aromatic hydrocarbons (PAHs), to inform DG Environment of the European Commission and other relevant parties, including the Working Group on PAHs and the Steering Group on Ambient Air Quality. The PAHs are the second type of genotoxic carcinogen to be considered under the Framework Directive on Ambient Air Quality, following earlier work on benzene (EC Working Group on Benzene, 1999; AEA Technology, 1999).

PAHs are a large group of compounds with a similar structure comprising two or more benzene or other aromatic hydrocarbon rings. They are formed through incomplete combustion of carbon compounds. PAHs vary with respect to sources, and chemical and physical characteristics (importantly, they may be in the gas phase or particulate bound).

Concern over PAH emissions relates to their health effects. The earliest relevant observations were made by Sir Percival Pott in London as early as 1775, regarding the incidence of testicular cancers among chimney sweeps' apprentices. However, it took many years before specific carcinogens were identified, the first of these being benzo[a]pyrene (B[a]P), extracted from coal tar in 1933.

## 1.2 LIMIT VALUES ASSESSED

The Working Group on PAHs and the Steering Group on Ambient Air Quality convened by EC DG Environment requested that five limit values be investigated for B[a]P. These were:

**0.01, 0.05, 0.5, 1.0 and 5.0 ng.m<sup>-3</sup> annual averages to be met by 2010**

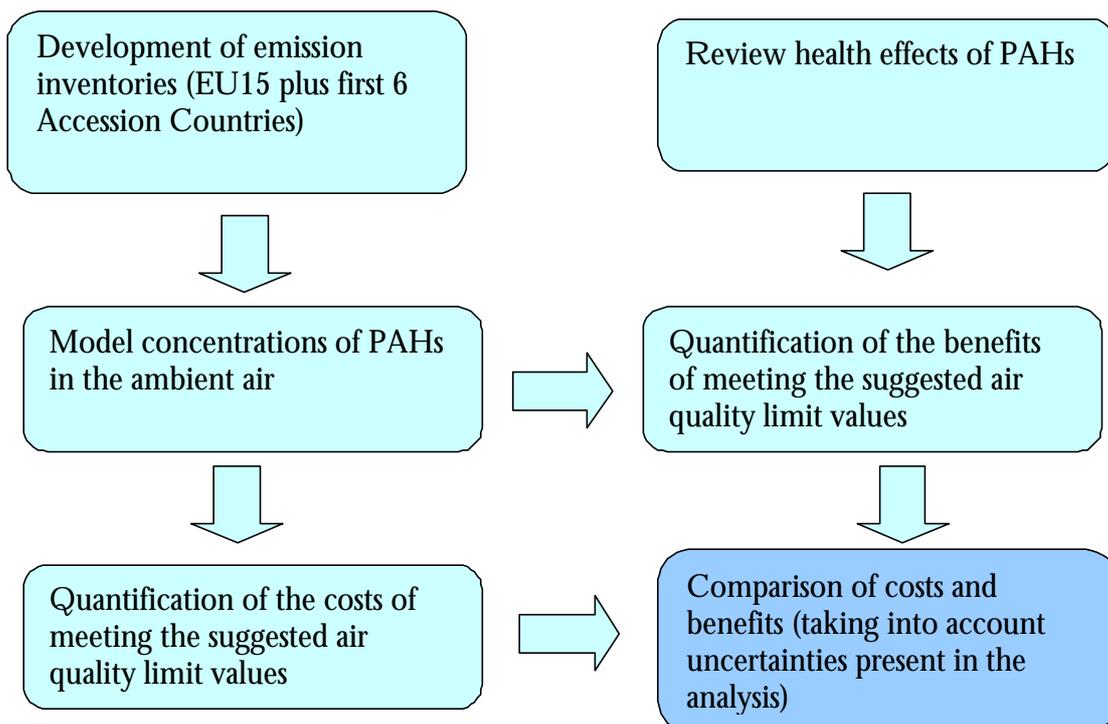
The 0.01 ng.m<sup>-3</sup> value was derived from the World Health Organisation's (WHO) estimate of risk of cancer in humans of  $1 \times 10^{-6}$  per life-time per ng.m<sup>-3</sup>. At the other end of the scale the 5 ng.m<sup>-3</sup> value was included in the range following concerns expressed at the Steering Group over potential problems with certain industrial plant, particularly cokeries. None of these values were to be regarded during the analysis as preferred to any other. The Commission also required that the analysis should take specific account of both background and 'hot-spot' locations, particularly around polluting plant and partially enclosed areas where emissions are high, such as 'canyon streets'.

Taking account of the limit of applicability of the Framework Directive and the interpretation of these limitations in an earlier study on CO and benzene, some types of location and exposure routes are excluded from the analysis:

- The workplace (for which the appropriate instrument is legislation on health and safety at work).
- Tunnels and enclosed car parks (excluded because the amount of time that members of the public spend in either place is likely to be small, and hence for which associated cumulative exposures are low).
- The home, to the extent that exposures there are influenced directly by indoor sources, including smoking and residential use of solid fuels. This exemption excludes assessment of the impacts of indoor exposure to PAHs through the ingress of air from the outdoor environment.

### 1.3 OVERVIEW OF THE METHODOLOGY

The methodology for this study is outlined in Figure 1. The main analysis carried through the chapters of the report concentrates on the development of best estimates of emissions, future concentrations, and the costs and benefits of meeting the possible targets under assessment. At each stage of the analysis uncertainties are identified, and an indication of their potential magnitude provided.



**Figure 1. Key stages of the cost-benefit analysis.**

The overall format for the uncertainty analysis follows that used in the earlier study on CO and benzene :

1. Identify sources of uncertainty
2. Identify which uncertainties are likely to have the most important impact on the results of the study
3. Assign ranges to the main uncertainties
4. Apply ranges through the analysis to establish potential error in the results.

## 1.4 STRUCTURE OF THIS REPORT

The report is structured as follows:

- Section 2 provides a short overview of PAHs, their chemical and physical characteristics and effects.
- Section 3 describes the approach followed to come to the 2010 emission projection according to our best estimate. It describes available data as well as the steps undertaken in order to update the projection methodologies.
- Section 4 provides details of the concentration modelling including a description of the model, the phases of analysis, projected concentrations and comparison with measured values.
- Section 5 provides details of the options for emissions abatement and of the estimated costs of meeting each of the five proposed air quality targets.
- Chapter 6 provides a description of the effects of human exposure to B[a]P together with details of the analysis methods used to estimate the benefits of attaining each of the five proposed air quality limits;
- Chapter 7 provides details and discussion of the comparison of the estimated benefits of attaining the air quality limits with the estimated costs; and
- Chapter 8 provides details of the overall conclusions from the study.

Additional data are provided in a series of Appendices.

## 2 Polycyclic Aromatic Hydrocarbons

### 2.1 CHARACTERISTICS

Polycyclic aromatic hydrocarbons consist of two or more benzene rings which may be joined in different configurations and which comprise carbon and hydrogen atoms only. PAHs are generated by both natural and anthropogenic sources. The former includes volcanoes and natural fires. However, anthropogenic sources dominate with PAHs being formed in all processes that either combust or carbonise carbonaceous fuels (coal, oil, wood, etc.). There is very limited manufacture of PAHs for use in a small number of products, such as some dyes, and for research and analytical purposes. These sources are believed to add an insignificant amount to the emission inventory and are not considered further in this assessment.

Formation of PAH compounds principally occurs during incomplete combustion through oligomerisation and cyclisation of acetylenes. These reactions occur in the gaseous phase. However, PAH compounds are predominantly found in the particulate phase under ambient conditions, attached to dusts or fine particles (UBA, 1998). The extent of the association of PAH compounds with particulate matter varies with individual compounds, the nature of the particles (e.g. size, surface area, chemical properties etc) and, most importantly, with temperature. The semi-volatile property of PAHs makes them highly mobile throughout the environment via deposition and re-volatilisation between air, soil and water bodies. Therefore, a proportion of PAHs released into the atmosphere are deposited in the oceans and/or undergo long range transport making them both a local and global environmental problem.

Recognising that PAHs do not occur in isolation, but in mixtures, along with other products of fossil fuel combustion, the process of identifying the hazards posed by emissions from individual PAH compounds remains complex and subject to significant uncertainty. Even today IARC (the International Agency for Research on Cancer) recognises certain PAHs only as 'probable'

An additional complication is the contribution of PAH derivatives to overall health effects. In particular, nitro-PAH derivatives that may be readily formed during combustion processes have been shown to induce leukaemia and tumours of the mammary glands and the colon in laboratory animals. The extent to which these derivatives contribute to the overall PAH mixture and carcinogenicity will vary between sources and are the subject of debate. It is not currently possible to quantify associated effects in the absence of data linking impacts to concentrations and exposure.

Human exposure to PAH compounds principally occurs through inhalation of contaminated air, dermal contact and ingestion. The principal effect from exposure via the ambient air is regarded as lung cancer, though there are also links to skin cancers and bladder cancer. There is currently insufficient data to characterise the risks of ingestion of PAH deposited on agricultural land.

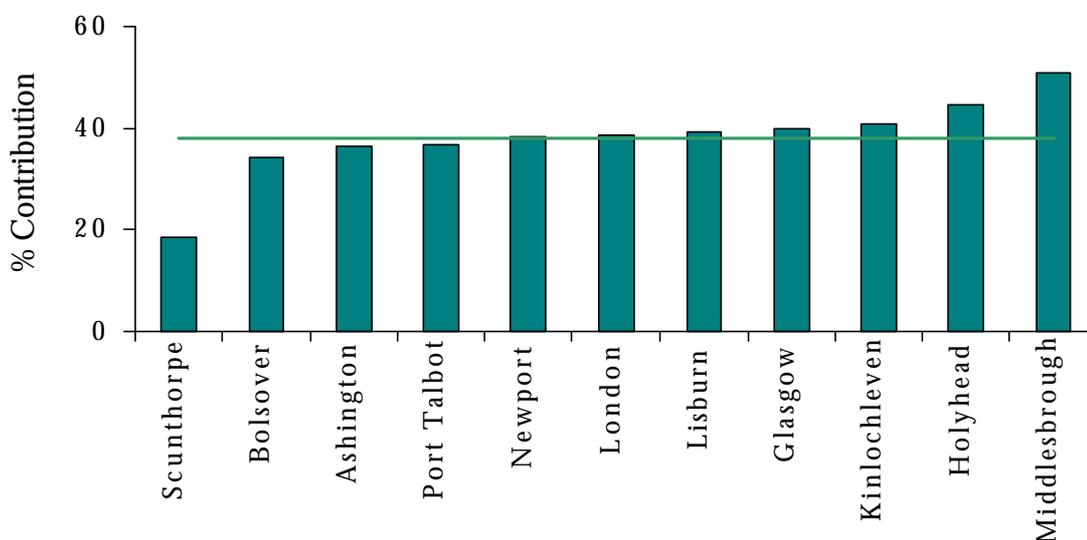
## 2.2 B[A]P AS A MARKER

The fact that exposure to PAH compounds is always to a mixture, rather than single compounds in isolation, has important implications for setting air quality standards. Following the recommendation of the PAH Working Group, B[a]P is adopted in this study as a marker for the carcinogenicity of the overall PAH mix. Provided that this is valid, an air quality standard for emitted PAH mixtures may be expressed in terms of a standard for B[a]P alone. The Working Group justified this on the grounds that available monitoring data from a variety of locations show that B[a]P makes a consistent contribution to the total carcinogenic activity of the overall mixture of PAHs. However, there are three main problems with analysis in this area:

- A lack of consistency in monitoring methods (an issue to be returned to later, when considering the extent to which modelled data may be verified).
- A lack of consistency in the range of PAHs monitored by individual studies.
- Different views on the carcinogenicity of different PAHs relative to B[a]P.

The first two problems mean that there are relatively few studies for which results can be considered truly comparable.

Figure 2 illustrates the basis of the Working Group's assumptions, drawing on measured average concentrations of benz[a]anthracene, B[a]P, dibenzo[ah]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, indeno[1,2,3-cd]pyrene and chrysene at a variety of UK sites over a ten-year period. It shows the contribution of B[a]P to the total carcinogenic activity associated with these 7 compounds to be between 18% and 50%, but mostly around 40%. Although the data shown are all from the UK, they do represent a wide range of locations, covering roadside, industrial, and urban and rural background sites.



**Figure 2. Contribution of B[a]P to carcinogenic potency of PAH mixtures at 11 sites around the UK.**

Broadly similar figures have been reported elsewhere. Petry *et al* (1996) estimated the relative contribution made by B[a]P in mixtures at various factories to be between 27% and 67% of total

carcinogenicity. From Sweden (Larsen and Larsen, 1998), estimates of between 50% and 58% have been derived. Calculations by the Working Group (Maynard, 2000) using data from a study on conditions in a Canadian aluminium smelter (Farant and Gariépy, 1998) suggested that B[a]P contributes 49% of the carcinogenic activity of PAH exposure. The estimate of 40% therefore looks reasonable, within an overall range of 18% to 67%. One exception is the study from Canada by Meeks *et al* (1994) which reported a greater contribution from B[a]P- between 70% and 100% of total PAH carcinogenicity. However, although the range is different to the other studies, the result simply strengthens the case for B[a]P as a marker, on the grounds that it forecasts a more important role for B[a]P.

As noted already, these studies are limited through measurement of only a subset of PAH compounds, and by the respective authors' views on carcinogenicity. The recent review for the Swedish Environmental Protection Agency (Bostrom *et al*, 1999) highlighted this problem. Here, we have considered the variation in carcinogenicity of PAH compounds as reported in the literature, and a larger number of compounds, seventeen in all. The analysis proceeded by quantifying the ratio of each compound (PAH<sub>i</sub>) against B[a]P from a variety of studies of PAH levels in urban areas, taking data from 16 studies conducted in 10 countries, as reported by Bostrom *et al* (1999). The average for the ratio PAH<sub>i</sub>/B[a]P across the series of studies was then calculated for each compound. Inspection of results at this stage demonstrated very significant variation between studies, sometimes by 2 orders of magnitude. These average values were then combined with maximum, minimum and mean values (respectively) for carcinogenicity reported by Chu and Chen (1984), Clement (1986), Nisbet and LaGoy (1992), RIVM (1989), Collins *et al* (1998), Meek *et al* (1994), Muller (1997) and Larsen and Larsen (1998) (see Table 1). Inspection of the table identifies major differences between maximum and minimum values for this element of the analysis, also.

Final results are shown in Table 2, which identifies total carcinogenicity by compound and state in the ambient air. For the column headed 'max', maximum reported values of carcinogenicity for each compound relative to B[a]P are used, in which case B[a]P accounts for only 5% of overall carcinogenicity. Taking minimum values for carcinogenicity of other PAHs relative to B[a]P, the contribution of B[a]P rises to 41%. The figure is 13% where mean values are used. These results suggest that the contribution of B[a]P may not be so great as previously estimated, at least for the mainly urban locations considered in the analysis. The contribution of B[a]P would fall further were more of the dibenzopyrenes to be included.

Table 2 provides opportunity to consider whether an emphasis on B[a]P would lead to disproportionate attention being paid to those abatement methods that reduce exposure to particle-bound PAHs, but not to all PAHs. For example, emissions of these particle-bound species would be reduced through the use of particle traps on road vehicles, but emissions of others, such as fluoranthene, would not.

**Table 1.** Estimated carcinogenicity of different PAH compounds, drawing on results from numerous studies (see text).

	Carcinogenicity relative to B[a]P		
	max	min	mean
Anthanthrene	0.32	0.28	0.3
Anthracene	0.01	0	0.0026
Benz[a]anthracene	0.145	0	0.067
<b>Benzo[a]pyrene</b>	<b>1</b>	<b>1</b>	<b>1</b>
Benzo[b]fluoranthene	0.14	0.06	0.099
Benzo[c]phenanthrene	0.014	0.005	0.0095
Benzo[e]pyrene	0.004	0	0.002
Benzo[ghi]perylene	0.03	0.01	0.017
Benzo[j]fluoranthene	0.1	0.045	0.061
Benzo[k]fluoranthene	0.1	0.03	0.061
Chrysene	0.89	0.001	0.13
Cyclopenta[cd]pyrene	0.023	0.012	0.018
Dibenzo[ah]anthracene	5	0.69	1.8
Dibenzo[ae]pyrene	1	0.2	0.73
Dibenzo[ah]pyrene	10	1	4.1
Dibenzo[ai]pyrene	10	1	4.1
Dibenzo[al]pyrene	100	1	37
Fluoranthene	0.06	0	0.028
Indeno[123-cd]pyrene	0.232	0	0.091
Phenanthrene	0.01	0.0005	0.0044
Pyrene	0.081	0	0.021

**Table 2.** Contribution of different PAHs to carcinogenicity of monitored mixtures, accounting for state in ambient air and variation in carcinogenicity of different compounds relative to B[a]P.

Compound	Typical state in ambient air	Contribution to total carcinogenicity of PAH mixture		
		Max	Min	Mean
Anthanthrene	Particle bound	1%	5%	2%
Anthracene	Vapour phase	0%	0%	0%
Benz[a]anthracene	Semi-volatile	2%	0%	2%
<b>Benzo[a]pyrene</b>	<b>Particle bound</b>	<b>5%</b>	<b>41%</b>	<b>13%</b>
Benzo[b]fluoranthene	Particle bound	2%	6%	3%
Benzo[c]phenanthrene	Vapour phase	0%	0%	0%
Benzo[e]pyrene	Particle bound	0%	0%	0%
Benzo[ghi]perylene	Particle bound	0%	1%	1%
Benzo[j]fluoranthene	Particle bound	1%	3%	1%
Benzo[k]fluoranthene	Particle bound	1%	2%	1%
Chrysene	Semi-volatile	27%	0%	10%
Cyclopenta[cd]pyrene	Semi-volatile	0%	2%	1%
Dibenzo[ah]pyrene	Particle bound	45%	38%	49%
Fluoranthene	Vapour phase	4%	0%	5%
Indeno[123-cd]pyrene	Particle bound	2%	0%	2%
Phenanthrene	Vapour phase	5%	2%	6%
Pyrene	Vapour phase	5%	0%	4%
Total		100%	100%	100%
<b>Total in vapour phase</b>		<b>15%</b>	<b>2%</b>	<b>15%</b>
<b>Total semi-volatile</b>		<b>29%</b>	<b>2%</b>	<b>13%</b>
<b>Total particle bound</b>		<b>56%</b>	<b>96%</b>	<b>71%</b>

The results show that the particulate-bound PAHs (including B[a]P, but excluding 'semi-volatile' species) are estimated to contribute between 56% and 96% of total carcinogenicity of PAH mixtures. These numbers increase (perhaps to around 70% at the lower end), when the fraction of semi-volatile species bound to particles is added in. On this basis, end-of-pipe measures to abate particles will make a significant contribution to reducing exposure to PAHs (weighted by potency), providing that adsorption to particle surfaces takes place within the system, soon after combustion. To optimise abatement of PAHs, this may influence the siting of particle abatement equipment and conditioning of the exhaust gas stream prior to cleaning.

This analysis omits consideration of conditions at some types of hot-spot where the balance between particle-bound and vapour-phase PAH is markedly different, for example, where emissions are linked to evaporation from (e.g.) wood-preservatives.

The various results shown in this section raise a number of issues:

1. Taking a broader sample of PAH data than has been used in previous analyses, B[a]P makes a contribution to the overall carcinogenicity of reported PAH mixtures of between 5% and 41%. However, it may not be the most harmful PAH compound.
2. Available evidence supports the view that the particulate-bound PAHs, including B[a]P, cause most harm.
3. Better characterisation of PAH mixtures is needed, work that will be greatly assisted by the development of more consistent monitoring protocols.
4. More consensus on the potency of different PAHs relative to B[a]P is needed. There is now a substantial body of literature on the subject providing the opportunity for informed debate.

## **2.3 EFFECTS OF PAHS ON HEALTH AND THE ENVIRONMENT**

Effects are reviewed in more detail in Section 6 of this report, alongside the quantification of the benefits of abating PAHs to varying degrees. In brief:

1. The effect of most concern is elevated incidence of lung cancer, observed in a number of studies, particularly of workers in the aluminium, gas and coke industries. Extrapolation of results from this group to the general population is complicated because of the difference in exposure conditions – high concentration for a period of several years, compared to low concentrations over a lifetime of exposure.
2. Other health effects include increased incidence of skin and bladder cancers, though there is less evidence for these than for lung cancers.
3. There are no quantifiable effects of PAHs on ecosystems at the present time.

## 3 Emission Sources

### 3.1 INTRODUCTION

There are many sources of PAH releases to air. These can be divided into two types, evaporative and thermal. Evaporative sources include the use of products containing coal tar from which PAH slowly evaporates to the air during and after use or application. Traditional sources of this kind include wood preservation using creosote and carbolineum, and tar for roofing and road paving. During the past decade, emissions from these sources have decreased due to restrictions on both use and the PAH content of materials.

Thermal emissions come from a wider range of sources, combustion of solid fuels like wood, peat and coal, and the use of automotive fuels like diesel oil. Important industrial process emissions arise from coke and aluminium production. The results of earlier work on a consistent and complete PAH emission inventory for all known sources covering the whole of Europe, are reported in Berdowski *et al* (1997). These results for 1990 form the basis of the emission estimates and projections used for this study. Table 3 presents an overview of the relative importance of traditional emission sources of PAH on a European scale. This overview has been based on the results of the European inventory study and evaluation of new information on source strengths.

Emission estimates for inventory purposes are usually made for a small selection of PAH compounds. Frequently used groupings are the Borneff-6 (see Figure 3), the EPA-14 and the UN/ECE-4 (POP Protocol). These selections always include B[a]P.

- **benzo(a)pyrene (B[a]P)**
- benzo[ghi]perylene
- **benzo[k]fluoranthene**
- fluoranthene
- **indeno[1,2,3-cd]pyrene**
- **benzo[b]fluoranthene**

*Those PAH compounds shown in **bold** are also covered explicitly in the UNECE Persistent Organic Pollutants (POPs) Protocol*

**Figure 3. The Borneff-6 group of polycyclic aromatic hydrocabons.**

**Table 3. Source matrix for the Borneff 6 PAH in Europe.**

CORINAIR SNAP 1 level	Description	Relevance as source of Borneff 6 PAH
1	Power Generation	Minor
2	Residential, commercial and other combustion	
	Small combustion sources, wood-fired	Major
	Small combustion sources, coal-fired	Major
	Small combustion sources, other fuel	Minor
3	Industrial combustion	Minor
4	Industrial processes	
	Coke production	Major
	Aluminium production (including anode baking)	Major
	Other industrial processes	Moderate
5	Oil and gas industry	Minor
6	Solvent use	
	Wood preservation using coal tar-based products	Major
7	Road transport	
	Road transport, gasoline	Moderate
	Road transport, diesel oil	Major
8	Non-road transport	Moderate
9	Waste management	Minor
10	Agriculture	Minor
11	Nature	Minor

### 3.2 COUNTRY DATA AND ESTIMATES FOR 1990

The starting point for the emission projection is 'The European Emission Inventory of Heavy Metals and Persistent Organic Pollutants' (Berdowski *et al*, 1997). The report presents the results of an air emission inventory for 1990 on the basis of submissions of emission data from the Parties to OSPARCOM, HELCOM and the Convention on LRTAP. The report's

substance list includes PAH as Borneff-6 totals, but in the underlying data a full specification of all six PAH compounds is provided. For the countries, sources or compounds lacking in official submissions, default emission estimates have been prepared to complete the inventory. The inventory covers practically the whole of Europe and the results are available on a 50x50km EMEP grid. The methodology used in the default inventory, including emission factors, is described in Berdowski *et al* (1996). During compilation of the inventory all interim and final results were sent to appropriate national emission experts for comment. The report was intended to be a preparatory step towards the preparation of the European CORINAIR emission inventory. At this moment it is the only complete and consistent emission inventory for PAH.

### **3.3 DATA NEEDED FOR THE EMISSION PROJECTIONS**

#### **3.3.1 Updates of 1990 reference inventory, relevant for 2010**

In order to start from emission data as accurate as possible for the projection for 2010, the 1990 reference inventory was reviewed for the EU15 and the six Accession Countries. A selection was made of the most important sources and a search was carried out for new information. Consistent, complete and high quality information appeared to be available for a number of sources, these being wood preservation, domestic wood burning and the use of coal tar products. For domestic wood burning, the improvements comprise revised wood consumption figures. For wood preservation the methodology for estimating semi-volatile PAH compounds like B[a]P has been revised. The use of coal tar products like tar roofing and asphalt paving is reviewed as a potential emission source. The updated methodology used to estimate emissions in 1990 and 2010 from wood preservation is discussed in Appendix 1, whilst Appendix 2 deals with domestic wood combustion. Appendix 1 also addresses asphalt paving and tar roofing.

#### **3.3.2 Existing 2010 emission projection prepared for EU Priority Studies**

TNO has previously prepared most emission projections for the European State of the Environment projects. In the EU Priority studies (EU, 1998) spatially disaggregated emission projections of B[a]P were prepared for the year 2010. The basis for this projection are the results of the TNO inventory for 1990, described above (Berdowski *et al*, 1997). More detailed information on the projections for the EU Priority Studies can be found in Visschedijk *et al* (1998).

##### **3.3.2.1 Methodology used for EU Priority Projections**

The general approach used to estimate air emissions of substances, per country and per emission source category, is to multiply a specific emission that is characteristic for a certain activity (emission factor) by an activity level. In the projections there are two aspects which influence projected emissions, these being an increase/decrease of the activity (expressed in the activity rate) or technological developments which affect the specific emission (expressed in the emission factor). Sources of information on likely variation of activity levels from 1990 to 2010 were as follows:

- The New Targets and Policies Energy Scenario (NTPS) (Capros, 1999) for development of fossil fuel consumption by country and economic sector. This scenario includes only EU Member States. The default Official Energy Pathway developed by IIASA (Cofala, 1998) has been applied to the other countries in Europe. All major energy-consuming sectors are included in these data.
- Capros (1997) for growth indices for industrial production and other macro-economic developments from 1990 to 2010. Also, for some particular sources (like waste incineration), specific activity scenarios were developed by consortium partners of the EU Priority Studies.

Technological developments generally result from national or international emission regulations, or economically motivated actions with side-effects on emissions. The EU Priority Study took the following developments into account:

- The introduction of the Second Sulphur Protocol for large combustion facilities. This Protocol has some effect on PAH, although it is aimed at sulphur reduction. In the new emission projection for B[a]P, prepared for this project, the existing EU Priority results have been used for large combustion plants (industrial and utilities). The National Emission Ceilings Directive or the Gothenburg Protocol might also have some additional effect on particulate emissions and therefore a tertiary effect on PAHs.
- EU regulations on new transport vehicles up to the EURO 4 emission standards. Although these regulations do not specifically set emission limit values for PAH, they will bring about a reduction of PAH as an important side-effect. In the EU Priority work, it was estimated that the penetration of cleaner vehicles would probably not be uniform in Europe. Three groups of countries were distinguished, each with a distinct implementation schedule. The first group included the EU-15 plus Norway and Switzerland, and was expected to follow the agreed schedule. The second group covered the Accession Countries now under consideration, and would follow Group 1 with an expected delay of 5 years. The third group (all other countries) would follow with an expected delay of 10 years. It is assumed that the emission reduction for B[a]P, like other less volatile PAH compounds, is proportional to the reduction in carbonaceous aerosol as a result of the EURO standards. In the emission projection for B[a]P prepared for this project, the existing EU Priority results have been used. An exception is the UK for which national data from the 1997 National Atmospheric Emission Inventory Projections are used.

More detailed information on the methods used in the projections for the EU Priority studies can be found in Visschedijk *et al* (1997).

### **3.3.2.2 Methodology for the 2010 projection**

Since the publication of the State of the Environment reports for the EU Priority Study, the list of currently foreseen emission reduction measures for PAH has been extended. New regulations have been taken into account in the new B[a]P projection prepared for this project.

It is assumed that all foreseen and planned emission reduction policies will be implemented according to schedule. The following new emission reduction measures have been evaluated:

- For industry it is assumed that the IPPC directive will be implemented in the EU15 according to schedule. Thus, by 2010, Best Available Technology (BAT) will be fully

implemented and all industries under the IPPC directive must meet stringent emission standards.

- A second important development will be the entry into force of the UN/ECE POP Protocol. Assuming ratification of the Protocol in 2003, the Protocol would enter into force in 2007 and hence, by 2010 all major industries in all of UN/ECE Europe should have implemented BAT as prescribed by the Protocol. The Protocol includes a range of emission reduction measures regarding the production of aluminium using the Soderberg process and coke and restrictions on the use of products like coal tar or creosote and carbolineum in construction and by consumers. Also for transport vehicles measures comparable with the EURO 4 standards are defined in the UN/ECE POP Protocol.

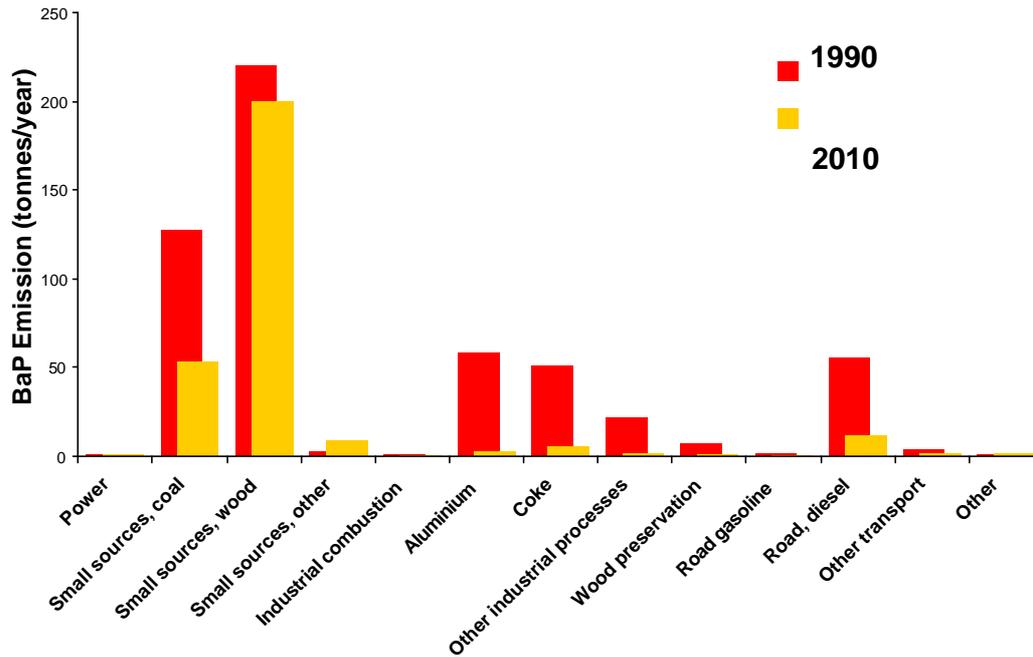
For transport the results of the existing EU Priority results have been used for this source category since these results are considered still valid. An exception concerns the UK for which national data from the 1997 National Atmospheric Emissions Inventory (NAEI) Projections are used.

For domestic wood combustion and for wood preservation new projections have been prepared for the 2010 projections which are based on the revised 1990 levels for these sources, as described in Appendix 1 and 2.

No account has been taken of reduced emissions resulting from the earlier daughter Directives, as details are not yet available on either the extent of limit value exceedences or on the measures to be introduced to deal with any problems. This legislation may have important implications for the need to control PAH levels particularly in those small towns and villages where solid fuels are commonly used.

### **3.4 RESULTS OF THE 2010 EMISSION PROJECTION**

Figure 4 presents the projected emission in 2010 of B[a]P by source category summed for the EU15 Member States and the six Accession Countries. Estimates for 1990 are also given for comparison.



**Figure 4. Projected emission of B[a]P (tonnes/year) in 2010 (orange bars) versus 1990 (red bars), by sector, for the EU15 and 6 Accession Countries**

One of the most striking features of these results is the very high emission calculated for domestic wood consumption in both 1990 and 2010. The most recent estimates by EUROSTAT for the consumption of fuel wood are much higher than previous reports (e.g. by the International Energy Agency or the Food and Agriculture Organization). Even with high penetration of emission optimized stoves in 2010, wood burning remains the dominant B[a]P emission source, since activity levels probably do not decrease significantly. This emission is caused by many small sources that are rather evenly distributed over the area under study. There are no large point sources for domestic combustion and the emission is not concentrated in urban environments (see Appendix 2).

Besides wood burning, residential coal combustion is an important source in certain areas. However, according to the NTPS energy scenario, domestic coal use will be much lower in 2010 compared to 1990, hence the significant reduction that can be observed from 1990 to 2010. It is assumed that the spatial distribution of this source follows population density, which might result in higher modeled concentration levels in urban environments.

Towards 2010 there is some increase in the domestic consumption of renewable fuels other than wood (various other bio-fuels) which leads to higher emissions from this source compared to 1990.

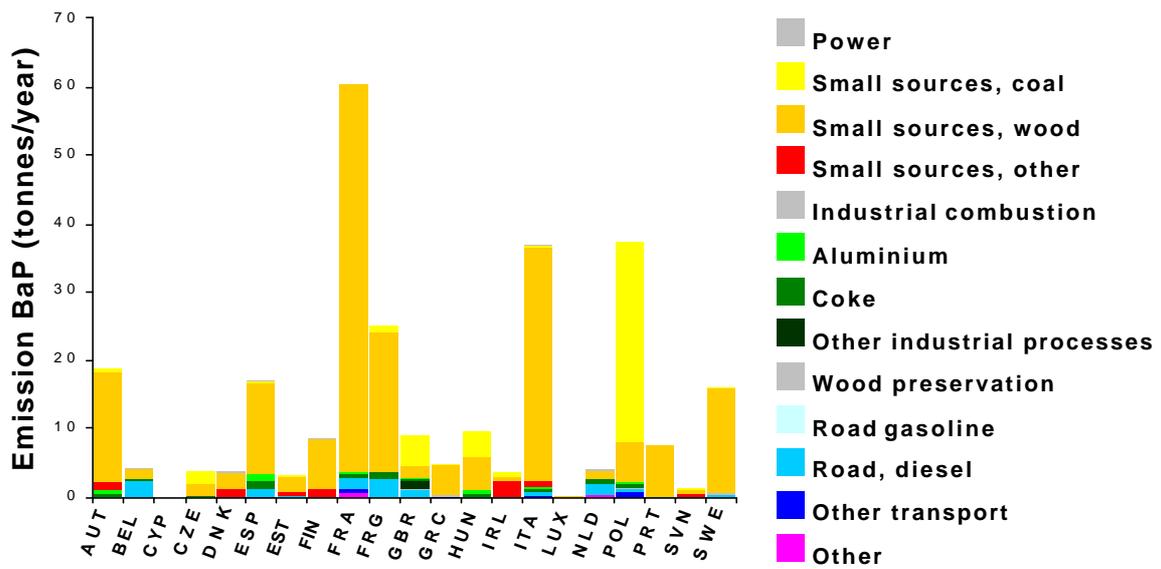
The more 'traditional' B[a]P sources become considerably less important by 2010, mainly as a result of emission reduction policies. Although road transport (especially freight traffic) is projected to grow significantly, the implementation of the EURO emission standards has a drastic effect on tailpipe emissions. It is expected that the emissions from this source will be concentrated in densely populated areas, therefore they are distributed according to population density.

The largest reductions in emissions can be observed for industrial sources like the Soderberg-based aluminium industry and coke production. By 2010 it appears likely that there will be no Soderberg plant left in the EU15. In the last 10 years most of the Soderberg plant operating in this region have been closed or adapted, and it is believed that there are similar plans for the remaining plant of this type. The position is not so clear for the Accession Countries, though it would appear likely that they will follow a similar trend. For coke, production levels are projected to decline somewhat compared to 1990 through reduced demand and an increase of imports. The major part of the emission reduction is due to the implementation of the IPPC directive and the UN/ECE POP Protocol for these sources. Emissions from other industrial sources (including road paving with asphalt) are projected to decrease markedly. Note that these industrial emissions are located in only a few areas. Local elevation of ambient B[a]P levels in the direct vicinity of these sources can therefore still occur in 2010, in spite of emission reduction measures.

Emissions of B[a]P from wood preservation are also predicted to fall markedly by 2010 due to a decline in product use and the different formulation of new creosote oils. This is, however, not necessarily the case for other PAH species; the emission of more volatile PAHs like fluoranthene is estimated to remain above 600 tonnes in 2010.

Figure 5 shows the projected emission of B[a]P in 2010 by country and sector, and again reflects the dominance of domestic wood burning. This source dominates the picture for Austria (which is confirmed by their latest independent national emission inventory), Spain, Estonia, Finland, France, Germany, Greece, Italy, Portugal and Sweden. In several other countries domestic coal combustion is the largest single contributor by 2010, these being the UK and Poland, and to lesser extent the Czech Republic and Hungary. In some countries other domestic fuels are important, like peat in Ireland and Finland. The more traditional sources like industry and road transport decrease in importance for all countries, as a result of the stringent emission reduction policy for these sources towards 2010.

Finally a remark is necessary on the high uncertainty of these estimates. PAH emissions are sometimes difficult to measure accurately and emission rates often vary highly between measurements for comparable sources. It is always difficult to make representative general statements based on these data.



**Figure 5. Projected emission of B[a]P (tonnes/year) in 2010, by country and by sector.**

## 4 Concentrations of B[a]P and Assessment of Exposure

### 4.1 AVAILABLE MEASUREMENTS OF CONCENTRATION

#### 4.1.1 Current ambient concentrations

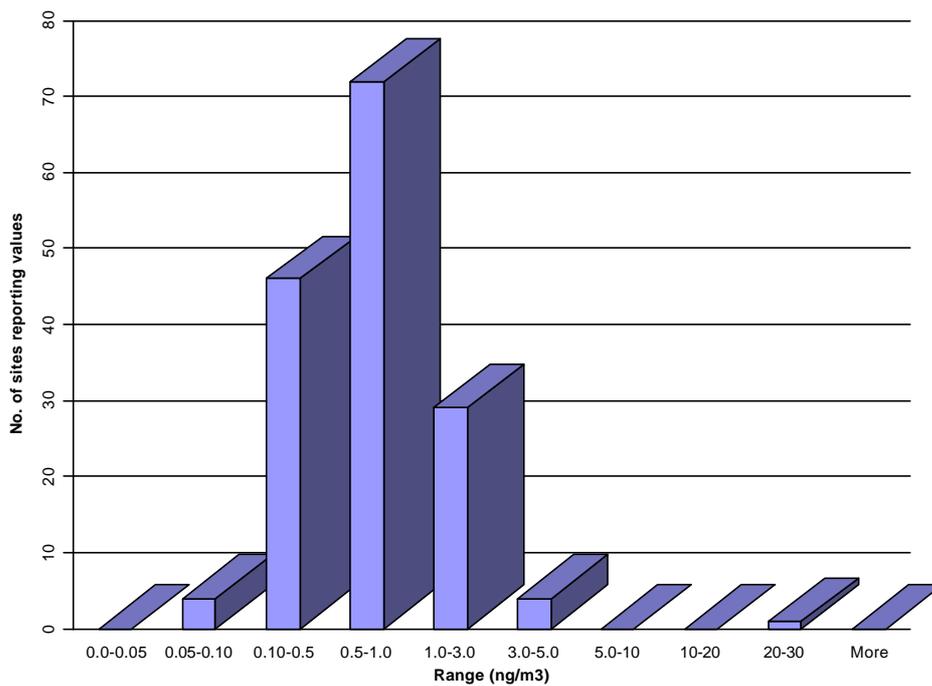
The Commission's Working Group on PAHs sent a questionnaire to all 15 Member States of the EU, the Accession Countries, Switzerland, Norway, Iceland and European industry associations to collect data on measured PAH concentrations, monitoring and analytical techniques and regulatory limits.

The countries initially reporting B[a]P data for 1998 are summarised in Table 4, together with the number and location of monitoring stations. An analysis of the distribution of B[a]P concentration values is graphically presented in Figure 6. This indicates European ambient concentrations of B[a]P are in the range 0.06 to 23.2 ng.m<sup>-3</sup> with a peak in values at or around 0.7 ng.m<sup>-3</sup>. The highest concentrations are restricted to certain industrial sites. Although this initial analysis provides a rough guide to the range and order of magnitude of B[a]P levels that may be expected across Europe, it should be regarded with some caution as only four Member States are represented. A particular problem arises because of a lack of data from Accession Countries. Another issue concerns the reliability and hence comparability of different monitoring methods.

**Table 4. Number and type of site at which B[a]P levels are reported by country for 1998.**

Country	Urban Background	Roadside	Industrial	Unknown or other	Total
Belgium	1	2	1		4
Germany	23	6	9	98	136
Netherlands	5	3	2	2	12
United Kingdom	2			3	5

Highest measured concentrations (23.2 ng.m<sup>-3</sup>) for 1998 were found around a cokery situated at Duisburg-Bruckhausen in Germany, though this is expected to close before 2010, the target year for this study. High concentrations were also reported around the Kinlochleven aluminium smelter in the UK, but this has recently closed, along with many other smelters based on the Soderberg process.



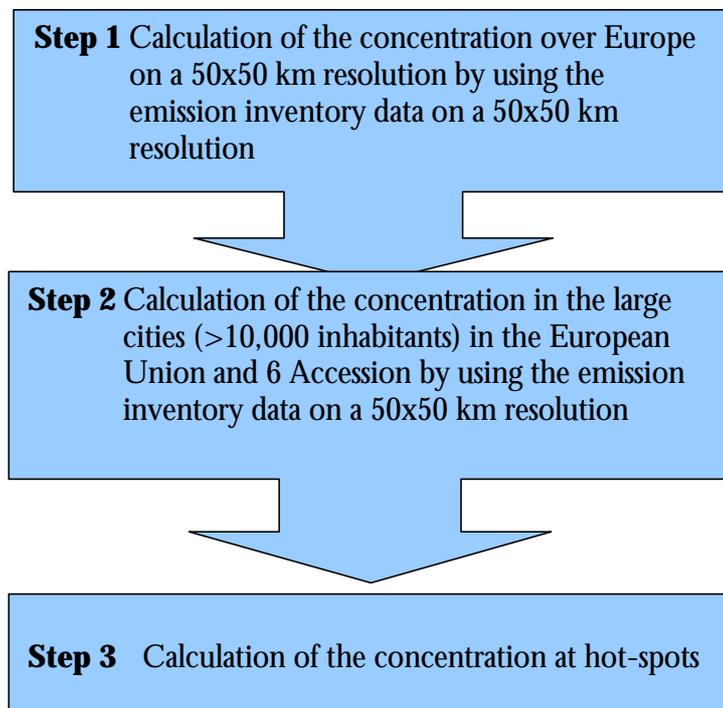
**Figure 6. Frequency distribution of B[a]P measurements for 1998 across selected European Countries.**

Additional review of some older data covering Denmark, the Netherlands, the UK and a remote site at Spitzbergen in Norway shows the following:

- Data for roadside levels in Copenhagen in the first quarter of 1992 show B[a]P levels of 4.4 ng/m<sup>3</sup>. Levels of a number of other PAHs were found to be similarly elevated. It is anticipated that levels will since have fallen through the introduction of cleaner vehicle technologies, though of course this must be balanced against possible traffic growth.
- The Dutch data comprise measurements for rural areas (representing the regional background, 0.05-0.15 ng B[a]P/m<sup>3</sup>), urban areas (both without and with direct heavy traffic around, 0.2-0.5 and up to 0.7 ng B[a]P/m<sup>3</sup>, respectively) and areas in the direct proximity of important industrial sources (e.g. coke plant, 0.2-0.5 ng B[a]P/m<sup>3</sup>).
- Urban background and semi-rural concentrations of B[a]P for the UK tend to be in the region of 0.2 to 1 ng/m<sup>3</sup> (from Coleman *et al*, 1999). Concentrations had fallen from 1991, when the range was broader, 0.3 to 2 ng/m<sup>3</sup>. The decline in this period is partly due to the use of catalytic converters and improved combustion efficiency in cars. Similar trends are visible for some other species of PAH, but not all, suggesting variation in speciation by source. Roadside concentrations tend to be higher (about a factor 2 for B[a]P, though this is based on only 1 roadside location), though again there is extensive variation between species.
- Data for Spitzbergen in 1995 provide illustration of concentrations at remote locations. Results were as follows – 0.009 ng/m<sup>3</sup> for B[a]P, 0.002 ng/m<sup>3</sup> for dibenz[a,h]anthracene, 0.03 ng/m<sup>3</sup> for benzo[b]fluoranthene and 0.01 ng/m<sup>3</sup> for indeno[1,2,3cd]pyrene.

## 4.2 MODELLING METHODS

Calculation of exceedence of possible target values follows three steps:



**Figure 7. Key steps in estimation of concentrations.**

In **Step 1** an overall concentration map of B[a]P over Europe was calculated at a 50 x 50 km resolution. This omitted local scale concentration patterns, so a calculation for all cities with more than 10,000 inhabitants in the study area was performed (**Step 2**). In this calculation specific characteristics, such as roughness length, were altered in order to get better estimates for urban concentrations. The final step was the calculation of the concentration of B[a]P at hot-spots.

### 4.2.1 Background Concentrations

The model used in this study for background concentration assessment was EUTREND version 1.17. This covers the whole of Europe including the surrounding seas, and calculates concentration and deposition as functions of surface characteristics. The model can describe both short and long-distance transport by combining a Lagrangian trajectory model with a Gaussian plume model. Advection is based on meteorological data (6-hour time step, 1000 and 850 hPa pressure levels) obtained through the Netherlands Meteorological Institute (KNMI) from the European Centre for Medium range Weather Forecasts (ECMWF) in Reading, England. The basic resolution of these data is 1° longitude by 0.5° latitude. Because the model is here used on a 50x50 km resolution (see below), some additional uncertainties are introduced, though these are likely to be small compared to some of the other problems encountered in the study. Small scale processes such as dispersion, dry and wet deposition are described on the basis of surface observations of wind speed, cloud cover, temperature, humidity and precipitation. These surface

observation data are obtained mainly as 6-hourly values from databases kept by ECMWF, the American National Centre for Atmospheric Research (NCAR) and the Deutsche Wetter Dienst (DWD). The spatial resolution of these small-scale processes is limited by the (local) density of the meteorological stations. For Europe as a whole, data from more than 1300 stations are available. For practical purposes the EUTREND model distinguishes 50 'climatological areas' in Europe, but the model can also be used on a sub-European scale with the same number of climatological areas.

Atmospheric processes included in the model are dispersion, advection, chemical conversion and wet and dry deposition. An important aspect with regard to dispersion and advection is that the model describes long range transport using well mixed trajectories while local transport and dispersion is described using a Gaussian plume model. The latter model describes the air concentration as a function of source height and meteorology related dispersion parameters but - in the case of high stacks - it also allows for (temporarily) transport of pollutants above the so called mixing layer. Dry deposition is modelled using the so-called resistance approach, in which the dry deposition velocity is constructed from resistances of both the atmosphere and the receiving surface.

Chemical reaction rates are used in the model independently of concentration levels, which means that only linear chemistry can be described. A general description of the modelled processes is given in Van Jaarsveld (1995). The model can be applied with a variable spatial resolution, using a fixed receptor grid or using a set of individual receptor points. In such a case each receptor point is characterised by its co-ordinates, land use class and roughness length.

#### **4.2.2 Types of hot-spot**

Although the concentration model described above provides a solid basis for the assessment of hot-spots by identifying regions where exceedences are likely to occur, it does not have the resolution to specifically model localised hot-spot concentrations within the region. Therefore further analysis is required. In the context of this study a hot-spot is defined as a geographic area where an exceedence of any of the air quality target values under investigation occurs. Given the diverse nature of the sources of PAHs, four types of hot-spot can be defined:

- industrial sites (where the emission sources responsible are dominated by large industry),
- traffic (where the emission sources responsible are dominated by traffic),
- domestic (where the emission sources responsible are dominated by use of solid fuels in houses and other small premises), and
- mixed sites (where no single source type is dominant).

#### **4.2.3 Modelling approach used in assessment of concentration and exposure for industrial hot-spots**

Emissions from industrial sources fall into two broad categories, fugitive emissions (e.g. release of PAHs from coke oven seals) and stack emissions (i.e. release via a chimney). This has important implications for the abatement of pollution - of the two types of release the latter provides ready opportunity for gas cleaning, whilst the former does not. The analysis of the hot-spots therefore proceeds through the following stages:

- Identification of the type(s) of industrial plant giving rise to the hot-spot;
- Consideration of whether plant are likely to continue in operation to 2010;
- Identification of the size of the plant and projected emissions;

- Review of data on actual concentration measurements around sites, where available and scale by projected emissions to sites where data are not available;
- Extrapolation of concentration data against distance in order to assess the variation of PAH concentration away from the site; and
- Assessment of the probable distribution of affected populations with respect to plant specific concentration fields.

Assessment of emissions in the previous sector found that the major industrial hot-spot areas are likely to be linked to coke and aluminium production. Firstly, the concentration distribution of a model plant was calculated, by constructing a source receptor matrix by means of the OPS model (Van Jaarsveld, 1995). The OPS model is a national version of the EUTREND model, described above. Calculations were performed using average meteorological circumstances and specific input parameters for the different source categories. These input parameters are listed in Table 5, which includes data for domestic sources also, given that the methods used in the analysis were similar.

**Table 5. Input parameters for the source receptor matrix calculations.**

Category	Parameter	Value
Wood/coal stoves	Emission	$1.28 \times 10^{-6} \text{ g B[a]P.s}^{-1}$
	Heat content	7.5 kW
	Source height	6 m
Industry – Aluminium	Emission	$0.0175 \text{ g B[a]P.s}^{-1}$
	Heat content	0 kW
	Source height	15 m
Industry – Cokeries	Emission	$0.0007 \text{ g B[a]P.s}^{-1}$
	Heat content	0 kW
	Source height	10 m

Due to the linearity between emission input and concentration output, calculated concentrations can be scaled according to the actual emissions of the source under consideration. The result of scaling these concentrations will be a concentration distribution resulting from the emission of an individual source. Accumulating these concentration distributions provides the total concentration due to all the individual sources.

Obviously, there are uncertainties involved in this approach. First of all, the calculated source receptor matrix is based on an average Dutch climatic situation. This will result in an over- or under-estimation of the calculated concentration for other regions. By calculating the source receptor matrix using average source characteristics, other uncertainties may be introduced. Emission strength is taken into account through the linearity of the model, but differences in other characteristics (heat content and source height) can only be introduced by recalculating the source receptor matrix.

Model calculations were made for 'typical' situations. The standard emission rates assumed were based on a 100,000 tonne/year Soderberg-based aluminium plant<sup>2</sup>, and a 1,000,000 tonne/year coke plant (both using BAT for emission control). Emission factors for B[a]P were discussed in the previous chapter and all sources were assumed to be continuously operated in these model calculations.

The dispersion model calculates the B[a]P concentration gradient in all four directions (North, West, East, South) using annual average meteorological data. For each source the concentration dispersion profile has been averaged over one direction. This averaging has been found to introduce an uncertainty of only 2% at most.

Given the pan-European scope of this study it was not possible to take account of topographical variation in and around hot-spots.

Results are shown in Figure 8. The figure shows the expected dispersion profile: high concentrations close to the source, decreasing quickly at greater distances. Ground-level concentrations do not increase with distance from the plant in the near-field region, as relevant emissions are fugitive, rather than from the stack.

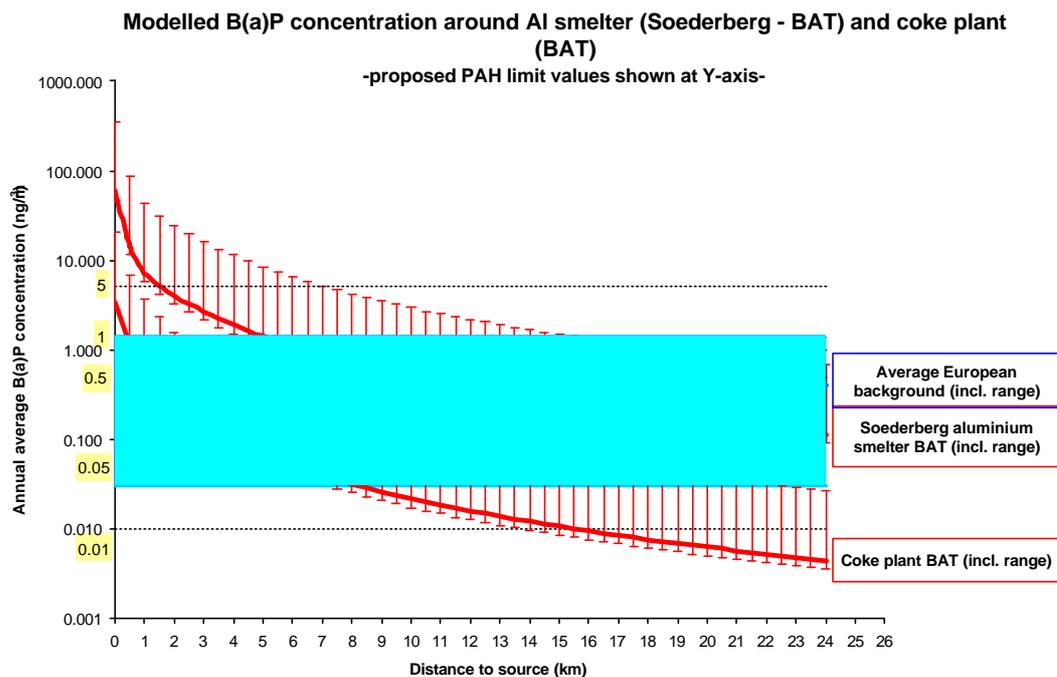
To move from *concentration* to *exposure* of people around plant, each cokery forecast to operate in 2010 was assessed. Dispersion profiles for plant are based on the fact that dispersion profiles are linearly proportional to emission rates (which are known), and hence can be extrapolated from the results shown in Figure 8. Variation in plant capacity is accounted for, though variation in emission factors is not, given a lack of plant specific data. However, the move to BAT will substantially reduce variability in emission factors between plant by 2010.

On this basis, monitoring data obtained by the PAH Working Group for the Kaiserstuhl cokery in Germany, shows reasonable agreement with these results. The data were obtained over a period of three months from a measuring site located about 200 m north of the cokery. The measurement data showed the mean B[a]P concentration to be around 14 ng.m<sup>-3</sup> with individual measurements ranging from 0.07 ng.m<sup>-3</sup> to 75.2 ng.m<sup>-3</sup>: not surprisingly, variation was largely a function of wind direction.

The concentration profiles for all individual plant were subsequently averaged over distance, and translated into "spheres of influence" with a certain radius and a certain average B[a]P concentration. Or, alternatively, spheres of influence are calculated in which the plant-induced concentrations are either lower than 0.01 ng/m<sup>3</sup>, between 0.01 and 0.05, 0.05 - 0.5, 0.5 - 1, 1 - 5 and finally above 5 ng/m<sup>3</sup>.

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<sup>2</sup> Accepting, of course, that these plants have and are being phased out, it is still useful to know whether any that may continue to operate in the EU21 to 2010 would cause problems.



**Figure 8. Dispersion profiles of B[a]P from a typical coke and a Soderberg-process aluminium plant.**

These data are combined with a population map to estimate population density (inhabitants/km<sup>2</sup>) around each plant, and then to estimate exposure. It is assumed that indoor/outdoor concentrations arising from industrial sources of B[a]P are equal, following IEH (1999).

Given the likelihood of closure or modification of the remaining Soderberg plant, a slightly simpler approach has been used for emissions from the aluminium sector (see below) though the general details remain the same.

#### 4.2.4 Modelling approach used in assessment of concentration and exposure for domestic hot-spots

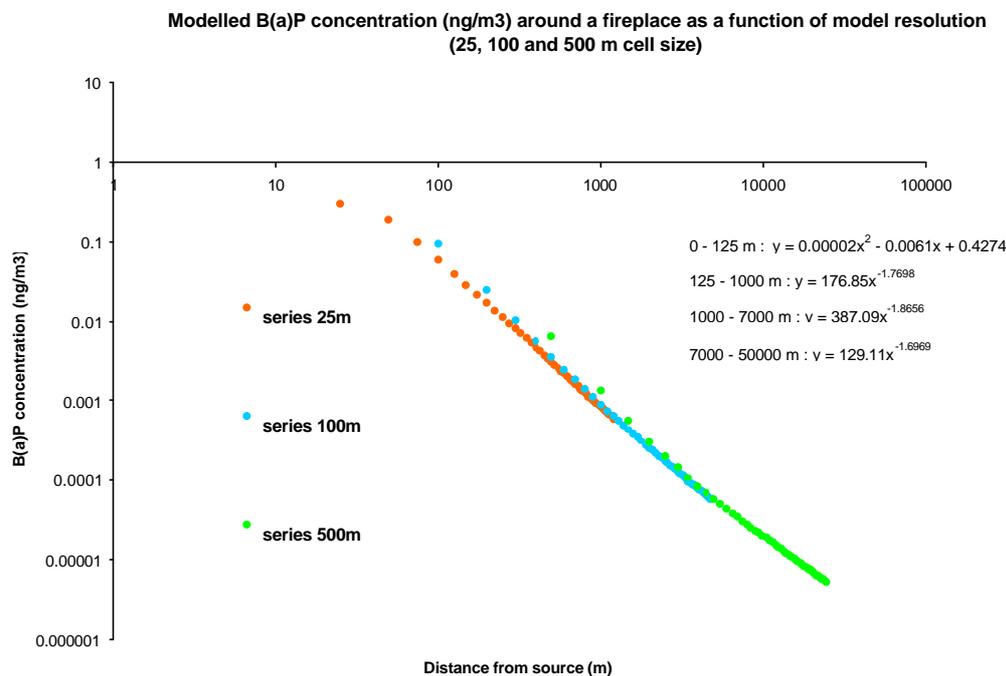
Analysis of domestic hot-spots proceeds through the following stages:

- Assess typical domestic emissions;
- Evaluate how many emission sources/homes within a given area are required to produce an exceedence for each of the target values under assessment;
- Scale emissions for hypothetical villages of different sizes; and
- Evaluate exposed populations.

It is assumed that the use of wood largely takes place in rural areas, away from large cities. Within cities other fuels dominate (natural gas and electricity). Further details are given in Appendix 2.

The general modelling approach was similar to that defined already for the industrial hot-spots, and indeed, Table 5 above contains details relevant to the domestic sources. Figure 9 shows the results of three model runs on concentration assessment around houses with wood fires, each run using an increased grid resolution. These runs were combined in order to model the concentration close to the source as well as further removed. An empirical regression function with a four-section domain was fitted through the model results.

A single stove will of course only give rise to a very modest increase in B[a]P concentration. However, the combined contribution of the many stoves and fire-places present in a village or town can be significant. The significance of these high concentrations is further emphasised since they naturally occur within the village boundaries, with the effect that personal exposures may be very high.

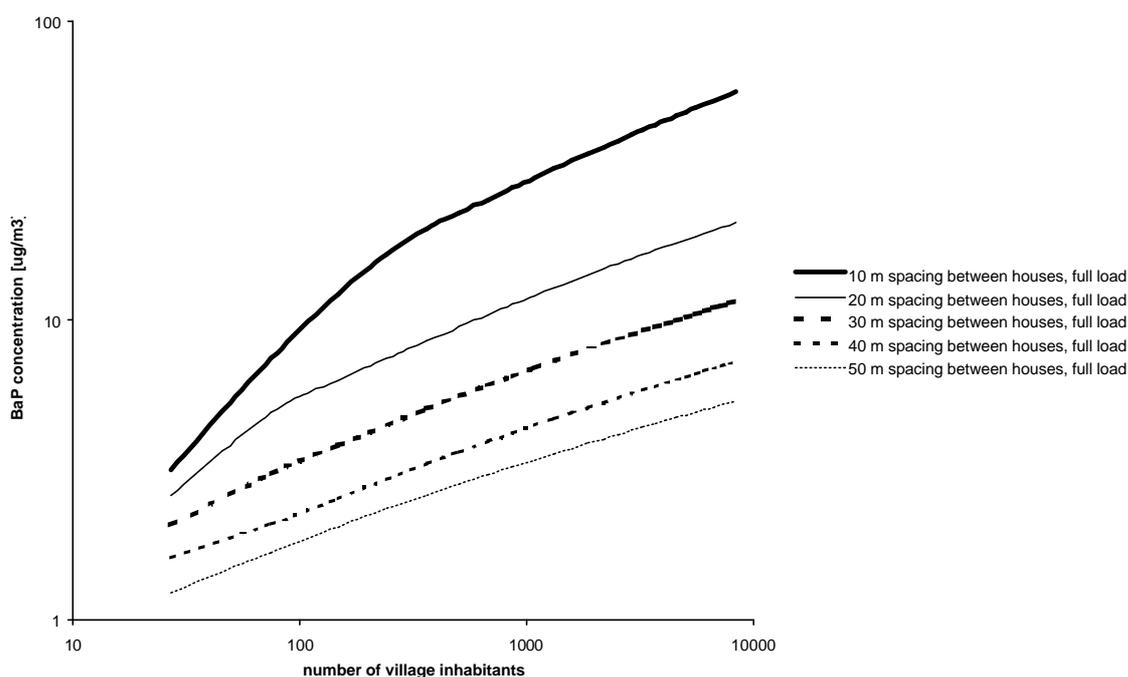


**Figure 9. Dispersion profile from a typical residential combustion situation.**

The combined contribution of many stoves has been calculated as a function of village size. For this a rectangular grid was defined, the dimensions of which were extended step by step, starting at 3x3 (9 houses, 27 inhabitants) going all the way up to 60x60 (approximately 3,000 houses, 10,000 inhabitants). For small towns of this size (10,000 inhabitants), the assumptions on specific fuel use for typically rural environments are not believed to be valid, and a different approach is followed. Calculations were based on the derived regression functions, described above. A key element in the outcome appears to be the assumed average distance between the sources, taken here to vary somewhere between 10 and 50m. The effect of this parameter is demonstrated in Figure 10.

The concentrations plotted for each village size comprise an arithmetic average of the concentrations on each place in the grid.

After analysing and evaluating Dutch and some foreign data on the distribution of fireplaces and actual village areas, we have opted for a distance of 30 m as the best estimate for the actual average distance between sources, for rural situations. It is acknowledged that there will be significant variation around this figure in different areas. For example, the distance between the chimneys of individual terraced houses in coal-mining areas of the UK may be as little as 5 m. Figure 10 provides guidance on the possible error introduced to the analysis under alternative assumptions.



**Figure 10. Relationship between the ambient B[a]P concentrations due to residential wood combustion and the village size (number of inhabitants) at different distances between houses.**

The next step is to apply these generic results to the actual situation in the countries involved in this study.

For assessment of exposure, two situations are distinguished for domestic heating: heating in rural environments and heating in urban environments. These situations vary with respect to both population density and the pattern of the specific consumption of wood.

The next step towards assessment of exposure is the translation of the general relation between village size and B[a]P concentration, into specific relations per country, given the reported national fuel wood, coal and peat consumption. It was assumed that the specific coal consumption is identical for both urban and rural situations. Taking the national domestic

consumption of coal or peat from the 2010 energy projections and dividing this through by the number of inhabitants in each country generates the country-specific consumption.

The specific consumption of wood is calculated differently. Based on Dutch surveys, it is assumed that on average the specific wood consumption in rural areas is 10 times higher than in urban areas.

### ***Domestic - rural***

The specific fuel consumption rates, which can be interpreted as average fuel use per inhabitant, are subsequently processed into emission rates by fuel type, using an emission factor. The emission rates are then compared to the emission rates for full load (determined earlier) and a ratio between the two is derived. With this ratio the earlier established dependency of average village concentration with village size can be scaled per country.

Given the relation between village size and average B[a]P concentration by country, it is necessary to apply social geographic data on the actual distribution of inhabitants over residential agglomeration classes. This distribution is estimated for all countries in the studied area, based on data on degree of urbanisation and detailed information on inhabitants in different village size classes in the Netherlands. Whilst the fraction of a country's population that lives in rural environments is country-specific, the assumed distribution is not, and is described as follows:

Scattered	10%
<100	0.20%
100-1000	15%
1000-5000	25%
5000-10000	50%

Given the data on the B[a]P concentrations in villages as a result of wood coal and peat use, and the rural population distribution over different settlement size classes, exposure can be calculated.

### ***Domestic-urban***

A similar approach to that followed for the rural case is applied to estimate urban exposure to B[a]P from domestic burning. There are, however, some differences. Firstly, the specific consumption of wood is only 10% of the rural specific consumption. A second major difference is the average distance between stoves in cities, which is estimated to be 50m instead of the 30m for rural environments (it is lower, given the lower frequency of wood burning in urban areas). It is expected that coal will be relatively more important in urban areas compared to rural. In addition, the multiple source grid simulations described under "Domestic – rural" have been extended up to cities with 10,000,000 inhabitants.

## **4.2.5 Modelling approach used in assessment of concentration and exposure for traffic hot-spots**

Analysis of hot-spots linked to traffic emissions proceeded as follows.

1. Background concentrations of PAH attributable to traffic emissions in 13 UK cities (as shown in Table 6) in 1998 were assessed using high-resolution maps (K. Vincent, AEA Technology, personal communication). These maps had been generated using ADMS-2 Version 2.2 to calculate the average annual B[a]P contribution from vehicular sources on a 1 km receptor grid covering the whole of the United Kingdom. Contributions at each

receptor from traffic sources at distances greater than 15 km in the north-south or east-west directions were ignored. A uniform surface roughness of 1 m, corresponding to typical urban areas was used for the whole country. Wet and dry deposition were ignored. B[a]P emissions for each 1 km square were obtained from the UK's National Atmospheric Emissions Inventory (NAEI). Highest concentrations were found in London, Glasgow and Liverpool. The 13 cities were selected to give a broad coverage in terms of city size and location within the UK. The analysis did not consider other countries at this stage because of a lack of high-resolution data elsewhere.

**Table 6. Population of UK cities for which B[a]P exposure was modelled in detail.**

City	Population (M)	City	Population (M)
Belfast	0.28	Manchester	1.7
Birmingham	1.4	Norwich	0.12
Bristol	0.37	Nottingham	0.26
Edinburgh	0.40	Oxford	0.11
Glasgow	0.73	Plymouth	0.24
Liverpool	1.1	Sheffield	0.45
London	6.4		

- Concentration data were then scaled back by the ratio of emissions in 1998 to emissions in 2010.
- Background concentrations were combined with population using GIS to link population with concentration, and a frequency plot of cumulative population vs. concentration was generated.
- To estimate the degree of abatement required to meet each of the proposed limit values for B[a]P it was also necessary to calculate peak concentrations attributable to traffic. Measurements of PM<sub>10</sub> in the UK suggest a factor 3.0 difference between urban background and road concentrations. Traffic-generated PM<sub>10</sub> was considered a reasonable proxy for B[a]P, given the link between the two and the availability of a reasonable level of data on PM<sub>10</sub> levels.
- Extrapolation from the UK to the rest of the study area accounted for variation in population and B[a]P emissions, the latter using data on population-weighted emissions of PM<sub>10</sub> by vehicle type taken from the reports of the cost effectiveness study of the Second Auto-Oil Programme (AOP-II). Given a lack of data on the Accession countries in AOP-II, the factor used to extrapolate from UK conditions to those countries was based on the average of the factors for Greece and Spain, the two countries with lowest per capita GDP in the EU15 for which extensive data were available in AOP-II. Portugal was not used in this averaging as PM emissions there were extrapolated from Spanish data. Adjustment factors to account for variation in emissions per capita are shown in Table 7.

**Table 7. Adjustment factors for variation in emissions between countries in 2010, based on AOP-II data (UK=1).**

Country	Adjustment factor	Country	Adjustment factor
Austria	0.88	Ireland	0.86
Belgium	1.56	Italy	0.63
Cyprus	0.82	Luxembourg	1.18
Czech Republic	0.82	Netherlands	0.79
Denmark	0.57	Poland	0.82
Estonia	0.82	Portugal	0.94
Finland	1.20	Slovenia	0.82
France	1.94	Spain	1.04
Germany	0.76	Sweden	0.55
Greece	0.61	UK	1.00
Hungary	0.82		

#### 4.2.6 Determination of change in exposure

Overall exposure to PAH is estimated for each grid cell as:

$$P \bullet [C_{industry} + C_{domestic} + C_{traffic} + C_{other}]$$

where  $P$  = population in each grid cell, and  $C$  = concentration in the grid cell attributable to the different sources, as shown.

The assessment of exposure is limited to dose received from ambient pollution. In the earlier study on benzene (AEA Technology, 1999) an exposure model was developed, estimating the time spent by different fractions of the population in different locations at different periods of the day. It is now possible to improve on this, using data from the MACBETH (Monitoring of Atmospheric Concentrations of Benzene in European Towns and Homes) Project conducted under the EC's LIFE Programme. MACBETH sampled benzene as ambient concentrations as well as levels in the houses of 50 non-smoking volunteers in each of Antwerp, Athens, Copenhagen, Murcia, Padua and Rouen using personal exposure monitoring. Final results from MACBETH show that for benzene the ratio of urban concentrations to exposures was 2.00 (FSM, 1999). Urban concentrations were based on a monitoring scheme with 85% of samplers in background locations, 10% in hot-spots and 5% in peri-urban sites.

The use of a factor 2 to move from concentration to exposure was therefore applied in this study for traffic sources (on the basis that benzene in the ambient air is principally a traffic pollutant). Uncertainties relate to the behaviour of benzene and PAHs with respect to ingress from outdoors into homes. Available data suggest that PAH levels (in the absence of significant indoor sources) are roughly equal indoors and out (IEH, 1999). Results from MACBETH, however, found higher benzene levels indoors, by an average factor of 1.51, though this factor varied from 0.49 in Athens to 2.14 in Antwerp. A further complication arises through the 'personal cloud effect', whereby personal exposure of non-smokers to some pollutants has been found to be greater than suggested by ambient concentrations (Monn *et al*, 1997). Overall, the uncertainties in the assessment of exposures from concentration data are unlikely to be as

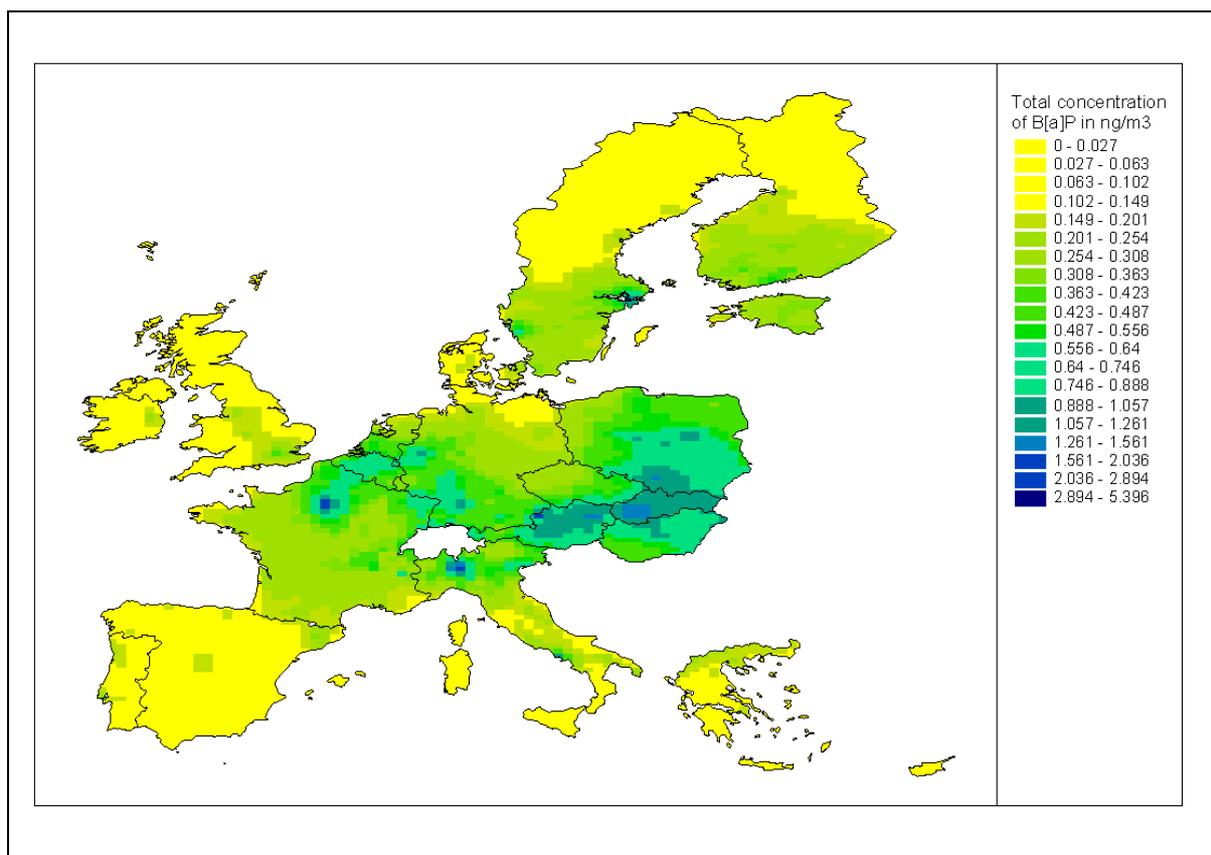
significant as uncertainties in other parameters (for example, the factor 4 in unit lifetime risk factors).

It was not necessary to consider total exposure – in other words to include the contribution from sources that are outside the scope of the Framework Directive on Ambient Air Quality (smoking, work, etc.) because of the assumed shape, linear with no threshold, of the dose-response function for PAHs and other genotoxic carcinogens. With this form of function the change in effect resulting from new legislation will be in direct proportion to the change in concentration/deposition. Total exposure would be important only for alternative shapes of the dose-response function, particularly where thresholds are present.

## 4.3 2010 PROJECTIONS

### 4.3.1 Background Concentrations

Estimated rural background concentrations of B[a]P for Europe in 2010 are shown in Figure 11. Even without considering hot-spots, it is clear from the figure that most of Europe is subject to levels in excess of  $0.15 \text{ ng.m}^{-3}$ , i.e., to levels greater than the lower two possible limit values under investigation here. Background levels are predicted to exceed  $2 \text{ ng.m}^{-3}$  in some locations.



**Figure 11. Rural background concentration of B[a]P across Europe in 2010.**

In Table 8, a comparison is made of the modelling results with some measurements from monitoring sites. From the table it is clear that the comparison for the background sites shows good agreement, while there is less agreement for sites influenced by significant local sources. It should be noted in this comparison that the modelling refers to 2010 while the measurements are from the 1990's (during which emissions were higher than forecast for 2010). Overall it is considered that agreement between the modelled results and measurement data is reasonably good. However, given the uncertainties in estimation of PAH emissions and a lack of consistency in monitoring data, this comparison of results should only be considered as a 'ball-

Table 9 gives an overview of the modelled results for background conditions in the 21 countries separately. The maximum concentration, in most cases, occurs in urban areas. For the categories 'Rural background' and 'Urban background' (background concentrations in urbanised areas, not actual city centres) grid cells from the overall results from Figure 11 were taken that are representative for these specific situations.

**Table 8. Comparison of observations on background levels with modelling results.**

Location description	B[a]P (ng/m <sup>3</sup> ) 1990s observations vs. 2010 background modelling	
	Measurements	Model results
Urban background and semi-rural concentrations UK 1998	0.2 – 1	0.2
Regional background Netherlands 1998	0.05 - 0.15	0.16
Remote background Spitzbergen 1995	0.009	0.009
Road-side level Copenhagen 1992	4.4	1 – 3
Busy street Vienna 1997 – 1998	4.2	1 – 3

Table 10 presents results of background concentrations in towns and cities (defined as having more than 10,000 inhabitants) in the study area. As noted above, these were modelled separately to take more specific account of the micro-meteorological characteristics of cities. Again, however, the results represent averages across broad areas of cities, rather than levels typical of urban centres. The table also shows the total number of inhabitants in these urban areas for the countries and the contribution of the individual PAH emission sources to the city background concentrations. There is quite some variation in urban background concentrations and in contributing sources. For example, for the Netherlands diesel fired road transport is a major source, while in Austria the contribution of residential wood combustion is the most important source.

**Table 9. Statistics on background B[a]P concentration per country (in ng.m<sup>-3</sup>).**

	Average	Minimum	Maximum	Rural background	Urban background
Austria	0.784	0.336	1.574	0.471	1.140
Belgium	0.523	0.221	0.841	0.404	0.650
Cyprus	0.005	0.002	0.008	0.006	0.006
Czech Republic	0.474	0.222	1.196	0.254	0.602
Denmark	0.113	0.006	0.331	0.121	0.230
Estonia	0.233	0.101	0.418	0.159	0.313
Finland	0.146	0.017	0.498	0.030	0.367
France	0.310	0.058	2.894	0.127	1.258
Germany	0.338	0.073	1.041	0.117	0.925
Greece	0.113	0.032	0.391	0.054	0.151
Hungary	0.705	0.368	1.382	0.391	0.984
Ireland	0.065	0.015	0.214	0.040	0.214
Italy	0.251	0.029	2.216	0.061	0.983
Luxembourg	0.382	0.328	0.459	0.355	0.355
Netherlands	0.320	0.116	0.665	0.162	0.490
Poland	0.550	0.134	1.364	0.268	1.250
Portugal	0.122	0.061	0.361	0.061	0.298
Slovenia	0.225	0.133	0.755	0.142	0.326
Spain	0.091	0.012	0.334	0.075	0.193
Sweden	0.144	0.027	1.274	0.030	0.488
UK	0.072	0.006	0.398	0.030	0.398
Overall figures	0.252	0.002	2.894		

**Table 10. Average ambient air B[a]P concentrations and source contributions of city background**

	Total city population (x1,000,000)	Weighted average concentration (ng/m <sup>3</sup> )	% Power plants	% Other transport	% Road transport gasoline	% Road transport diesel	% Wood preservation	% Process emissions other	% Process emission coke	% Process emission aluminium	% Industrial combustion	% Small sources other	% Small sources wood	% Small sources coal	% Other
Austria	2.9	1.1	0.0	0.2	0.0	2.1	3.0	0.0	6.2	1.2	0.0	8.7	72.9	5.5	0.1
Belgium	4.3	0.53	0.9	0.5	0.5	39.0	1.1	0.2	1.8	0.1	0.0	0.4	49.3	4.2	2.1
Cyprus															
Czech Republic	4.8	0.42	0.1	0.5	0.0	3.9	2.1	0.0	1.4	0.8	0.0	1.9	46.2	42.6	0.4
Denmark	2.5	0.19	0.3	1.9	0.1	6.1	2.4	0.2	1.0	0.4	0.0	23.1	55.9	8.2	0.3
Estonia	0.68	0.21	0.0	0.5	0.0	2.9	7.6	0.0	0.5	0.3	0.0	20.0	54.9	13.0	0.1
Finland	2.5	0.36	0.0	0.3	0.0	1.8	3.4	0.0	0.3	0.2	0.0	16.2	73.5	4.2	0.1
France	31	0.99	0.1	1.6	0.1	6.5	2.3	0.1	1.2	0.4	0.0	0.2	83.5	1.5	2.5
Germany	53	0.45	0.5	0.3	0.2	14.3	2.0	0.1	1.8	0.3	0.0	0.7	71.8	7.5	0.5
Greece	5.3	0.15	0.0	3.6	0.2	10.6	16.2	1.4	0.6	0.2	0.0	0.6	46.4	7.1	12.9
Hungary	5.4	0.80	0.0	0.4	0.0	1.9	5.6	0.0	5.5	3.6	0.0	2.8	32.5	43.2	4.3
Ireland	1.3	0.16	0.1	0.4	0.1	4.2	1.2	0.4	0.4	0.2	0.0	63.6	4.4	24.9	0.2
Italy	31	0.60	0.0	0.5	0.1	5.0	5.7	0.0	2.5	1.1	0.0	2.3	80.1	1.9	0.8
Luxembourg	0.10	0.34	0.4	1.0	0.2	15.3	1.5	0.1	1.4	0.1	0.0	0.3	76.1	2.2	1.5
Netherlands	6.9	0.41	7.0	0.6	0.3	30.8	2.1	0.3	9.1	0.1	0.0	0.6	40.8	3.4	5.0
Poland	18	0.76	0.0	2.2	0.0	1.8	2.2	0.0	1.1	0.6	0.0	1.1	20.5	69.9	0.5
Portugal	3.6	0.22	0.0	0.4	0.0	3.4	0.6	0.0	0.4	0.5	0.0	0.0	94.5	0.1	0.0
Slovenia	0.64	0.30	0.4	0	0.2	8.3	0.0	0.0	0.0	0.0	0.0	44.5	32.1	14.4	0.0
Spain	24	0.12	0.1	1.4	0.1	8.1	1.4	0.1	8.8	2.6	0.0	0.1	75.9	1.0	0.4
Sweden	4.0	0.45	0.1	0.2	0.1	4.5	1.4	0.0	0.4	0.2	0.0	0.0	90.9	2.0	0.1
United Kingdom	39	0.22	0.1	0.2	0.5	24.1	1.8	1.9	1.4	0.4	0.0	3.0	25.4	40.6	0.6

### 4.3.2 Hot-spots - cokeries

Table 11 presents the calculated contribution of coke plants in 2010 to local ambient B[a]P concentration. It shows some general data used for the analysis, namely the estimated coke production by country and plant and the emissions, assuming implementation of BAT. The fourth column shows the radius of the circle around a coke plant, where the contribution is 0.3 ng/m<sup>3</sup> or more. The average contribution within these circles is 0.5 ng/m<sup>3</sup> (fifth column). The last column shows the number of people exposed to concentrations greater than or equal to 0.3 ng/m<sup>3</sup>, and how this varies between countries, the largest numbers being for Poland and Germany.

**Table 11. Area around a coke plant, including the number of people living in such an area, where the contribution of coke plant emissions to ambient B[a]P concentrations is at least 0.3 ng/m<sup>3</sup>.**

Country	Estimated production in 2010 (ktonnes)	Average production by plant (ktonnes/yr)	Emission BAT (tonnes)	Radius (km) of circle, where contribution is $\geq 0.3$ ng/m <sup>3</sup>	Average concentration increase within the circle area (ng/m <sup>3</sup> )	Average population density around plants (inh./km <sup>2</sup> )	Inhabitants in circle area around the plants
Austria	1400	700	0.014	2.4	0.5	137	2518
Belgium	4300	717	0.014	2.4	0.5	189	3504
Czech Republic	4300	1075	0.022	2.7	0.5	150	3415
Finland	390	195	0.0039	2.0	0.5	27	355
France	5800	725	0.015	2.4	0.5	338	6297
Germany	33000	3667	0.073	4.6	0.5	401	26518
Hungary	390	390	0.0078	2.2	0.5	75	1131
Italy	5100	1020	0.020	2.7	0.5	275	6064
Netherlands	2000	2000	0.040	3.4	0.5	253	9005
Poland	14000	2000	0.040	3.4	0.5	889	31660
Portugal	180	180	0.0036	2.0	0.5	280	3659
Spain	2600	2600	0.052	3.8	0.5	61	2767
Sweden	870	435	0.0087	2.2	0.5	13	196
United Kingdom	6700	838	0.017	2.5	0.5	456	9089

Table 12 provides a more detailed breakdown of population exposure attributable to cokeries in areas local to these plant. For Austria the second column shows that in 2010 46,891 and 6,507 inhabitants have been estimated to live in an area where coke plants contribute 0.01-0.05 and 0.05-0.5 ng/m<sup>3</sup> respectively to local ambient B[a]P concentrations. The last two rows of the table show the sum of the national results as a percentage of the total population in the study area (i.e. the EU21), a little under 1%. The table shows significant variation between countries.

**Table 12. Number of inhabitants exposed to the contribution of B[a]P emissions by coke plants**

<b>Country</b>	<b>Inhabitants in circle around plant with contribution of 0.01-0.05 ng/m<sup>3</sup></b>	<b>Inhabitants in circle around plant with contribution of 0.05-0.5 ng/ m<sup>3</sup></b>	<b>Inhabitants in circle around plant with contribution of 0.5-1 ng/ m<sup>3</sup></b>	<b>Inhabitants in circle around plant with contribution of 1-5 ng/ m<sup>3</sup></b>
Austria	46,891	6,507	940	68
Belgium	65,877	9,167	1,309	97
Czech Republic	76,806	11,196	1,273	161
Finland	4,500	548	129	2
France	118,957	16,575	2,352	177
Germany	1,052,540	171,238	8,662	4,438
Hungary	17,038	2,213	417	14
Italy	133,123	19,292	2,263	267
Netherlands	272,373	42,298	3,232	868
Poland	957,616	148,713	11,364	3,051
Portugal	45,628	5,518	1,321	15
Spain	94,608	15,021	961	345
Sweden	3,062	402	73	3
United Kingdom	182,641	25,878	3,397	310
Total for study area	3,071,660	474,566	37,692	9,815
Total as % of EU21 population	0.8	0.1	0.01	0.003
Cumulative % of EU21 population	0.8	0.9	0.94	0.94

#### 4.3.3 Hot-spots – aluminium plants based on the Soderberg process

Table 13 presents the calculated contribution of Soderberg aluminium plants to B[a]P burdens. It shows some general data used for the analysis, namely the estimated primary aluminium production by country for each plant and its emissions, assuming implementation of BAT. Compared with cokeries there are relatively few Soderberg aluminium smelters and these are set to decline as they are replaced with the less polluting pre-baked anode processes. As stated elsewhere in this report it is possible that no Soderberg plant will remain in the EU21 by 2010. The average European Soderberg smelter is predicted to produce 47 kt of aluminium per year, emitting around 230 kg of B[a]P in the process, with an affected population of around 125,000.

**Table 13. Estimated production, total B[a]P emissions and exposed populations from Soderberg aluminium plant. Figures represent an expected worst case for 2010, as most, possibly all, of the plant listed are expected to have been closed or upgraded.**

Country	Assumed production 2010 (kt)	Emission (kg/yr)	Population close to the plant
Hungary	34	170	113,000
Poland	39	192.5	79,315
Spain (Plant 1)	57	287	55,758
Spain (Plant 2)	55	273	251,342

Analysis was based on the modelled B[a]P concentration/dispersion curve provided in Figure 8. Those results were extrapolated using predicted production of each of the identified aluminium plants in order to provide a concentration profile (concentration against distance from the point of emission) for each plant. The concentration of B[a]P close to the source is in the order of tens of  $\text{ng}\cdot\text{m}^{-3}$ , but this decreases by around an order of magnitude at a distance of 1 km from the point of release (this distance roughly corresponding with the plant fence in many cases).

Given the likelihood that these plant will all be closed by 2010, the methods used for assessing associated exposure are simpler than those adopted for the cokeries, based on less intensive information on population distribution around each plant. A number of scenarios are considered:

- Absolute worst case, assuming that the total population are exposed to the maximum concentrations at the point of release. This is clearly an unrealistic assumption, but is useful for placing an absolute upper bound on the assessment. If effects estimated using this assumption are found to be large, it is reasonable to conduct a more detailed analysis of population distribution.
- Best estimate, assuming that the total population is resident 4.5 km from the point of release, except where more precise information was readily available.
- Minimum, assuming that the total population are resident 24 km from the point of release (again to scope the analysis).

The total population exposed under each of these scenarios is around 500,000. This result feeds through to the assessment of impacts in Section 6.

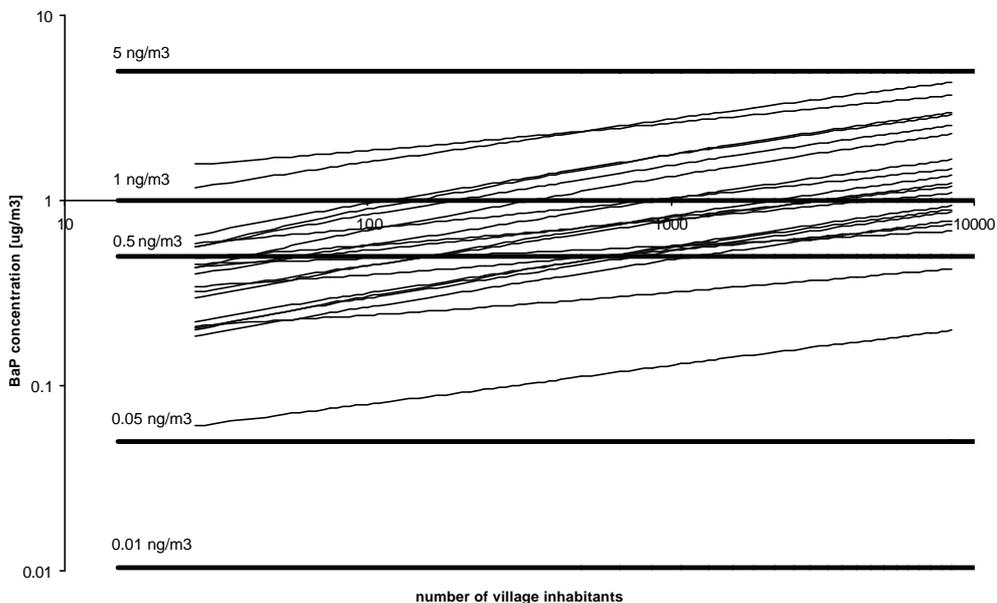
#### 4.3.4 Hot-spots - domestic sources - rural

In Figure 12 and Figure 13 the results are presented of the calculation of the contribution of residential wood combustion to ambient air concentrations. Each line represents one country. In Figure 12 only the effect of wood combustion is shown. It can be seen that the contribution of this emission source to local air quality depends (not surprisingly) on the village size and the amount of wood consumed. Figure 13 shows that residential combustion of wood and coal together with the rural background is forecast to provide between concentrations of between 0.5 and 5  $\text{ng}/\text{m}^3$  in many villages. These results are quantified in Table 14. The left side of the table shows the contribution from residential wood combustion to ambient air concentration in rural areas per country. The right side of the table shows the sum of background concentration plus the contribution from residential combustion of solids (wood, coal, and peat) in rural areas. It can be seen in the two last rows of the table, that 43% of all inhabitants of the rural part of the

study area and 14% of the inhabitants of the whole study area are estimated to be exposed to at least 1-5 ng/m<sup>3</sup> B[a]P due to residential combustion and background concentrations. Again, the differences between the countries are large.



**Figure 12. Average ambient B[a]P concentration within a village as a function of size, for all countries in the study area due to residential wood combustion. Each line represents one country.**



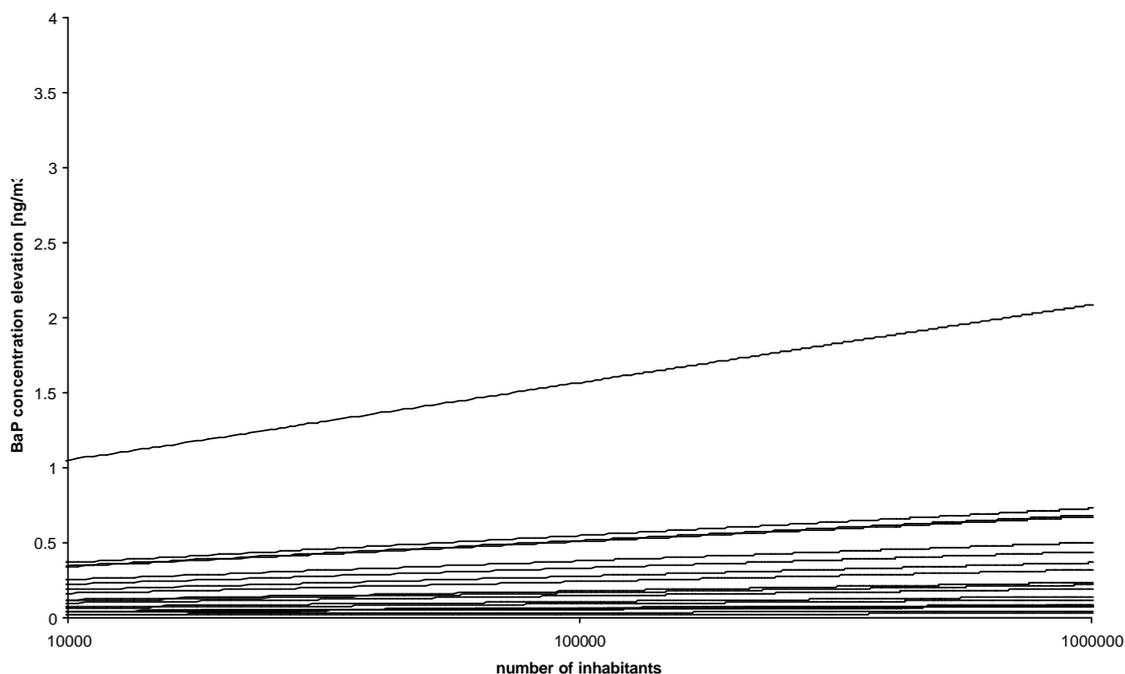
**Figure 13. Average ambient B[a]P concentration within a village as a function of size, for all countries in the study area due to residential wood and coal combustion plus the average rural background concentration. Each line represents one country.**

**Table 14. Number of inhabitants in rural areas exposed to the contribution of B[a]P emissions by either residential wood combustion (left part) or the residential combustion of solids and the background concentration (right part).**

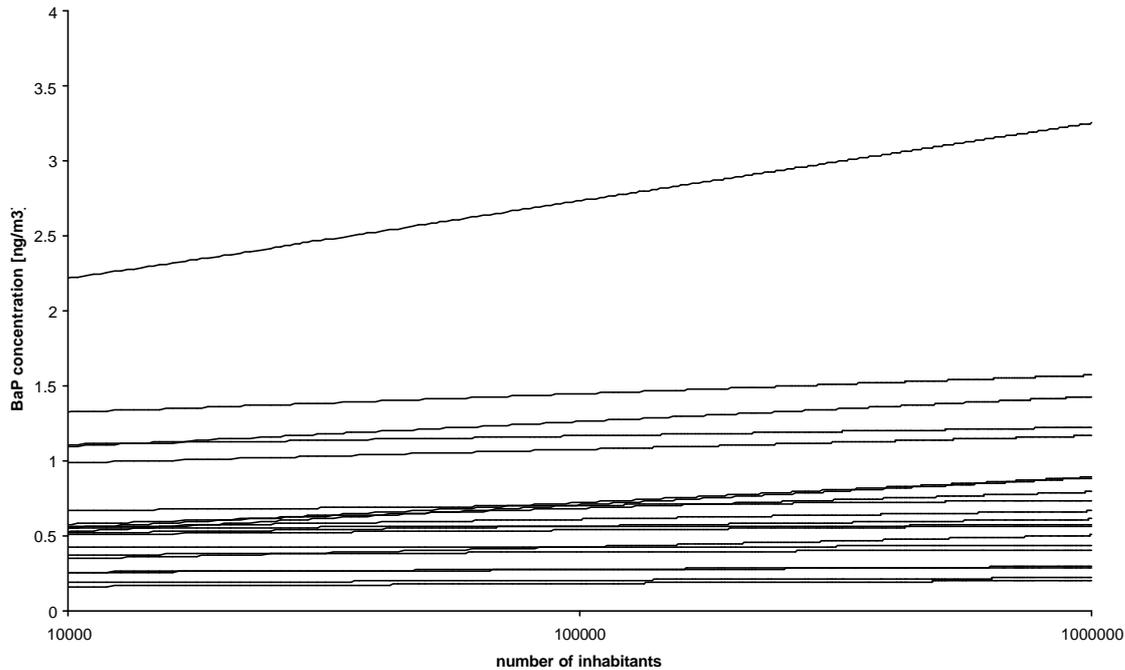
Country	Total rural population (cities <10,000 inhabitants)	Without background					With background				
		0.01-0.05	0.05-0.5	0.5-1	1-5	>5	0.01-0.05	0.05-0.5	0.5-1	1-5	>5
Austria	3417300	0	0	384439	3032861	0	377054	0	0	3040246	0
Belgium	3959200	438247	3520953	0	0	0	0	477520	3481680	0	0
Czech Republic	4155600	0	4155600	0	0	0	0	770830	3384770	0	0
Denmark	1027000	0	144401	882599	0	0	0	113316	77184	836500	0
Estonia	640000	0	71999	17988	622012	0	0	70616	6575	562809	0
Finland	1991600	0	220452	59577	1711571	0	219747	0	20461	1751393	0
France	16848600	0	1864985	504012	16344588	0	0	1859019	509977	14479603	0
Germany	12195400	0	1714734	10480666	0	0	0	1470889	3995737	6728774	0
Greece	5035500	0	1420790	3614710	0	0	0	1420790	3614710	0	0
Hungary	5273000	0	1487802	3785198	0	0	0	0	978099	4294901	0
Ireland	1956350	235956	1720394	0	0	0	0	362887	1593463	0	0
Italy	20185550	0	2838188	17347362	0	0	0	2434583	6613653	11137314	0
Luxemburg	91500	0	25817	65683	0	0	0	10128	54527	26844	0
Netherlands	2229600	246796	1982804	0	0	0	0	2229600	0	0	0
Poland	19180500	2123105	17057395	0	0	0	0	2157768	6439951	10582781	0
Portugal	7899750	0	2228952	5670798	0	0	0	1465340	6434410	0	0
Slovenia	1352550	0	541020	811530	0	0	0	95618	1082040	0	0
Spain	11884200	0	3353183	8531017	0	0	0	2204424	9679776	0	0
Sweden	1681800	0	189199	47270	1634530	0	185564	595	50310	1445331	0
United Kingdom	11428400	2119876	9308524	0	0	0	1260972	10167428	0	0	0
Total	131080850						2043337	27215733	46935283	54886496	0
Percentage of total rural population in the study area							1.5	20	35	43	
Percentage of total population in the study area							0.5	7	11	14	

#### 4.3.5 Hot-spots - domestic sources – urban

For urban situations, defined as places with more than 10,000 inhabitants, Figure 14 and Figure 15 show the results of the calculation of the contribution of residential solid fuel (wood, coal, peat) combustion to urban ambient air concentrations. Again, each line represents one country. It can be seen that the contribution of these residential emissions to air quality depends on the number of inhabitants and the amount of solid fuel consumed. The effect of residential combustion in the urban situation does not depend as much on the number of inhabitants as in the rural situation (Figure 12 and Figure 13). For most countries the contribution of this source in urban areas is estimated to be smaller than  $0.5 \text{ ng/m}^3$ . Figure 15 provides results including urban background levels. Further details are given in Table 15. The left side of the table shows the contribution from residential combustion processes to ambient air concentration in urban areas per country, without the urban background contribution. The right side of the table shows the sum of background concentration plus the contribution from residential solid fuel combustion in urban areas. 68.7% of all inhabitants of the urban part of the study area and 42.5% of the inhabitants of the whole study area are estimated to be exposed to at least  $0.5\text{--}5 \text{ ng/m}^3$  B[a]P due to combined residential combustion and background concentrations. For the  $1\text{--}5 \text{ ng/m}^3$  B[a]P concentrations, these percentages are 26.8% and 16.6% respectively. Again, the differences between the countries are large.



**Figure 14. Average ambient B[a]P concentration within a city as a function of number of inhabitants for all countries in the study area due to residential solid fuel combustion. Each line represents one country.**



**Figure 15. Average ambient B[a]P concentration within a city as a function of number of inhabitants for all countries in the study area due to residential solid fuel combustion plus the average rural background concentration. Each line represents one country.**

#### 4.3.6 Hot-spots - urban traffic

Maximum urban background levels of B[a]P in 2010 in the UK that were attributable to traffic emissions in 2010 were estimated to be slightly over  $0.2 \text{ ng.m}^{-3}$ . Similar maximum concentrations were quantified for three cities, Glasgow, London and Liverpool, suggesting that the overall results were not unduly influenced by any single city. Scaling by a factor 3 to estimate roadside levels provides a peak concentration across the UK as a whole of  $0.6 \text{ ng.m}^{-3}$ . These figures ignore background concentrations from other sources. Extrapolating to other cities in Europe (taking account of population and variation in the gasoline/diesel mix in the vehicle fleet) provides the results shown in Table 16, with highest estimated concentrations identified for France and Belgium. The Table identifies exceedences of all of the possible limit values investigated in this study except  $5.0 \text{ ng/m}^{-3}$  in all countries.

Recent data from Baden-Wurttemberg gives peak roadside values of  $1 \text{ ng.m}^{-3}$ , but details of precise location, type of road, traffic levels, etc., are lacking from the information supplied to the Working Group. The corresponding figure in the table ( $0.87 \text{ ng.m}^{-3}$ ) appears to be in the right range, though it does represent the result for the whole of Germany in 2010, results. However, substantial uncertainty will remain so long as there is not better data for validation purposes.

**Table 15. Number of inhabitants in urban areas exposed to the increase in B[a]P concentration due to emissions by residential combustion of solid fuels (left part) or by residential combustion of solids plus the influence of urban background concentration (right part).**

Country	Total urban population	Number of inhabitants exposed in concentration class (ng/m <sup>3</sup> )							
		Exposure due to domestic heating (coal, wood, peat) in urban areas, without urban background				Exposure due to domestic heating (coal, wood, peat) in urban areas including urban background			
		0.01-0.05	0.05-0.5	0.5-1	1-5	0.01-0.05	0.05-0.5	0.5-1	1-5
Austria	4446099		1348923	3097176				4446099	
Belgium	4235055	4235055					4235055		
Czech Republic	4756468		4756468				4756468		
Denmark	4242997		4242997			1161115	3081882		
Estonia	676000		77000	599000			676000		
Finland	2970035		2970035				2970035		
France	39163802		39163802					39163802	
Germany	52389093		52389093				52389093		
Greece	6162795	920809	5241986			6162795			
Hungary	5425106		5425106					5425106	
Ireland	382917		157605	225312			382917		
Italy	33572365		33572365				33572365		
Luxembourg	104066	25142	78924			104066			
Netherlands	6916877	6916877				6916877			
Poland	18718600		6231100	12487500				18718600	
Portugal	2275686	549969	1725717			2275686			
Slovenia	637450		223108	414342		95618	541832		
Spain	24098432		24098432			24098432			
Sweden	3744269		3744269				3744269		
United Kingdom	38165689	5083999	33081690			38165689			
Total					0	78980278	106349916	67753607	
Percentage of total urban population in the study area					0.0	31.2	41.9	26.8	
Percentage of total population in the study area					0.0	19.3	25.9	16.6	

The method used in this analysis implicitly assumes that a number of factors are constant across Europe:

- meteorological conditions,
- average levels of congestion, speed of traffic, etc.,
- relationships between urban background and roadside levels,
- patterns of population distribution relative to traffic,
- etc.

Given that these factors vary significantly, it is obvious that these assumptions introduce additional and significant uncertainties to the assessment. However, here we prefer to keep the analysis straightforward in the absence of additional data that would permit validation of results.

Table 18 provides a summary of the exposure to urban traffic emissions across a series of (nationally-specific) concentration bands. Exposure is calculated by multiplying population by twice the urban background concentration linked to traffic. The factor of 2 introduced here accounts for the movement of people relative to traffic, for example, during rush-hour. It is based on the results of the MACBETH study.

**Table 16. Estimated maximum concentrations ( $\text{ng}\cdot\text{m}^{-3}$ ) in background and roadside locations attributable to traffic emissions, in the EU21. Columns on the right hand side of the table include the contribution of all sources to background levels (taken from Table 10).**

	Without emissions from other sources		With emissions from other sources	
	Background	Roadside	Background	Roadside
Austria	0.16	0.49	1.26	1.59
Belgium	0.29	0.87	0.82	1.40
Czech Republic	0.15	0.46	0.57	0.88
Denmark	0.11	0.32	0.30	0.51
Estonia	0.15	0.46	0.36	0.67
Finland	0.22	0.67	0.58	1.03
France	0.36	1.08	1.35	2.07
Germany	0.14	0.42	0.59	0.87
Greece	0.11	0.33	0.26	0.48
Hungary	0.15	0.46	0.95	1.26
Ireland	0.16	0.48	0.32	0.64
Italy	0.12	0.35	0.72	0.95
Luxembourg	0.22	0.66	0.56	1.00
Netherlands	0.15	0.45	0.56	0.86
Poland	0.15	0.46	0.91	1.22
Portugal	0.17	0.52	0.39	0.74
Slovenia	0.15	0.46	0.15	0.46
Spain	0.19	0.58	0.31	0.70
Sweden	0.10	0.31	0.55	0.76
United Kingdom	0.19	0.56	0.41	0.78

**Table 17. Comparison of data in Table 16 against limit values. The rows headed ‘Traffic...’ relate only to concentrations attributable to vehicle emissions. The rows headed ‘All sources...’ relate to the concentrations attributable to traffic combined with urban background concentrations from other sources.**

Concentrations (ng.m <sup>-3</sup> )	Number of countries with maximum B[a]P level for the selected location/source combination...					
	<0.01	>0.01	>0.05	>0.5	>1.0	>5.0
<b>Location/source combination</b>						
Traffic - background	0	20	20	0	0	0
Traffic - roadside	0	20	20	7	1	0
All sources – background	0	20	20	12	2	0
All sources - roadside	0	20	20	11	7	0

**Table 18. Exposure of urban populations to traffic emissions (excluding background from other sources) in each concentration band for each country. Background concentration ranges for each band are shown for the UK. These values are multiplied by the figures given in Table 7, to obtain background concentration ranges in each other country.**

Country	Exposures in each band								
	UK range (ng.m <sup>-3</sup> )	0-0.003	0.003-0.014	0.014-0.029	0.029-0.057	0.057-0.086	0.086-0.114	0.114-0.143	0.143-0.171
Austria		0.002%	0.05%	0.32%	0.50%	0.42%	0.17%	0.04%	0.01%
Belgium		0.003%	0.08%	0.55%	0.85%	0.71%	0.28%	0.07%	0.02%
Czech Republic		0.002%	0.05%	0.32%	0.51%	0.42%	0.17%	0.04%	0.01%
Denmark		0.001%	0.03%	0.20%	0.31%	0.26%	0.10%	0.02%	0.01%
Estonia		0.000%	0.01%	0.05%	0.07%	0.06%	0.02%	0.01%	0.00%
Finland		0.002%	0.04%	0.29%	0.46%	0.38%	0.15%	0.04%	0.01%
France		0.036%	0.93%	6.30%	9.83%	8.19%	3.26%	0.78%	0.27%
Germany		0.019%	0.48%	3.28%	5.11%	4.26%	1.70%	0.40%	0.14%
Greece		0.002%	0.05%	0.31%	0.48%	0.40%	0.16%	0.04%	0.01%
Hungary		0.002%	0.05%	0.37%	0.58%	0.48%	0.19%	0.05%	0.02%
Ireland		0.000%	0.00%	0.03%	0.04%	0.04%	0.01%	0.00%	0.00%
Italy		0.010%	0.26%	1.75%	2.73%	2.27%	0.91%	0.22%	0.07%
Luxembourg		0.000%	0.00%	0.01%	0.02%	0.01%	0.01%	0.00%	0.00%
Netherlands		0.003%	0.07%	0.45%	0.71%	0.59%	0.23%	0.06%	0.02%
Poland		0.007%	0.19%	1.28%	1.99%	1.66%	0.66%	0.16%	0.05%
Portugal		0.001%	0.03%	0.18%	0.28%	0.23%	0.09%	0.02%	0.01%
Slovenia		0.000%	0.01%	0.07%	0.11%	0.09%	0.04%	0.01%	0.00%
Spain		0.012%	0.31%	2.08%	3.25%	2.70%	1.08%	0.26%	0.09%
Sweden		0.002%	0.04%	0.31%	0.48%	0.40%	0.16%	0.04%	0.01%
United Kingdom		0.018%	0.46%	3.16%	4.93%	4.10%	1.63%	0.39%	0.13%
<b>Totals</b>		<b>0.123%</b>	<b>3.1%</b>	<b>21%</b>	<b>33%</b>	<b>28%</b>	<b>11%</b>	<b>2.6%</b>	<b>0.9%</b>

## 4.4 UNCERTAINTIES

### 4.4.1 Modelling background concentrations

The overall uncertainties involved in modelling B[a]P concentrations by means of a dispersion model like EUTREND, are caused by the following aspects:

- 1) Uncertainties in the emission data, and
- 2) Uncertainties introduced by the model concept.

Different aspects of the emission data used in the modelling exercise are important for the final result. These are: emission strength, source height, heat content of the source, temporal behaviour of the emission and spatial distribution. The uncertainties in the emissions were discussed above.

Many issues contribute to the overall uncertainties introduced by the model concept: meteorological aspects of the model, substance-specific parameters used in the model, resolution related aspects.

The uncertainties introduced by the general meteorological aspects are assumed to be relatively small (in the context of this analysis), of the order of  $\pm 30\%$ ; Van Jaarsveld, 1995). Unknown uncertainty arises through using meteorological data for 1990 alone (the only year for which good quality data were available for this model).

The choice of deposition and degradation parameters can introduce large errors to the calculated concentration and deposition to a large extent. A sensitivity analysis on this subject carried out by Van Jaarsveld (1995) showed that this error is highest at large distances from the main emission areas and can reach  $\pm 200\%$  for total deposition on a yearly basis. Within (high) emission areas this kind of uncertainty has a magnitude of only  $\pm 30\%$  for the depositions of B[a]P. This is also valid for concentrations of B[a]P.

### 4.4.2 Hot-spot assessments

Uncertainties in characterisation of hot-spot concentrations are more problematic than those associated with background levels. There are several reasons for this:

1. Ideally, the analysis of hot-spots would be carried out using more locally specific data than was possible within the remit of this EU-wide analysis.
2. Concentrations in hot-spots will dictate compliance with any directive on PAHs.

The uncertainties for industrial facilities appear not to be significant in the context of the overall analysis, in terms of numbers of people affected. However, given that the highest concentrations are linked to cokeries and Soderberg-process aluminium plant, they are clearly very important in determining the potential for compliance with different limits. The major uncertainty here (i.e. with respect to potential compliance) lies in quantification of concentration at the nearest relevant location (house, school, hospital, etc.).

For hot-spots linked to domestic emissions from solid fuel consumption the main errors relate to:

1. Characterisation of fuel use in individual settlements.
2. Characterisation of average emissions for any settlement.
3. Assumptions on distance between houses burning solid fuels.

Given existing limitations in monitoring data specific to the domestic emission situation, the overall consequence of these uncertainties is that conditions in settlements across Europe may be considerably better or worse than estimated here. There is a clear need for more measurement data in this area in order to improve validation of results.

The same issues generally apply to hot-spot concentrations associated with traffic emissions. It is elsewhere accepted that the uncertainties of estimating traffic-related concentrations are high (AOPII-WGI, 2000).

No account has been taken of the consequences of the implementation of the first daughter Directive, which could be significant for both domestic and traffic hot-spots.

# 5 Options for Abatement

## 5.1 INTRODUCTION

A general characteristic of analyses of this type is that they have a tendency to overestimate the costs of abatement (even though *some* elements in the costing *may* be underestimated). These errors arise because the costing is rarely complete, with analysis focused on end-of-pipe abatement technologies, and with options such as fuel switching and energy efficiency left out. The sectors that are affected are also likely to consider their own approaches to abatement in a more wide ranging way once legislation is passed than when it is under negotiation. The problem is clearly worst where the required level of abatement is large, and where several sectors are each important, as is the case with PAHs.

Bearing this issue in mind, the costs presented here should be regarded as indicative, and like other parts of this assessment should be considered to be prone to significant uncertainty.

## 5.2 POTENTIAL ADDITIONAL PAH ABATEMENT MEASURES

Generic options for the abatement of emissions of PAHs from the industrial, domestic and traffic sources assessed in this report are shown in Table 19. With the partial exception of domestic solid fuel burning, however, these sources are already subject to extensive legislation to reduce emissions.

**Table 19. Potential additional PAH abatement measures**

<b>Source</b>	<b>Suggested measures</b>
Domestic solid fuel burning	Switch to alternative fuels (natural gas, LPG...) Optimisation of combustion Energy efficiency measures Prohibition
Traffic	Accelerated replacement of vehicles not complying to EURO IV Use of alternative fuels/technologies Improved vehicle efficiency Reduction in vehicle weight Prohibition in hot-spots Bus prioritisation Provision of subsidies to public transport Parking charges Road pricing Improving traffic flows
Industrial	Prohibition Relocation of plant Alternative technologies/methods

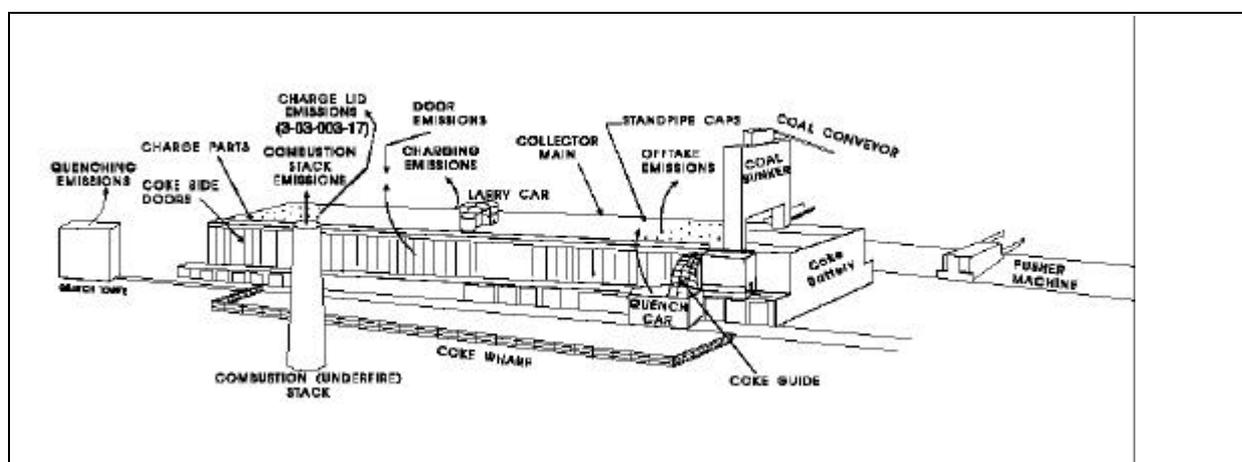
## 5.3 COKERIES

### 5.3.1 Additional details on the coking process and industry

To understand the potential for abatement from cokerries, and the constraints that exist, it is useful to understand something of the coking process. Prepared coal is 'coked', or heated in an oxygen-free atmosphere until all volatile components in the coal evaporate. Most coke is produced for use in the iron and steel industry. However, coke is also produced for 'smokeless fuels' for domestic and industrial use. Most coke plants are dedicated producers for, and hence co-located with, iron and steel production facilities. Not surprisingly therefore, demand for coke is highly dependent on the iron and steel industry.

The number of coke works in the EU15 has declined from 104 in 1981 to 42 in 2000, largely due to decreased demand for steel and consolidation in the ferrous metals industry. Table 20 presents a list of the coking plants in each country of the EU15. In addition there are 7 cokerries in the Katowice District of Poland, 4 in the Czech Republic and 1 in Hungary.

Figure 16 shows a coke oven 'battery' (typically comprised of 10 to 100 coke ovens operated together) used to produce metallurgical coke for use in iron and steel processes. The ovens must remain airtight under the cyclic stress of expansion and contraction. Each oven has 3 main parts: coking chambers, heating chambers, and regenerative chambers. Each chamber is lined with refractory (silica) brick and fitted with ports in the top for charging of the coal. After heating for 14 to 36 hours at 1,150-1,350°C, the coked mixture is pressed or pushed out of the coke chambers into special wagons and cooled.



**Figure 16 Schematic diagram of a metallurgical coke battery (source: US EPA 2000)**

**Table 20. Coke plants in the EU15 in 2000 (Fisher and Hein, 2000)**

<b>Country / total number of plants</b>	<b>Name of plant</b>
<b>Austria/ 1</b>	Linz
<b>Belgium/ 4</b>	N.V.Sidmar Seraing Marchienne Tertre
<b>Finland/ 1</b>	Raahe
<b>France / 6</b>	Pont-a-Mousson Seremange Fos Dunkerque Carling Drocourt
<b>Germany/ 5</b>	ATH (August Thyssen), Duisburg Dillingen Huckingen Prosper Salzgitter
<b>Italy/ 6</b>	Taranto Genova Cairo Montenotto Trieste Piombino Giuseppe di Cairo
<b>Netherlands/ 2</b>	Plant 1 Plant 2
<b>Portugal/ 1</b>	Seixal
<b>Spain/ 5</b>	Nalon Chemical Industrias Doy Santa Agueca Aviles Verina
<b>Sweden/ 2</b>	Lulea Oxelösund
<b>United Kingdom/ 9</b>	Appleby Dawes Lane Redcar South Bank Lianwem Morfa Grange Cwm Monckton

Releases of B[a]P occur principally as fugitive emissions during charging and pushing operations. Emissions also occur in the off-gases during production, which are controlled by collecting and ducting to treatment plant fitted with secondary abatement where by-products such as ammonia and tar are recovered. Fugitive emissions can also occur during production through leaks in oven door seals. These are difficult to prevent completely because of the stresses of expansion and contraction, together with the positive pressures (necessary to prevent the ingress of air) within the ovens during the coking process.

### **5.3.2 Use of alternative technologies/methods**

As shown in Sections 3 and 4, and in spite of the implementation of BAT, cokeries will still emit significant amounts of B[a]P in 2010. However, no additional reduction measures beyond BAT are known for coke plants. Given the problems of PAH concentrations around cokeries it may be appropriate for the question of BAT in this area to be looked at again by experts in the design and operation of such plant.

### **5.3.3 Relocation**

The Daughter Directives on air quality are limited in their impact to areas where people are likely to be exposed in non-occupational situations. For cases where industrial installations are located in densely populated areas, relocation of the entire facility might be an option, either to eliminate exceedences of standards in areas where people live, or to reduce exposure of the overall population. This operation would involve complete reconstruction and hence investment costs would be correspondingly high. This measure would only be economically feasible for cases where reconstruction is necessary for other reasons. Associated costs will be highly site-specific, and little, if any, of these costs would be attributable to legislation on PAHs. No estimate of cost is therefore made here.

### **5.3.4 Prohibition**

As a final resort, coking could simply be prohibited by law. As a consequence, domestic production would cease to exist, and demand would be satisfied by imports. This is already the case in some parts of Europe where import of cheap foreign coke replaces domestic production. Without additional measures to alter demand for the products of highly polluting processes, prohibition merely moves problems to other countries, where environmental regulation is less advanced. This is clearly against the spirit of the Framework Directive, even if it concerns countries outside the EU.

### **5.3.5 Summary**

Details of the various abatement options discussed for cokeries are summarised in Table 21.

**Table 21. Review of abatement options for cokeries.**

<b>Measure</b>	<b>Feasibility</b>	<b>Cost</b>	<b>Effectiveness</b>
Use of alternative technologies	None identified beyond BAT	not available	not available
Relocation	Could be used to reduce population exposure, but would not be undertaken solely as a PAH control measure	not relevant	not relevant
Prohibition	Would shift activity, and associated environmental problems, to a non-EU location.	unknown, but there are good reasons for not taking this approach	100% (locally)

## 5.4 ALUMINIUM WORKS

There are two types of aluminium-works, those using the Soderberg process and those using pre-baked anodes. Whilst the results shown in Section 4 demonstrated that PAH concentrations around Soderberg plants are extremely high, few, if any, of these plants will be operating in the EU15 by 2010. Indeed, out of 21 primary aluminium smelters operating in the EU15 there are only three Soderberg plants left. One is situated in Sweden (Sundsvall) and two in Spain (Aviles and La Coruna). The plant in Sweden has plans in place for an upgrade of the Soderberg process and is waiting for government approval. The two Spanish plants have not yet taken a decision on upgrading, but will also need to have something in place before 2010 as a result of the IPPC Directive and the OSPARCOM recommendations (Eirik Nordheim, communication on behalf of the European Aluminium Association). The situation in the Accession Countries is less clear, though assuming plant in these countries to be exposed to the same legislative framework and financial conditions as those in the existing EU; it would appear likely that they too, would close.

Emissions from anode-baking are roughly 2 orders of magnitude lower than those for the Soderberg process, per unit of aluminium produced. Given this much lower level of emission it is not anticipated that anode-baking plant lead to significant local hot-spot concentrations. The following discussion therefore concentrates on improvements around Soderberg plants, should any still be operational in the EU21 in 2010.

### 5.4.1 Alternative technologies

There are two options for modification of Soderberg plant:

1. Modification of the Soderberg process, leading to an order of magnitude reduction of the PAH emissions compared to the traditional Soderberg process. Estimated costs for an average plant are in the order of tens of millions of EURO.
2. Rebuilding plant to the pre-bake process. This will, for all practical purposes eliminate PAH emissions from the sector, but costs would be in the order of hundreds of millions of EURO. However, given the observed willingness of the industry to move away from the Soderberg process there are also (presumably) savings to be made from plant conversion, though these would be influenced by plant lifetime and numerous other factors.

### 5.4.2 Relocation

In the event that a plant is relocated away from a built-up area, the prime motivation is extremely unlikely to concern PAH limits. Trends in the industry clearly show that a relocated plant would not be built with Soderberg technology, and so this measure would be combined with newer technology.

### 5.4.3 Prohibition

As discussed already for cokerries, Soderberg plant could simply be prohibited by law. Without additional action to change demand for aluminium, or increased recycling, this would simply increase production in economies subject to lower environmental standards. However, demand for aluminium looks likely to increase, particularly in the transport sector, and the opportunity for enhanced recycling of aluminium appears very limited given the high rates already achieved.

### 5.4.4 Summary

Details of the various abatement options discussed for aluminium works are summarised in Table 22.

**Table 22. Review of abatement options for Soderberg-process aluminium works.**

Measure	Feasibility	Cost	Effectiveness
Modification of the Soderberg process	Soderberg process will either completely or largely have been displaced by 2010 through other legislation	>10 MEURO/plant	Around 90%
Rebuilding plant to the prebake process	As above	>100 MEURO/plant	>95%
Relocation	Could be used to reduce population exposure, but would not be undertaken solely as a PAH control measure	not relevant	not relevant
Prohibition	Would shift activity, and associated environmental problems, to a non-EU location, unless there are changes in demand (e.g. through increased recycling).	unknown, but there are good reasons for not taking this approach	100% (locally)

## 5.5 DOMESTIC SOLID FUEL BURNING

### 5.5.1 Issues and attitudes

Solid fuels are used in many houses instead of less polluting fuels for a variety of reasons, including:

- Cost of buying fuel
- Lack of access to alternative fuels
- Habit (some people have always used solid fuels and see no reason to change)
- Desire to use a renewable supply of energy
- Aesthetic appeal of a natural fire.

The strength of each of these issues will largely dictate the success of any policy to reduce emissions from this sector, and will determine to what extent mandatory measures will be needed to reach the limit values under investigation.

### 5.5.2 Emission optimisation for stoves

A reduction of 45% of the PAH emission rate can be achieved with a simple emission optimised stove compared to a standard stove. The optimisation includes the use of heat insulation in order for the stove to quickly achieve its operating temperature, improved air balance and a flame retaining plate. From experience of TNO-MEP with the Dutch national type approval programme for stoves, the incremental cost of optimisation is about 10-15%. With an average production cost of about 500 EURO this would be an additional cost of about 70 EURO per stove. No increases in operating costs or energy costs are expected. From the results in Section 3 it can be concluded that about 4-6 million stoves would have to be replaced by new stoves and have to be emission optimised as well in the rural part of the study area to reduce the contribution of residential solid fuel combustion to ambient air quality to  $\leq 1 \text{ ng/m}^3$ . The total costs for this would add up to 2.3 – 3.4 billion EURO.

Given the assumptions made in Appendix 2 on the distribution of wood use, the cost of applying this measure to stoves in urban locations would be 230 to 340 million EURO. Cost-effectiveness, in terms of reduced impact on the population (rather than simply in terms of reduced emission) will be higher for abatement from urban sources than from rural ones because of the locally higher population density.

A certain degree of emission optimization will result from national and European regulations on new stoves that might become more relevant in the future. In that case replacement might be considered cost-neutral. Further details are given in Appendix 2. For eight countries covering 28% of EU21 domestic wood consumption and which already have type approval of wood stoves, it is assumed that there will be a 60% penetration of new stoves by 2010. In other existing EU member states (covering 65% of EU21 wood consumption) optimised stoves are predicted to achieve a penetration of 20%, whilst in the accession countries (7% of EU21 wood consumption) it is forecast that there will be no optimised stoves.

In some cases progress has been made to further reduce emission from wood combustion units. These units comprise advanced automatically controlled systems that use for instance wood pellets as fuel. This fuel can be fed to the combustor at a precisely controllable rate. It is at this moment not yet clear whether this technology would be suitable for large-scale application in Europe. In any event, the time taken for this technology to penetrate the market to an appreciable degree is likely to be substantial.

### 5.5.3 Replacement of open fireplaces with optimised stoves.

Enforcement would be relatively straightforward for new building stock, at a cost of 500 EURO per stove. The extent to which the technology would be taken up in the existing housing stock by 2010 is debatable.

#### 5.5.4 Emission optimisation for open fireplaces

It may be thought that there is relatively little that can be done to improve efficiency of open fireplaces without moving to the use of stoves. However, this would assume that people who use fireplaces are following best practice. Variation in the experience of users will vary greatly, from people who have used them all their lives to those who grew up with gas and electric heating, and who thus perhaps have no past experience of burning solid fuels. Good practice in this area covers the following list, which, by its length, demonstrates that the use of a domestic fireplace is not as straightforward as may be thought:

1. Optimising fuel characteristics
  - a) moisture content
  - b) hardwoods vs. softwoods
  - c) use of different types of coal
2. Laying of the fire to ensure that it lights well and attains high temperatures quickly
3. Optimising combustion conditions
  - a) clearing ash to improve air flow
  - b) ensuring flues are not blocked
4. Keeping the fire going
  - a) controlling the size of the fire
  - b) stoking
  - c) adding fuel without reducing temperature significantly
  - d) knowing which waste materials can be added to the fire and which should not
5. Putting the fire out
  - a) extinguishing the fire vs. letting it die down naturally

A straightforward public information campaign would be required to disseminate good practice. This could presumably be undertaken at minimal cost by, for example, distributing leaflets with other information supplied by local authorities. The impact of this on emission levels is not known, because there have been so few measurements of emissions from the domestic sector. However, given current concern about emissions of both PAHs and PM<sub>10</sub> from domestic fires, it would clearly be a prudent measure.

#### 5.5.5 Switch to alternative fuels (e.g. natural gas, LPG)

Where applicable, this is the most effective measure in terms of emission reduction potential. In most cities in Europe coal-fired stoves are being gradually replaced by central heating systems using gas or oil, and in many cases have been replaced already, partly as a result of earlier clean air legislation. It may be expected that in the long term this replacement will be close to complete. However in the European energy scenarios there will still be a significant use of coal for residential heating purposes (see results) in 2010.

The cost of this replacement is not known and difficult to estimate since it is often a standard part of the complete renovation of a dwelling. It is also location specific – there are some areas where this type of measure is problematic for reasons which might be:

- **legal** (e.g. in parts of the UK and probably elsewhere in Europe where retired miners are supplied with free coal under the terms of their earlier employment),
- **political** (e.g. for areas where there is not yet the local legislation to mandate the use of smokeless fuels),

- **economic** (e.g. where it is technically feasible to provide a gas pipeline, but so expensive that no-one is willing to pay the cost), or
- **geographic** (e.g. there are physical barriers to the provision of alternative fuels).

The costs of fuel substitution are not known, and are dependent on the mechanism adopted.

### 5.5.6 Energy efficiency measures

Energy efficiency measures will have some effect on emissions, approximately proportional to the energy saving. Associated costs are, however, attributable to a number of other issues, ranging from global warming to energy security, according to the drivers regarded by individual Member State governments as most important. Without data on the potential for further implementation of energy efficiency in the types of homes likely to burn solid fuels it is not possible to estimate the future effectiveness of such a measure in reducing PAH. However, given that it is likely to be older homes that burn solid fuels, particularly in open fireplaces, this measure may be highly effective.

### 5.5.7 Prohibition

Prohibition of domestic solid fuel burning in some form has been a key feature of regulation in many urban areas going back at least as far as the 1950s, and possibly much earlier. However, it is of course necessary to ensure that prohibition is combined with other measures (such as provision of substitute fuels) to ensure that people are not denied energy services. It may be thought that wood combustion primarily for the cosmetic effect of a natural fire rather than for heating purposes could be targeted, though it is difficult to see how this would be enforced, particularly in rural areas. A further problem in gaining public support for measures to control domestic solid fuel use in rural areas may arise because air quality is assumed to be a problem of urban and industrial locations. Within this study it is not possible to estimate the costs of prohibition, which would depend on the following:

- price paid by individuals for solid fuels
- price of alternative fuels

However, where wood is the main source of fuel and obtained free of charge, individuals would probably need to pay several hundred Euro per year for substitutes. The study team has found no data to suggest how many people may be in this position. In cases where wood is sold, a ban would affect the local economy.

### 5.5.8 Summary

Details of the various abatement options discussed for abatement from domestic fires are summarised in Table 23. A number of measures are possible, that could lead to a reduction in PAH emissions from the domestic sector by perhaps 50% across Europe, or (obviously) more if prohibition is considered. There is a lack of data in this area, however, and so it is not possible to provide an appropriately detailed cost-curve for domestic abatement. To do so would require further measurement of emissions, and surveys to quantify the number of people burning solid fuels, what fuels they are using, and where they obtain them from. Consideration would also need to be given to the potential for enlarging the networks for distribution of other fuels.

**Table 23. Review of abatement options for domestic fires.**

Measure	Feasibility	Cost	Effectiveness
Optimisation of stoves	Technology available	2.5 to 3.7 billion Euro	45%
Replacement of open fireplaces with optimised stoves	Good for new houses, take-up may be limited in the older housing stock	500 Euro per stove	>50%
Information campaign to promote best practice for domestic combustion	No problems	negligible if combined with other activities	unknown, but potentially high
Switching to alternative fuels	Requires an economically attractive alternative fuel source to be available. Further potential may thus be very limited.	unknown – very location specific	unknown
Energy efficiency	There is potential for significant further savings through energy efficiency gains, though measures would be unlikely to be adopted specifically for PAH abatement.	0 MEuro (assuming costs attributable to other concerns)	proportional to energy savings
Prohibition	Domestic burning primarily for aesthetic benefits may be reasonable to target, though could be difficult. Requires availability of alternative fuels to ensure access to energy services is maintained.	Dependent on costs of fuels to users	up to 100%

## 5.6 TRAFFIC

The automotive industry has already agreed to implement far-reaching measures in order to limit vehicle exhaust emissions. Estimates of the additional investment costs of the EURO IV standards are estimated to be high (see, e.g., Cofala and Syri, 1998). Any new agreement on vehicle emission standards would presumably enter into force some time after EURO IV which will apply from the end of 2005. This would leave very little time, if any, for penetration of vehicles equipped to the higher standard to meet PAH standards in 2010. Given the potential extent of exceedences of the limit values investigated here, it may well be that further technical advances would be necessary. However, the time of implementation of any Directive on PAHs may then have to be pushed back beyond 2010 to allow time for the penetration of new technologies.

The main problems with emissions of PAHs from vehicles relate to the use of diesel fuel. Based on the emission factors used in this report, vehicles using gasoline are estimated to generate significantly less PAH.

Options for abatement of vehicle emissions can be split into three broad categories:

- **Technical options** which can reduce pollutant emissions by improving vehicle technology and fuel quality, including alternative fuels and inspection & maintenance programmes;
- **Management options** which can reduce pollutant emissions by promoting the most economically and environmentally efficient use of road space;
- **Policy options** which set the context for the take-up of other options, including economic and regulatory measures.

Table 24 lists options available for abatement of emissions from vehicles. The list is reasonably comprehensive, though additional measures could no doubt be added. To some extent these options will be introduced under existing legislation, for example, to address exceedences of limits for NO<sub>2</sub> and PM<sub>10</sub>. Unfortunately, it is not yet clear precisely what will be introduced to meet the limits set down for other pollutants, though some studies are starting to emerge that will assist, such as the work by W. S. Atkins (1998) on the UK's National Air Quality Strategy and the second Auto-Oil programme (European Commission and others, 2000).

The W.S. Atkins report highlights the need to consider packages of measures, rather than individual options, particularly when the reduction in emissions required is large, and the measures considered are not mandatory technical refinements. One of the main reasons for this is that different measures within the cost curve often interact with one another. Even bringing together measures into groups, however, does not go far in solving the problems of assessment. Further issues relate to the scale (e.g. the area over which a measure is, or can be, applied) and the intensity (such as level of subsidy provided) of a measure. Both will influence the effectiveness of control options, as will numerous other local factors, such as existing levels of congestion, availability of alternative transport options and so on. Overall, the Atkins report regards the problems associated with generalisation in identification of transport options and associated costs as 'extreme'.

## 5.6.1 Technical Measures

### 5.6.1.1 Emission standards

Working Group 2 of AOP-II reported that to meet Euro-III and Euro-IV standards, diesel vehicles would be improved by introducing various combinations of 4-valve engines, increased injection rates, electronically controlled injectors, low swirl, catalytic converters (reducing particles and VOCs) and exhaust gas recirculation with optimised supercharging. They further reported that "*Which technology combination will prevail in the market is again uncertain. Some experts expect that all diesel vehicles will be equipped with oxidation catalysts whilst particle traps will be required for the upper range diesel cars*". Without further information it is not possible to define precisely the technologies that will be in place as a result of the current legislation. However, given the problems that are anticipated for meeting particle limits, it is to be expected that particle control will remain central to future technological improvements in vehicles, alongside improving fuel efficiency. Given the link between PAHs and particles, and PAHs and fuel consumption, action in both areas is anticipated likely to reduce PAH emissions. The report of WG2 of AOP-II also discusses the provision in Directive 1999/96/EC for 'Enhanced Environmentally Friendly Vehicles' (EEV). These may be promoted through tax incentives where there is an air quality need. A review by DG-ENTR provided the following results on effectiveness and cost (Table 25).

**Table 24. Options for abatement of vehicle emissions.**

	<b>Category</b>	<b>Option</b>
<b>Technical Options</b>	Emission standards	<ul style="list-style-type: none"> <li>• Particle traps, exhaust gas treatment such as catalytic converters</li> </ul>
	Reducing fuel use	<ul style="list-style-type: none"> <li>• Improved efficiency of engines</li> <li>• Reduction in vehicle weight</li> </ul>
	Improved durability of vehicle systems	<ul style="list-style-type: none"> <li>• Increased severity and frequency of compulsory inspection and maintenance checks</li> <li>• On-board diagnostics</li> <li>• Remote roadside sensing</li> <li>• Extension of manufacturer's liability</li> <li>• Conformity of Vehicles in Circulation (CVC) programmes</li> </ul>
	Alternative fuel types / composition	<ul style="list-style-type: none"> <li>• Alternative fuels (CNG, LPG, biofuels)</li> </ul>
	Alternative drive-trains	<ul style="list-style-type: none"> <li>• Electric vehicles</li> <li>• Hybrid vehicles</li> </ul>
<b>Management Options</b>	Demand management	<ul style="list-style-type: none"> <li>• Traffic restrictions</li> <li>• Traffic speed regulations</li> </ul>
	Restrictions on vehicle access	<ul style="list-style-type: none"> <li>• Establishment of low emission zones open only to vehicles meeting specified standards</li> </ul>
	Congestion reduction	<ul style="list-style-type: none"> <li>• Traffic light controls, e.g. UTC systems</li> <li>• Construction of new roads</li> </ul>
	Encouraging modal shift	<ul style="list-style-type: none"> <li>• Better integration of transport systems</li> <li>• Reduced public transport fares</li> <li>• Park-and-Ride</li> <li>• New and improved public transport systems</li> <li>• Public transport priority</li> <li>• Freight subsidies</li> </ul>
<b>Policy Options</b>	Economic	<ul style="list-style-type: none"> <li>• Road Pricing</li> <li>• Higher fuel taxation</li> <li>• Tax differentials to promote cleaner fuels/emission taxes</li> <li>• Differentiated vehicle purchase taxes to promote lower emitting vehicles</li> <li>• Subsidies for scrappage of old vehicles</li> </ul>

**Table 25. EEV options applicable to all new heavy duty vehicles and buses (AOP-II, WG2). Costs expressed as % increase on catalyst costs (assumed Euro 1,500 to 3000 per vehicle).**

Vehicle types	% cost increase in cost (an increase on catalyst cost)	% reduction in emissions on top of standards for 2005 and in advance of 2008			
		CO	HC	NO <sub>x</sub>	PM
All fitted with oxidation catalysts	0-10%	60-95	60-95	0	20-25
All fitted with particulate traps	200%	-	-	-	50-80 <sup>(1)</sup>
All fitted with deNO <sub>x</sub> catalysts, deNO <sub>x</sub> traps or selective catalytic reduction (SCR)	Euro 1500 SCR <sup>(2)</sup>  15-300% over oxidation catalysts for deNO <sub>x</sub> catalysts or traps	0-60	0-60	50-70	0-50
All fitted with more than one or all above devices	20-400%	50-95	50-95	50-70	50-80
(MeOH) reformer or direct MeOH fuel cell vehicles	not available	95	95	99	99

<sup>(1)</sup> All reduction in particulates is in mass. For particulate trap systems incorporating catalysts the highest reductions assume the use of < 10 ppm sulphur fuel since on 50 ppm fuel the sulphate part of the total particulate mass would be equivalent to the 2005 limit values.

<sup>(2)</sup> SCR -- Selective Catalytic Reduction

An EEV Working Group has explored the same issues for passenger cars and light commercial vehicles. Results for diesel vehicles are shown in Table 26.

**Table 26. After treatment options applicable to all new diesel passenger cars and light commercial vehicles. Costs expressed as % increase on catalyst costs (assumed Euro 500 to 800 per vehicle). % abatement additional to achievements under Euro-IV.**

Vehicle types	% increase in cost (an increase on catalyst cost)	% reduction in emissions on top of standards for 2005			
		CO	HC	NO <sub>x</sub>	PM
All fitted with oxidation catalysts	0-30%	20-50	20-50	0-50	0-50
All fitted with particulate traps	25-100%	25	25	20-50	80-95 <sup>(1)</sup>
All fitted with deNO <sub>x</sub> catalysts, deNO <sub>x</sub> traps or selective catalytic reduction (SCR)	Euro 400 SCR <sup>(2)</sup> 20-100% for deNO <sub>x</sub> catalysts or traps	0-50	0-50	20-80	5-50
All fitted with more than one or all above devices	35-200%	10-50	10-50	50-80	80-95
(MeOH) reformer or direct MeOH fuel cell vehicles	not available	90-95	90-95	99	99

<sup>(1)</sup> All reduction in particulates is in mass. For particulate trap systems incorporating catalysts the highest reductions assume the use of < 10 ppm sulphur fuel since on 50 ppm fuel the sulphate part of the total particulate mass would be equivalent to the 2005 limit values.

<sup>(2)</sup> SCR -- Selective Catalytic Reduction

Vehicle manufacturers apparently regard the % abatement figures given in these two tables as overestimates, and regard the cost estimates at the present time as speculative. Overall, it seems that there will be scope for further technological improvements that could have a significant impact on emissions. However, at the present time data are too uncertain to make estimates of cost-effectiveness with a reasonable level of confidence.

### 5.6.1.2 Reducing fuel use

There is already pressure on the auto industry to improve the efficiency of fuel use by vehicles. There are three main ways to do this:

- Improving engine efficiency.
- Reducing vehicle weight.
- Offering features that increase fuel use (e.g. air conditioning, power steering) as options, rather than as part of a standard package, or increasing visibility of associated fuel penalties.

Measures introduced to limit fuel use could have a substantial effect on reducing PAH emissions. However, at the present time it is not clear how extensive these savings will be. Also, costs are not reasonably attributable to PAH control, but to concern about climate change. Improvements in this area may come about from a diverse range of sources, for example research into new adhesives that enable the use of lighter materials without compromising safety.

#### **5.6.1.3 Improved durability of emission control systems**

Significant reductions in vehicle emissions can be achieved by reducing the rate of deterioration in emissions control systems. The Auto Oil Programme identified the following mechanisms for achieving this:

- Increased severity and frequency of compulsory inspection and maintenance checks;
- Extension of the manufacturer's liability for the performance of the emission control systems from 80,000 to 160,000 km;
- Electronic sensors installed on the vehicle to monitor the performance of the systems (on board diagnostics);
- Re-call procedures (e.g. Conformity of Vehicles in Circulation (CVC) programmes) whereby models can be re-fitted if their emissions performance deteriorates beyond an acceptable level;
- Improved mechanisms for the remote roadside detection of vehicles emitting above acceptable levels.

These measures are already being explored in the context of other pollutants, so the incremental costs attributable to PAH control may be zero.

#### **5.6.1.4 Alternative fuels and drive chains**

The use of alternative fuels to diesel such as compressed natural gas (CNG) has the potential to bring about significant reductions in emissions (ETSU, 1996) and is being widely implemented in some places, including Beijing. It has been identified as one of the cheapest measures for reducing emissions of other pollutants, including PM<sub>10</sub>, NO<sub>x</sub>, CO and benzene. However, once again, on this basis it would appear likely to be introduced in response to the limits on those pollutants rather than to new limits on PAHs. Again, therefore, the costs of implementing this measure that are strictly attributable to a new Directive on PAHs may be zero; though they may achieve important reductions in PAH emission.

Examples of new drive chains include electric vehicles, and fuel cells, which a number of major manufacturers claim to have developed to a near-market state. Without some form of encouragement, the penetration of these new technologies seems likely to be very limited by 2010. The application of alternative fuels and drive chains may be particularly suited to fleet operators who can dispense fuel from a central depot to vehicles operating in a limited geographical area. Thus the use of these fuels in buses, taxis or light delivery vehicles could play an important role. Again, however, it seems very unlikely that the penetration of these technologies is likely to be influenced to a significant degree by legislation on PAHs. Other factors such as driving performance, constraints on refuelling, cost, and, possibly, the potential for reducing greenhouse gas emissions will dominate. Some cost data are presented above in Table 25 and Table 26.

#### **5.6.1.5 Summary for technical options**

A variety of options have been identified that are expected to lead to a reduction in emissions of PAHs. However, it seems likely that, for the most part, these options will be introduced in

response to other pressures, the need to reduce greenhouse gas emissions and the need to meet the existing daughter Directives. Therefore, although these measures may significantly improve the position with respect to exceedence of possible PAH standards identified in this report, the incremental costs attributable to PAH control would in most cases be zero. In any event, the results of exercises to assess the cost-effectiveness of technical options for abatement suggest a high uncertainty in available data.

## 5.6.2 Management options

Traffic management options for reducing vehicle emission can be grouped as follows:

- Traffic restraint;
- Congestion reduction;
- Encouraging modal shift.

Both the effects of these measures on emissions and their costs are very difficult to predict because they act on emissions indirectly through price or regulatory mechanisms that in turn affect consumer behaviour. Indicative implementation costs are listed at the end of this section, based on W.S. Atkins (1998).

### 5.6.2.1 Traffic Restraint

Schemes that reduce the total volume of traffic in an area will tend to be associated with reduced emissions. However, where such changes cause a redistribution of traffic then environmental disbenefits can also be expected.

Road pricing is increasingly being considered by policy makers for reducing congestion. It has the dual effect of reducing the total volume of traffic and encouraging passengers to move from private to public transport. The EUCARS simulation model used in AOP I predicted a reduction in emissions in *urban* areas by 4 to 6% for an increase in road travel costs of 10%, rising to 18 to 25% for an increase in road travel costs of 50%. There would be slightly lower *total* emissions reductions ranging from 1.6 to 2.6% for a 10% increase rising to 18 to 23% for a 50% increase.

AOP I concluded that road pricing was potentially the most cost-effective local management measure available for emissions reductions. There are, however, a number of problems with this option. It is highly capital intensive, requiring significant investment outlays. In addition, the technology for charging still remains to be tested for large schemes. There is also a problem of equity, in that road user charges would hit the poorest motorist hardest.

Traffic restriction policies may be reinforced by parking policies to limit the supply or increase the price of parking in central areas. Particular types of traffic restriction schemes are:

- Licence plate bans, where vehicles are allowed to enter an area on alternate days only, the days being determined on the basis of the license plate;
- Permit entry systems, where access to areas is restricted to defined groups such as residents or commercial vehicles only;
- Total area bans, where all traffic is restricted from a usually small central area.
- Road closures during exceedence episodes

AOP I predicted that traffic restraint may have some potential as a method of reducing emissions in areas of high local pollution.

The direct costs of traffic restrictions are quite low. Putting in place traffic restrictions typically involves some capital investment for signs and minor highway works to impede traffic. There are, however, additional indirect costs for motorists related to the additional time spent making journeys.

### 5.6.2.2 Congestion Reduction

Schemes that reduce congestion tend to reduce emissions due to the reduction of stop/start operations. There are many different types of urban traffic control systems (UTC). Table 27 summarises the different types and shows estimates of changes in fuel consumption, where these have been estimated.

**Table 27. Impacts of urban traffic control (UTC) systems.**

<b>System</b>	<b>Change in Fuel Consumption</b>
TRANSYT fixed time signal co-ordination set to minimise delay compared to uncoordinated signals	up to -15%
Planned green wave allowing smooth driving at 40 km/h	-16% (Caen)
SCOOT signal co-ordination compared to fixed time	-9% (Glasgow / Coventry) -5% (Worcester)

Source: TRL (1995).

A number of large towns and cities in Europe already have some form of UTC system, so there is only limited potential for achieving further emissions reductions. However, these systems are being extended to integrate traffic and environmental monitoring and management. There are also potential environmental benefits indicated by using UTC systems to give priority to public transport vehicles and to achieve area wide traffic restraint.

### 5.6.2.3 Encouraging Modal Shift

Schemes that encourage a change from private car use to public transport (modal shift) should produce net emissions benefits. Measures include:

- Park-and-ride;
- New transport systems;
- Public transport priority;
- Reduced public transport fares;
- Freight subsidies.

Park-and-Ride schemes have the potential to reduce car use within the inner city area but the extent to which this is achieved has not been well documented. They may also stimulate additional trips and trips with further distances. For these reasons, together with the fairly high investment costs required, park-and-ride may not be a cost-effective option for reducing emissions.

Similarly, new transport systems require a very high capital cost for questionable emissions benefits. It is only with the use of complementary car restraint policies that the benefits of new, attractive public transport systems will be realised.

Bus priority measures can significantly increase service reliability, reduce journey times and bus emissions, but may increase emissions from other vehicles. Thus the overall effect on emissions can sometimes be small.

The evidence from reduced public transport fares is that they result in significantly increased patronage, but only a small decrease in car use. The reason for this is that the lower prices release suppressed demand among the poorer members of society for whom the previous price was too high, but do not offer enough benefit to entice car users away from their cars. For example, with a 40% reduction in public transport fares, AOP I predicted a 49% increase in public transport use but only a 5.6% reduction in total car mileage. As this shows, the potential for emissions reductions of reduced fares is limited, unless undertaken in conjunction with other measures to encourage modal switch (such as road pricing).

Freight subsidies for alternative modes of road freight have the potential to significantly reduce emissions and congestion in both urban and non-urban areas. The available evidence suggests, however, that this is relatively expensive and as such freight subsidies may not be the most cost-effective method of reducing pollutant emissions. The Commission is investigating the potential for this type of modal shift through the PACT (Pilot Action on Combined Transport) programme of DG TREN.

It should be stressed that the effectiveness of all modal shift options can be enhanced significantly when in combination with other measures such as road pricing or traffic restraint.

#### **5.6.2.4 Summary for management options**

Illustrative cost data on the above measures, and some additional ones, are provided by the W.S. Atkins (1998) report on the costs of meeting the UK's National Air Quality Strategy (limit values for which are broadly consistent with those under the existing daughter Directives). They are summarised in Table 28. It is stressed that the data are illustrative, and will vary very significantly from city to city, reflecting local conditions.

Further information is available through Part IV (Simulation and integrated assessment of policy measures), Annex 4 to the report on the cost-effectiveness study for AOP2 (European Commission and others, 2000), though that report is, at the present time, still draft. Again, the results are very specific to the limited number of cities for which assessments were made, though there appears to be broad consistency in the costs data. AOP2 raises other issues, notably that although the overall impact of non-technical measures on emissions tends to be small (<10%), the effect on a more local level can be much higher.

**Table 28. Indicative implementation costs for local transport measures (W.S. Atkins, 1998).**

<b>Measure</b>	<b>Costs</b>
New and improved heavy rail systems	Euro 7.5 to 300 million/km
New light rail systems	Euro 7.5 to 15 million/km
Guided bus systems	Euro 300,000 to Euro 2.1 million/km
Bus priority systems	Euro 150,000 to Euro 1.3 million/km
Park and ride (bus)	Euro 1 to 1.5 million/site
Park and ride (train, including major new station)	Euro 15 to 22 million/site
Public transport interchange (bus)	Euro 400,000 to Euro 3 million/facility
Public transport interchange (surface rail)	Euro 3 to 20 million/station
Public transport interchange (underground rail)	Euro 30 million/station
Public transport information	Euro 750,000 to Euro 7.5 million/system
Public transport fare reductions (noting that costs in major conurbations will be higher)	Capital costs: Euro 450,000 to Euro 3 million/city Operating costs: Euro 7.5 to 34 million/year/city
Cycle routes	Euro 7,500 to 150,000/km
Cycle parking	Euro 90 to 900/stand
Parking controls	Euro 400,000 to Euro 1.5 million/scheme
Traffic calming	Euro 40,000 to 75,000/km Euro 9 to 14 million/city
Pedestrianisation	Euro 1.5 to 3 million/small city centre Euro 15 to 40 million/major centre
Priority bus routes	Euro 700,000/km
Road pricing	Euro 15 million/city Euro 150/vehicle/year Capital costs for London: Euro 150 to 700 million Operating costs for London Euro 100 to 290 million/year
Traffic management	Euro 45,000 to 750,000/junction Euro 3 to 7.5 million/typical town centre Euro 150,000 to 750,000 for road network signing
Urban traffic control	Euro 60,000 to 75,000/junction, including signals Euro 7,500 to 15,000/junction, upgrading only Euro 1.5 to 3 million/city
Variable message signs	Euro 27,000 to 75,000/sign
High occupancy vehicle lanes	Euro 150,000/km
Awareness measures	Euro 300,000 to Euro 1.5 million/city

### 5.6.3 Policy options

#### 5.6.3.1 Economic and fiscal measures

There are several ways in which fiscal policy could be used to reduce road transport emissions. These range from increased taxation to reduce the total level of transport activity, to differential taxation to encourage the use of the less polluting vehicle types. Increased fuel taxes have the potential to reduce emissions by encouraging consumers to travel less by car, drive more frugally, and purchase more fuel-efficient vehicles. In addition, fuel taxes provide an incentive to vehicle manufacturers to produce more fuel-efficient vehicles. Higher fuel taxes are easy and cheap to implement, and directly target fuel use and therefore emissions. The first Auto Oil study suggested that increases of fuel tax of 70% could reduce emissions by 10 to 15%. However, higher fuel taxes have limitations as they result in significant welfare costs for motorists. In addition, this form of taxation is regressive and has no way of targeting those areas where the greatest reductions in emissions are required.

Differential fuel taxes can be used to alter the tax differential between petrol and diesel. The use of the price mechanism is a very effective method of achieving the desired outcome, as the rapid introduction of unleaded fuels in the EU since the late 1980's has demonstrated. In addition, it is also very simple to administer. The main problem with this approach, however, is that although certain target pollutants may be reduced, there will be knock on effects on the other emissions, which may lead to increases.

Structured vehicle purchase taxes can achieve improvements in the average fuel consumption of the vehicle fleet, rather than emissions reductions per se. The EUCARS simulation of the first Auto Oil study suggests that fuel consumption per km is reduced by up to 3.4%. However, the relative impact on emissions (other than CO<sub>2</sub>) is very small at less than 1% even for the highest rate of tax (Euro 700 for every litre per 100 km in excess of the target level of 3 litres per 100 km).

Scrapage subsidies are grants paid by the government to the consumer on release of old vehicles for scrapage. Effects on emissions seem very low – the AOP2 modelling for Athens suggesting that a scrapage subsidy of Euro 1000/vehicle would lead to a reduction in emissions of NO<sub>x</sub>, PM<sub>10</sub>, CO, CO<sub>2</sub> and VOCs of much less than 1%. It should be noted, however, that at the start of 2010, the EURO IV standards will have been in place for just 4 years, so significant improvements in emissions of PAH will still be gained (with roughly a 50% fall in emissions of PAHs from traffic to be expected between 2010 and 2018).

#### 5.6.3.2 Summary on policy options

Although fiscal policies can be very effective, they frequently lack the focus necessary for them to be cost-effective policy tools for meeting air quality targets when compared to well targeted regulation. In addition, their effect is gradual, making them unsuited for tackling the problem of peak air pollution concentrations.

### 5.6.4 Prohibition in concentration hot-spots

In some countries the idea of selective prohibition of diesel-fuelled and other high-emission vehicles in urban environments has been considered as an option to improve local urban air quality, creating what are known as 'low emission zones'. A problem in the context of PAH

control, common to all traffic related measures, concerns the high contribution to urban concentrations here estimated to be attributable to non-urban emissions.

# 6 Quantification of the Impacts of PAH Exposure in Europe

## 6.1 REVIEW OF EFFECTS AND RISK FACTORS

### 6.1.1 Effects

The brief review of health effects given here is largely based on that carried out by the PAH Working Group, and other reviews (e.g. Bostrom, 1999, for the Swedish Environmental Protection Agency; DETR, 1999).

No individual PAH compounds are classified as Group 1 carcinogens (proven to cause cancer in people) by IARC (see Table 1 in Appendix 3), though a number are in Groups 2A and 2B (probable and possible human carcinogens). Others tend to be unclassifiable through a lack of specific data. However, a number of pollutant complexes rich in PAHs, such as coal tar, soot, fumes from aluminium works and cokeries and tobacco smoke are Group 1 carcinogens. These data, together with the results of experiments on animals, establish a robust link between cancers and PAHs.

The main effects linked to human exposure to PAHs are:

- Lung cancer through inhalation, and thus the effect most closely related to air pollution. There are 204,000 deaths from lung cancer annually in the EU21 (WHO Databank). Mortality rates within 5 years of diagnosis are high, at 90%. The severity of lung cancer is partly a result of the fact that it is hard to diagnose in its early stages.
- Bladder cancer, which appears to result from a combination of ingestion and inhalation, and is hence contributed to by air pollution. Clear dose response relationships are not available, through there being only limited records in the literature, though excess cases of bladder cancer have been detected in some occupational health studies. The workers concerned appear likely to have been exposed to agents known to cause bladder cancer, such as aromatic amines, as well as PAHs. There are 33,500 deaths each year in the EU21 from bladder cancers.

A variety of other cancers (skin, pancreatic, kidney) have been linked to PAH exposure, though evidence for them is relatively weak. WHO's review found in favour of a relationship with lung cancer, but not bladder cancer, regarding the information available to be insufficient. A recent paper by Romundstad *et al* (2000) was not included in the previous reviews for WHO, the Swedish EPA or the British government. In part the paper illustrates the difficulties inherent in applying the results of epidemiological studies to the wider population. The study considered lung, bladder, kidney and pancreatic cancers among Norwegian aluminium smelter workers. In the core analysis of the study, based on comparison against national incidence rates, an overall excess of bladder cancers was detected, whilst there was also some elevation of kidney cancers, and of pancreatic cancers (though in this last instance there was no clear exposure-response relationship). An association with lung cancers was not identified. However, basing

the analysis on local county rates the situation was reversed, with an association against lung cancer but not bladder cancer. The Working Group has concluded that the study strengthens the case for bladder cancers, and weakens it for lung cancer. However, as a single study it does not provide sufficient evidence to move away from lung cancer to bladder cancer as the prime effect of PAH exposure.

It does not appear that there are any quantifiable effects of PAHs on ecosystems (though this is not to say, of course, that there are no such effects).

Contribution of the heavier PAHs to particulate mass and volume will affect the soiling of buildings, though this will be negligible given that PAHs are available in only trace concentrations.

### 6.1.2 Risk factors

Quantifiable risk data are available from studies on animal and human subjects, though for the purposes of this study it is only the latter that are considered. Overall, the animal data are of a similar magnitude to the human risk data, though the observed range is wider, particularly at the lower end. The following formed the core data used by the Working Group in consideration of probable risk factors (Table 29).

**Table 29. Summary of lifetime Unit Risk Estimates for PAH per  $\text{ng.m}^{-3}$  of B[a]P (taken from the summary given in the PAH Working Group). The WHO and RIVM “most appropriate” estimates are based on reviews, which included some of the other papers listed.**

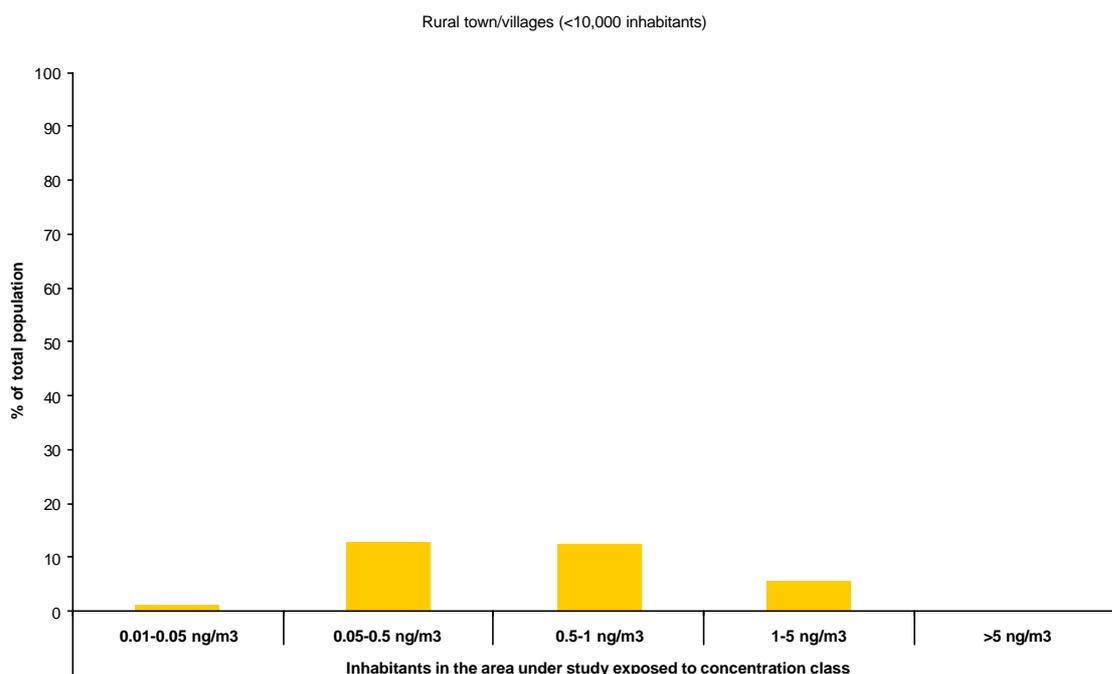
<b>Basis for calculation</b>	<b>Unit Risk</b>	<b>Reference</b>
US coke oven workers	$87 \times 10^{-6}$	WHO, 1987; 1988
US coke oven workers	$23 \times 10^{-6}$	Muller <i>et al</i> , 1997
US coke oven workers	$50 \times 10^{-6}$	Pott, 1985
UK gas workers	$430 \times 10^{-6}$	Pike, 1983
Smoky coal indoors in China	$67 \times 10^{-6}$	RIVM, 1989
"Most appropriate" estimate	$100 \times 10^{-6}$	RIVM, 1989
Aluminium smelters	$90 \times 10^{-6}$	Armstrong <i>et al</i> , 1994, converted from workplace exposure to continuous life-time exposure

The risk factor recommended by the Working Group, and used here for the best estimate, is that derived from the WHO (1987, 1988) studies on coke oven workers. The other studies shown suggest a range for the risk factor roughly a factor 4 higher and lower, and this is carried through to the sensitivity analysis. As noted above, the paper by Romundstad *et al* (2000) provides a different view, though not one that the Working Group found so compelling as to alter their views on the relevant risk factor.

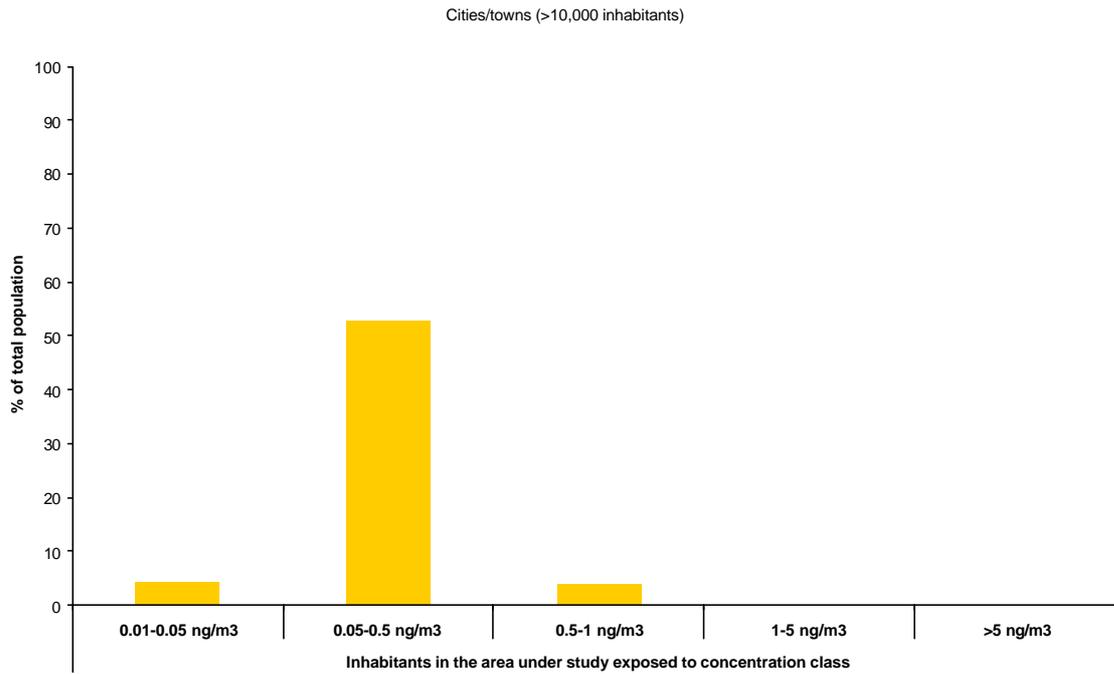
As note elsewhere, the risk factors shown are expressed against B[a]P concentrations, though B[a]P is simply used as an indicator of PAH concentration.

## 6.2 EXPOSURE DATA

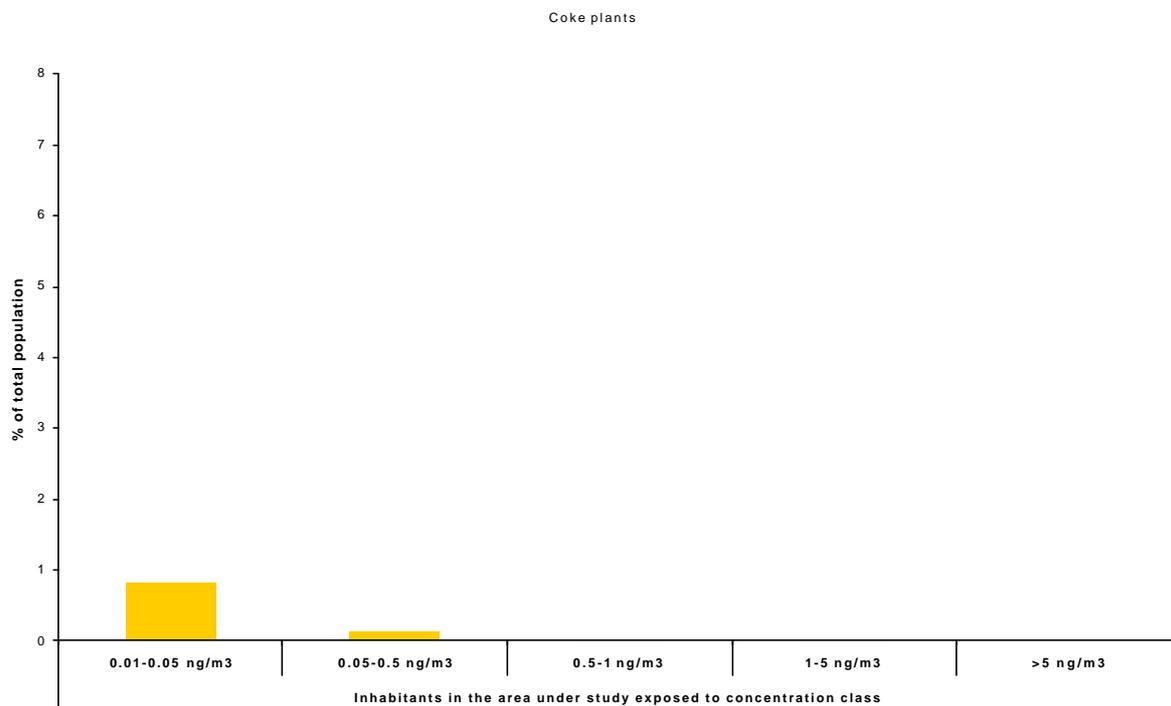
Population exposure was assessed in Section 4 of this report. Results are summarised in Figure 17 to Figure 19. The largest share of overall exposure is concentrated in urban areas. Cokeries (Figure 19) provide a relatively insignificant level of exposure.



**Figure 17. % of total exposure in the study area in settlements of less than 10,000 people.**



**Figure 18. % of total exposure in the study area in towns and cities of more than 10,000 people.**



**Figure 19. % of total exposure in the study area due to emissions from cokeries.**

## 6.3 METHODS

There are four stages in the quantification of benefits for PAHs following the quantification of exposure:

1. Quantification of the number of cancers linked to exposure using risk factors;
2. Calculation of recovery rates from the cancers linked to PAHs and of mortality rates;
3. Economic valuation of effects;
4. Consideration of secondary benefits and disbenefits of PAH abatement.

### 6.3.1 Quantification of the incidence and severity of cancers

The annual number of cancers linked to PAH emissions is calculated as:

$$N_{cancers} = Exposure_{B[a]P} \times population \times risk\ factor \div lifespan$$

Information on exposure and population were given above in Section 4:

Rural background: Table 9

Urban background: Table 10

Cokery hot-spots: Table 12

Soderberg plant hot-spots: Table 13

Traffic hot-spots: Table 16 and Table 18

Domestic hot-spots: Table 14 and Table 15.

Risk factors for lung cancer, the dominant effect of PAHs, were given above – a best estimate of  $87.10^{-6}$  lifetime risk per  $ng.m^{-3}$  exposure to B[a]P, where B[a]P is used as an indicator of total carcinogenic PAH exposure. A lifetime of 70 years is assumed, consistent with the calculations on which the risk factor is based. A factor four uncertainty is adopted for the risk factor, to provide maximum and minimum values (see the discussion around Table 29).

### 6.3.2 Valuation of PAH impacts

There are two elements to the valuation of lung cancers linked to PAH exposure. The first concerns the period of morbidity prior to cure or death and the second, death itself. In November 2000 DG ENV convened a workshop with experts from Europe and the USA to consider valuation of mortality and morbidity relating to air pollution. A best estimate and range for the value of statistical life (VOSL) was identified, together with a series of factors for adapting these values to specific problems, in this case. The following process was applied to derive values specific to lung cancer death and recovery:

- The starting point was a best estimate for the value of statistical life amongst the elderly of Euro 1 million, in a range of Euro 0.65 million to 2.5 million.
- Willingness to pay to avoid cancers was estimated at 0.5 times the VOSL<sup>3</sup>. This figure, the ‘cancer premium’, forms the basis for valuation of the proportion of cases likely to lead to recovery (in the case of lung cancers this is only 10%, EstEve *et al*, 1993). For fatal cases the cancer premium is added to the VOSL.
- There are strong theoretical and empirical grounds for believing that the value for preventing a fatality declines with age. Based on the results of research on valuation, the deaths of those aged less than 65 are given a value 1.43 times higher than those aged 65 years

<sup>3</sup> It is noted that evidence for the size of the cancer premium is minimal.

or older. Data on lung cancers shows 69% occur in the over 65s, and 31% in those younger than 65. Combining this information leads to a factor of 1.13 to correct for age.

- The next factor concerns the lag between exposure and effect. Data on trends in smoking and lung cancer incidence suggest a 20 year lag. Discounting over this period at the recommended rate of 4% introduces a factor of 0.456. Discounting at a rate of 2% introduces a factor of 0.673, and this figure is used to provide the upper estimate for the VOSL.

On this basis the VOSL relevant to the present case is calculated as:

$$(\text{baseline estimate} + \text{cancer premium}) \times \text{age factor} \times \text{discount factor}$$

$$\text{where the cancer premium} = 0.5 \times \text{baseline estimate}$$

The average value of a lung cancer is adjusted down to account for the 10% of cases that lead to recovery, and which are valued as:

$$\text{cancer premium} \times \text{age factor} \times \text{discount factor}$$

Final values are shown in Table 30.

**Table 30. Valuation of cancers related to air pollution, Euro M. The first block deals with fatal cancers, and the second block, non-fatal cancers. The final block averages these results in accordance with the proportion of cancers that will lead to death.**

<b>VOSL</b>	<b>Baseline estimate</b>	<b>Add cancer premium</b>	<b>Correct for age</b>	<b>Apply discount rate</b>
Low (4% discount rate)	0.65	0.98	1.1	0.50
Best (4% discount rate)	1.0	1.5	1.7	0.77
High (2% discount rate)	2.5	3.8	4.2	2.9
<b>WTP for avoidance of non-fatal cancers</b>	<b>Baseline estimate</b>	<b>Calculation of cancer premium</b>	<b>Correct for age</b>	<b>Apply discount rate</b>
Low (4% discount rate)	0.65	0.33	0.37	0.17
Best (4% discount rate)	1.0	0.50	0.57	0.26
High (2% discount rate)	2.5	1.3	1.4	0.95
<b>Average WTP for avoidance of lung cancers</b>	<b>Average assuming 90% of cases are fatal and 10% are recoverable</b>			
Low (4% discount rate)	0.47			
Best (4% discount rate)	0.72			
High (2% discount rate)	2.7			

### 6.3.3 Secondary benefits and disbenefits

Whilst secondary effects need to be identified clearly, the extent to which they should be factored into this analysis is open to debate. The logic of a Directive specifically on PAHs is surely that it should be defensible without consideration of other effects, such as reductions in noise or emissions of particles or greenhouse gases. Were such effects to play a major role in justifying a PAH standard, it would have to be asked whether that standard was justified - could not other, more cost-effective, measures be adopted for control of the secondary benefits?

Also, the methods for abatement identified in reports such as this are often not the methods that will be followed in practice as industry often finds alternative approaches. This could undermine the policy on PAHs, were it to be justified in terms of reductions in other burdens.

The secondary effects of PAH control may be both positive and negative;

**Positive:** for example where the measures taken to abate PAHs also abate other pollutants, such as fine particles, or increase the efficiency of plant operation, or:

**Negative:** for example when the application of emissions abatement reduces the efficiency of a process.

A listing of all identified secondary benefits and disbenefits is provided in Table 31, providing a breakdown by emission source and by abatement option. Some of the items listed are likely to be significant whilst others will not. The significance of many of the effects listed is hard to gauge, as in many cases they will be very site specific (for example, with respect to impacts on employment and local economies).

**Table 31. Secondary benefits/disbenefits of PAH control measures.**

Source	Abatement option	Secondary benefits	Secondary disbenefits
<b>Domestic burning</b>	Stove optimisation	Improved efficiency Reduced emissions of particles, greenhouse gases	Transfer of emissions, e.g. to coke works with respect to 'smokeless fuels'
	Use of alternative fuels	Possibly reduced emissions of particles, greenhouse gases	
	Energy efficiency	Reduced emissions of particles, greenhouse gases	Reduced choice for consumers Possible impacts on rural economies through restrictions on use of wood
	Prohibition	Reduced emissions of particles, greenhouse gases	
<b>Cokeries</b>	Relocation	Reduced exposure of the population to PM <sub>10</sub> , NO <sub>x</sub> , etc.	Increased emissions from transport to the new site? Material consumption for building the new plant Local reduction in employment Move of polluting technologies to other countries with less strict controls
	Prohibition		
<b>Aluminium smelters</b>	Relocation	Reduced exposure of the population to PM <sub>10</sub> , NO <sub>x</sub> , etc.	Increased emissions from transport to the new site? Material consumption for building the new plant Local reduction in employment Move of polluting technologies to other countries with less strict controls
	Prohibition	Possible stimulation for recycling	
	Conversion of Soderberg plant to pre-bake plant	Energy efficiency, and reduction in all associated emissions	
<b>Traffic</b>	Numerous non-technical measures for traffic abatement	Reduced urban emissions of particles, NO <sub>x</sub> , greenhouse gases, etc. Reduced congestion and noise	Some equity issues, depending on measures adopted.  Reduced fuel efficiency
	Particle traps	Reduced urban emissions of particles	
	Low emission zones	Reduced urban emissions of particles, greenhouse gases, etc. Reduced congestion and noise	Diversion of traffic to other areas Perceived reduction in choice for travellers Possible increase in lifecycle emissions taking account of the whole vehicle fleet?
	Scrappage subsidies	Reduced urban emissions of particles, NO <sub>x</sub> , greenhouse gases, etc.	
	Alternative fuels	Reduced urban emissions of particles, NO <sub>x</sub> , greenhouse gases, etc.	

## 6.4 RESULTS

This section first quantifies the total impact attributable to the following:

- urban background exposures
- rural background exposures
- hot-spots surrounding cokeries
- hot-spots surrounding Soderberg-process aluminium works
- hot-spots arising from domestic emissions
- hot-spots linked to traffic emissions.

It then proceeds to quantify the benefits that would be attributable to attainment of the limit values explored in the study.

### 6.4.1 Impacts attributable to general background concentrations

Estimated annual numbers of lung cancers arising in association with predicted background levels of B[a]P in 2010 are shown in Table 32, drawing on the exposure data presented in Table 9 and Table 10.

**Table 32. Estimated lung cancers linked to exposure to urban and rural background concentrations of PAHs.**

Country	Urban			Rural			Totals		
	Minimum	Best estimate	Maximum	Minimum	Best estimate	Maximum	Minimum	Best estimate	Maximum
Austria	1	4	16	1	3	12	2	7	28
Belgium	1	3	11	1	3	12	1	6	23
Czech Republic	1	3	10	0	2	7	1	4	17
Denmark	0	1	2	0	0	2	0	1	4
Estonia	0	0	1	0	0	1	0	0	1
Finland	0	1	4	0	0	0	0	1	5
France	10	38	153	1	4	17	11	42	169
Germany	7	30	119	1	4	16	8	34	135
Greece	0	1	4	0	0	1	0	1	5
Hungary	1	5	21	1	3	10	2	8	32
Ireland	0	0	1	0	0	0	0	0	1
Italy	6	23	92	1	2	8	6	25	101
Luxembourg	0	0	0	0	0	1	0	0	1
Netherlands	1	4	14	0	2	7	1	5	21
Poland	4	17	68	2	7	27	6	24	95
Portugal	0	1	4	0	1	2	0	2	6
Slovenia	0	0	1	0	0	1	0	0	2
Spain	1	4	14	0	1	6	1	5	20
Sweden	1	2	9	0	0	1	1	2	10
United Kingdom	3	11	43	0	1	3	3	11	45
<b>Total</b>	<b>37</b>	<b>147</b>	<b>588</b>	<b>8</b>	<b>33</b>	<b>133</b>	<b>45</b>	<b>180</b>	<b>721</b>

Table 33 attributes the impacts shown in Table 32 to source (cokeries, Soderberg smelters, residential, traffic and other sources), based on best estimates of impacts. Results are dominated by emissions from the residential sector.

**Table 33. Attribution to source of impacts (cases of lung cancer) of exposure to 'background' concentrations.**

Country	Cokeries	Soderberg	Residential	Traffic	Other	Total
Austria	0.43	0.01	6.10	0.16	0.22	7.0
Belgium	0.11	0.01	3.23	2.40	0.26	6.0
Czech Republic	0.06	0.03	3.63	0.18	0.10	4.0
Denmark	0.01	0.00	0.87	0.08	0.03	1.0
Estonia	0.00	0.00	0.22	0.01	0.02	0.2
Finland	0.00	0.00	0.94	0.02	0.04	1.0
France	0.50	0.02	35.78	3.44	2.10	42.0
Germany	0.61	0.10	27.20	5.03	1.05	34.0
Greece	0.01	0.00	0.54	0.14	0.31	1.0
Hungary	0.44	0.29	6.28	0.18	0.79	8.0
Ireland	0.00	0.00	0.23	0.01	0.00	0.3
Italy	0.63	0.02	21.08	1.40	1.63	25.0
Luxembourg	0.00	0.00	0.20	0.04	0.01	0.3
Netherlands	0.46	0.01	2.24	1.59	0.72	5.0
Poland	0.26	0.14	21.96	0.96	0.65	24.0
Portugal	0.01	0.01	1.89	0.08	0.01	2.0
Slovenia	0.00	0.00	0.44	0.04	0.00	0.5
Spain	0.44	0.13	3.85	0.48	0.10	5.0
Sweden	0.01	0.00	1.86	0.10	0.03	2.0
United Kingdom	0.15	0.04	7.59	2.73	0.48	11.0
<b>Total</b>	<b>4.1</b>	<b>0.8</b>	<b>147</b>	<b>19.1</b>	<b>8.7</b>	<b>180</b>

#### 6.4.2 Impacts attributable to cokeries in related hot-spots

Table 34 quantifies the number of cancers per year in each country, linked to exposure to PAHs from cokeries. Minimum and maximum values take account of variation in risk factors (a factor 4 either side of the best estimate) and variation in exposure within the concentration bands defined in Table 12. Compared to the effects of domestic and traffic emissions the quantified impacts are negligible, despite the very high PAH concentrations found in the vicinity of cokeries. According to the calculations made here, impacts in the vicinity of a cokery are significantly lower (total, best estimate, 0.32 cases/year) than the impacts arising from the contribution of cokery emissions to background concentrations (total, best estimate, 4.1 cases/year, from Table 33). This arises because the population in the area around cokeries where concentrations are significantly elevated is relatively low. Moving further afield, population exposed increases at a faster rate than the decline in concentrations linked to cokeries, with the result that the cumulative impact over long distances is greater than the impact close to the plant. This stresses the long distance transport of the pollutants concerned.

**Table 34. Estimated annual cases of lung cancer linked to exposure to PAHs from cokeries in the areas surrounding them, where concentrations are significantly elevated.**

<b>Country</b>	<b>Minimum</b>	<b>Maximum</b>	<b>Best estimate</b>
Austria	0.00041	0.034	0.0047
Belgium	0.00058	0.048	0.0066
Czech Republic	0.00066	0.057	0.0078
Finland	0.000043	0.0032	0.00045
France	0.0010	0.087	0.012
Germany	0.0087	0.84	0.11
Hungary	0.00016	0.012	0.0017
Italy	0.0011	0.099	0.013
Poland	0.0080	0.74	0.099
Portugal	0.00044	0.032	0.0045
Spain	0.00078	0.074	0.0099
Sweden	0.000028	0.0022	0.00031
The Netherlands	0.0023	0.21	0.028
United Kingdom	0.0016	0.13	0.018
<b>Total</b>	<b>0.026</b>	<b>2.37</b>	<b>0.32</b>

#### 6.4.3 Impacts attributable to Soderberg-process aluminium works in related hot-spots

Results of the analysis of the impacts associated with Soderberg plant are presented in Table 35. In this case, the minimum, maximum and best estimate figures shown are all based on the mid value for the risk factor, variation in the figures resulting from different assumptions on the population distribution around each plant in cases where data resolution was poor. The maximum figure (provided simply to scope the potential problems associated with Soderberg plant), is largely based on the assumption that the quantified local population reside at the site of emission, the exception being for the plant labelled Spain 2, for which better data on population distribution were easily available. In most cases few people, if any, will live within a kilometre of the site of emission. The minimum value is based on the assumption that the population is on average at a distance of 24 km from the plant, and the best estimate that they reside an average 4 km from the plant. The analysis for these plants was simplified compared to that used for cokeries because it is unlikely that any Soderberg plant will be operating in the EU21 in 2010. Results demonstrate that even if such plant are still in operation, associated effects on public health are expected to be small. The best estimate is only for 0.21 lung cancers per year amongst people living close to the plants, with estimates around individual plant varying in the range 0.02 to 0.05. Multiplying by a factor of 4 to account for variation in the risk factor increases the total annual number of cases to 0.84 across the six plants considered.

Comparing the results of Table 35 and Table 33 demonstrates the same type of relationship between the effects of emissions from Soderberg plant over short and long ranges as was observed for cokeries. Long-distance transport of PAHs again leads to a much higher impact than that observed over shorter distances (by a factor 4), though the results still appear insignificant when set against the impacts of other sources.

**Table 35. Estimated annual cases of lung cancer linked to exposure to PAHs from Soderberg-process aluminium works. The maximum figure represents an unrealistic worst case scenario, designed simply to place an absolute upper bound on the analysis.**

Plant	Population exposed	Number of cancers		
		Min	Max	Best estimate
Hungary 1	113,000	0.005	2.9	0.064
Poland 1	79,315	0.0042	2.3	0.051
Spain 1	55,758	0.0006	2.4	0.053
Spain 2	251,342	0.019	0.03	0.029
<b>Totals</b>	<b>651,415</b>	<b>0.029</b>	<b>7.6</b>	<b>0.20</b>

#### 6.4.4 Impacts attributable to hot-spots caused by domestic emissions

Table 36 shows the estimated number of cancers attributed to emissions from the domestic sector in areas where associated concentrations are elevated significantly. Total results are of a similar magnitude to those associated with exposure to background concentrations.

**Table 36. Impacts (number of lung cancers) attributable to domestic emissions of PAHs in non-background locations.**

Country	Minimum	Maximum	Best estimate
Austria	2.2	35.1	8.8
Belgium	0.5	7.9	2.0
Czech Republic	1.0	15.4	3.8
Denmark	0.6	9.5	2.4
Estonia	0.4	6.7	1.7
Finland	1.1	17.8	4.5
France	11.6	185.6	46.4
Germany	7.2	115.4	28.9
Greece	1.4	22.5	5.6
Hungary	1.5	23.3	5.8
Ireland	0.3	4.4	1.1
Italy	7.0	111.8	27.9
Luxembourg	0.0	0.4	0.1
Netherlands	0.3	5.4	1.3
Poland	5.1	80.9	20.2
Portugal	1.6	25.4	6.4
Slovenia	0.3	5.3	1.3
Spain	4.4	70.2	17.5
Sweden	1.1	18.3	4.6
United Kingdom	4.3	69.3	17.3
<b>Total</b>	<b>52</b>	<b>830</b>	<b>208</b>

#### 6.4.5 Impacts attributable to hot-spots caused by traffic emissions

Table 37 quantifies the number of cancers each year attributable to traffic emissions of PAH in urban areas. No account is taken of background exposures in this table.

**Table 37. Annual impacts (number of lung cancers) attributable to traffic emissions of PAHs in urban areas.**

<b>Country</b>	<b>Minimum</b>	<b>Maximum</b>	<b>Best estimate</b>
Austria	0.09	1.4	0.34
Belgium	0.14	2.4	0.60
Czech Republic	0.09	1.4	0.34
Denmark	0.06	0.89	0.23
Estonia	0.00	0.20	0.06
Finland	0.09	1.3	0.31
France	1.7	28	6.9
Germany	0.89	14	3.6
Greece	0.09	1.3	0.34
Hungary	0.11	1.6	0.40
Ireland	0.00	0.11	0.03
Italy	0.49	7.7	1.9
Luxembourg	0.00	0.06	0.00
Netherlands	0.11	2.0	0.49
Poland	0.34	5.6	1.4
Portugal	0.06	0.77	0.20
Slovenia	0.03	0.29	0.09
Spain	0.57	9.1	2.3
Sweden	0.09	1.3	0.34
United Kingdom	0.86	14	3.5
<b>Totals</b>	<b>5.8</b>	<b>93</b>	<b>23</b>

#### 6.5 COMPARISON OF RISKS WITH THE BENZENE STANDARD

The standard agreed in the EU for benzene is  $5 \mu\text{g}\cdot\text{m}^{-3}$  as an annual average, against a risk factor from WHO of  $6\cdot 10^{-6}/\mu\text{g}\cdot\text{m}^{-3}$  benzene. Combining these numbers gives an overall risk against the standard of  $30\cdot 10^{-6}$ . Applying the same overall risk to PAHs (taking the risk factor to be  $87\cdot 10^{-6}/\text{ng}\cdot\text{m}^{-3}$  B[a]P) would lead to a standard of  $0.34 \text{ ng}\cdot\text{m}^{-3}$ .

Against this a number of issues need to be considered that could affect the selection of standards:

- feasibility of meeting the standard
- cost of meeting the standard
- comparison of the severity of the diseases linked to each pollutant
- etc.

Taking these and other issues into account, it is of course not necessary to come to the same answer, in terms of risk, as was reached for benzene. However, it is relevant to consider

whether there is broad consistency in the limit values agreed for different pollutants, and where there is not, to understand the reasons why.

## 6.6 SUMMARY AND VALUATION

Estimates of the number of cancers and deaths linked to each source type are given in Table 38. Data for each country are given in Table 39. Valuation of these results is presented in Table 40 and Table 41, noting that adjustment of values in the Accession Countries has not been carried out.

**Table 38. Impacts of PAHs on health in 2010 (cases of lung cancer and associated deaths, best estimates only).**

Source of emission	Lung cancers	Deaths
Cokeries	4.4	4.0
Soderberg plant	1.0	0.9
Residential combustion	355	320
Traffic	42	38
Other	8.7	7.8
<b>Total</b>	<b>410</b>	<b>370</b>

Accounting for the factor 4 uncertainty in the risk factor, based on the variation seen across a range of studies, the annual number of lung cancers attributable to the PAH concentrations forecast for 2010 varies from 102 to 1,600. The estimated annual number of deaths ranges from 92 to 1,500.

Comparing these results to cancer death rates (WHO Mortality Database), suggests that were PAH related cases to be removed completely, lung cancer incidence would fall by between 0.05% and 0.8%, with a best estimate of 0.19%. Incidence for the EU21 is around 200,000 cases per year.

Valuation data suggest total damages attributable to PAH to be around Euro 300 million/year, in a range of Euro 192 million to Euro 1,100 million (accounting only for uncertainty in the valuation process, and noting the basis for the valuations as given in Table 30). Expanding these ranges to account for uncertainty in the risk factor extends the range to between Euro 48 million and Euro 4.4 billion/year. Note that at this stage the results simply show the total damages, rather than the benefits to be gained from reducing emissions in accordance with any particular possible limit value. This is addressed in the next chapter.

In this assessment it is arguable which assumptions should underpin the ‘overall best estimate’ with respect to valuation. It is clear that the effects of PAHs are not immediate – that they are related to cumulative dose – and as such there will be some lag between exposure and impact. However, the studies that are available give no guidance on the length of the lag. Here, a figure of 30 years is taken, roughly corresponding to half a lifetime, which accords with the view that it is lifetime exposure that is important.

**Table 39. Lung cancers per year in 2010 across the EU21, by country and source.**

	<b>Cokeries</b>	<b>Soderberg</b>	<b>Domestic</b>	<b>Traffic</b>	<b>Other</b>	<b>Total</b>
Austria	0.44	0.01	14.87	0.50	0.22	16.04
Belgium	0.11	0.01	5.22	3.00	0.26	8.60
Czech Republic	0.06	0.03	7.47	0.52	0.10	8.19
Denmark	0.01	0.00	3.24	0.31	0.03	3.59
Estonia	0.00	0.00	1.91	0.07	0.02	1.99
Finland	0.00	0.00	5.39	0.34	0.04	5.77
France	0.52	0.02	82.19	10.33	2.10	95.16
Germany	0.72	0.10	56.05	8.60	1.05	66.53
Greece	0.01	0.00	6.17	0.49	0.31	6.97
Hungary	0.44	0.36	12.10	0.58	0.79	14.28
Ireland	0.00	0.00	1.33	0.04	0.00	1.37
Italy	0.64	0.02	49.02	3.31	1.63	54.62
Luxembourg	0.00	0.00	0.29	0.04	0.01	0.35
Netherlands	0.48	0.01	3.58	2.07	0.72	6.86
Poland	0.36	0.20	42.18	2.36	0.65	45.75
Portugal	0.01	0.01	8.25	0.28	0.01	8.56
Slovenia	0.00	0.00	1.74	0.13	0.00	1.87
Spain	0.45	0.21	21.39	2.77	0.10	24.92
Sweden	0.01	0.00	6.43	0.44	0.03	6.92
United Kingdom	0.17	0.04	24.91	6.19	0.48	31.80
<b>Totals</b>	<b>4.4</b>	<b>1.00</b>	<b>355</b>	<b>42</b>	<b>8.7</b>	<b>410</b>

**Table 40. Valuation of cancers by source of emissions (Euro M). The variation shown (low/best estimate/high) relates only to uncertainty in the valuation process.**

	<b>Low</b> Euro 0.47M/case	<b>High</b> Euro 2.7M/case	<b>Best estimate</b> Euro 0.72M/case
Cokeries	2	12	3
Soderberg	0	3	1
Domestic	166	955	255
Traffic	20	114	31
Other	4	23	6
<b>Total</b>	<b>193</b>	<b>1107</b>	<b>295</b>

**Table 41. Valuation of cancers by country (Euro M). The variation shown (low/best estimate/high) relates only to uncertainty in the valuation process.**

	<b>Low</b> Euro 0.47M/case	<b>High</b> Euro 2.7M/case	<b>Best estimate</b> Euro 0.72M/case
Austria	8	43	12
Belgium	4	23	6
Czech Republic	4	22	6
Denmark	2	10	3
Estonia	1	5	1
Finland	3	16	4
France	45	257	69
Germany	31	180	48
Greece	3	19	5
Hungary	7	39	10
Ireland	1	4	1
Italy	26	147	39
Luxembourg	0	1	0
Netherlands	3	19	5
Poland	22	124	33
Portugal	4	23	6
Slovenia	1	5	1
Spain	12	67	18
Sweden	3	19	5
United Kingdom	15	86	23
<b>Totals</b>	<b>193</b>	<b>1107</b>	<b>295</b>

# 7 Comparison of costs and benefits

## 7.1 OVERVIEW

The results shown in several tables in this report demonstrate that achievement of the 0.01 and 0.05 ng.m<sup>-3</sup> limit values would in some cases require more than 99% abatement of emissions from 2010 levels. Table 42 demonstrates that the extent of the problem is severe even in locations with no significant local sources of PAH. Given the constraints that will apply in 2010 to future air quality improvements (as a result of legislation that is, or will be, implemented), this level of percentage abatement is not feasible, and the analysis therefore concentrates on the higher possible limit values ( $\geq 0.5$  ng.m<sup>-3</sup>).

**Table 42. Percentage abatement required to meet possible limit values in areas with no major local source contribution.**

Limit, ng.m <sup>-3</sup>	Rural background			Urban background				
	0.01	0.05	$\geq 0.5$	0.01	0.05	0.5	1	5
Austria	98%	89%	-	99%	96%	56%	12%	-
Belgium	98%	88%	-	98%	92%	23%	-	-
Czech Republic	96%	80%	-	98%	92%	17%	-	-
Denmark	92%	59%	-	96%	78%	-	-	-
Estonia	94%	69%	-	97%	84%	-	-	-
Finland	67%	-	-	97%	86%	-	-	-
France	92%	61%	-	99%	96%	60%	21%	-
Germany	91%	57%	-	99%	95%	46%	-	-
Greece	81%	7%	-	93%	67%	-	-	-
Hungary	97%	87%	-	99%	95%	49%	-	-
Ireland	75%	-	-	95%	77%	-	-	-
Italy	84%	18%	-	99%	95%	49%	-	-
Luxembourg	97%	86%	-	97%	86%	-	-	-
Netherlands	94%	69%	-	98%	90%	-	-	-
Poland	96%	81%	-	99%	96%	60%	20%	-
Portugal	84%	18%	-	97%	83%	-	-	-
Slovenia	93%	65%	-	97%	85%	-	-	-
Spain	87%	33%	-	95%	74%	-	-	-
Sweden	67%	-	-	98%	90%	-	-	-
UK	67%	-	-	97%	87%	-	-	-

The data in earlier sections provide information on the following types of location:

- a) Background sites with no major local source contribution,
- b) Areas around cokeries,
- c) Areas around Soderberg process aluminium plant,
- d) Rural areas with significant domestic heating emissions,
- e) Urban areas with significant domestic heating emissions,
- f) Locations near busy roads.

A problem for this analysis relates to the quantification of the worst exceedences likely. It is possible that hot-spots for industrial, domestic and traffic sources could all coincide, though this is likely to be a rare combination. An assessment of the extent to which they overlap would require a much more detailed analysis of concentrations than is possible at the present time for a Europe-wide analysis of peak concentrations. Situations [e] and [f] provide the greatest scope for significant multi-source contributions to hot-spots. The possibility for worse exceedences than those identified here should thus be considered, though it is not modelled.

Possible abatement options for industrial sites involve step-changes in emissions, whereas those for traffic and domestic combustion may involve a series of measures.

## 7.2 COSTS AND BENEFITS OF CONTROLLING INDUSTRIAL EMISSIONS

It is presumed that the desired reduction in emissions is determined by the house, school, hospital or other public building or facility at which the highest concentration of B[a]P is measured around any cokery or Soderberg smelter. Without detailed modelling around specific plant, including data on the distribution of houses, etc., it is not possible to assess how high that concentration will be with a good level of confidence. Clearly, modelling at that level of detail is outside the scope of this pan-European analysis. The results shown in Table 12 suggest that at least some of the population will be exposed to between 1 and 5 ng.m<sup>-3</sup> of B[a]P in all countries with cokeries. In the absence of other sources, and assuming a highest concentration attributable to an industrial facility of 3 ng.m<sup>-3</sup> at a location relevant to air quality planning, a reduction of 66% in emissions from the plant would be needed to meet a 1 ng.m<sup>-3</sup> limit, or 83% to meet a 0.5 ng.m<sup>-3</sup> limit. Likewise, assuming a highest concentration of 5 ng.m<sup>-3</sup> at a location relevant to air quality planning would lead to a need for reduction of 80% in emissions from the plant to meet a 1 ng.m<sup>-3</sup> limit, or 90% to meet a 0.5 ng.m<sup>-3</sup> limit. Adding in background concentrations attributable to other sources would obviously make the targets even more difficult to achieve.

For cokeries one option to further reduce emissions beyond BAT is plant closure. The iron and steel works typically associated with cokeries may then also close or rely on imported coke, presumably produced in a region of the world with a lower level of environmental protection. This would be against the spirit of the Framework Directive on Ambient Air Quality. A second option would be for the construction of a new plant that was capable of meeting PAH standards, though further assessment of available technologies is needed to see if this is possible. A third option is for the owners of a plant to purchase any affected properties. Comparison of costs and benefits of these options, to the extent that they are quantifiable, is given in Table 43.

**Table 43. Costs and benefits (Euro M) from abatement of PAHs from cokeries and Soderberg smelters to meet limits of 0.5 and 1.0 ng.m<sup>-3</sup>. The ranges for benefits include only effects of variation in valuations.**

<b>Plant/option</b>	<b>Annual benefits (Euro M)</b>	<b>Costs (Euro M)</b>
Cokeries		
Prohibition	3 (2 – 12)	unknown
Conversion to new plant able to meet limits	Up to 12	unknown, but Euro M <sup>1</sup>
Purchase of affected properties		
1 ng.m <sup>3</sup> limit	0.01-0.10	3,900 properties affected <sup>2</sup>
0.5 ng.m <sup>3</sup> limit	0.03-0.2	19,000 properties affected <sup>2</sup>
Soderberg smelters		
Plant modification	0.6 (0.4 to 2.5)	>5.4/year
Conversion to prebake process	Roughly 10% higher than plant modification	Likely to be an order of magnitude higher than the costs for plant modification

Notes:

- 1) Assuming that major design modifications would be needed, and plant is being rebuilt solely to meet PAH standards.
- 2) Assumes all affected properties are houses with an average occupancy of 2.5 people.

There is no unique ‘best solution’ of the options shown in Table 43. The purchase of affected properties appears particularly poor in terms of cost-effectiveness. Expanding the upper bound shown for the benefits to include variation in risk factors as well as variation in valuation parameters provides an upper bound benefit of Euro 800,000/year. With 19,000 properties affected this gives an average benefit per property of only Euro 40/year, a figure that is clearly well below the annual costs of housing which lie in the region of thousands of Euros for average properties. However, in some cases, where the population is extremely low in areas subject to exceedance, this may be the cheapest solution to the problem. Reference to Table 12 shows that in Finland and Sweden only 2 and 3 people (respectively) are estimated to be subject to levels in excess of 1 ng.m<sup>-3</sup>. In another case, where a plant is scheduled to be updated irrespective of concerns over PAH, upgrading to a standard that would meet the limit values may be the cheapest option.

For the Soderberg process, modification of plant would substantially reduce emissions, and is the most reasonable of the three measures identified. Costs would be dependent on the plant concerned. Results for annual costs and benefits are summarised in Table 43. The costs for improving plant are illustrative, assuming the cost of modification to be Euro 10 million per plant, discounting at 4% over a 15 year period (a cost of “at least Euro 10 million” was given in Section 5, so these costs should be regarded as a minimum). No account is taken of other aspects of cost, such as improvements in efficiency that may be expected at a new plant. Variation in benefit data shown in the Table relates to uncertainty in the estimate of the value of statistical life (VOSL). To account for uncertainty in the risk factor, a further factor 4 could be introduced. At the lower end of the range this would make benefits insignificant compared to costs. At the upper end benefits would rise to Euro 10 million/year, roughly double the costs shown for plant modification (though noting that these costs are minimum estimates). The

Table does not consider the limits of 0.5 and 1.0 ng.m<sup>-3</sup> separately, as the measures considered lead to a step change in emissions and would bring levels down below both limits.

The costs of the closure of cokeries would be experienced in 14 countries, Austria, Belgium, Czech Republic, Finland, France, Germany, Hungary, Italy, the Netherlands, Poland, Portugal, Spain, Sweden and the UK. Soderberg plant are thought to be present in only three countries, Hungary, Poland and Spain, and as noted elsewhere, may well be either closed or upgraded before 2010 in any case. Assuming this to be the case, both benefits and costs attributable to a new Directive on PAHs would fall to zero so far as aluminium production using the Soderberg process is concerned.

### 7.3 COSTS AND BENEFITS OF CONTROLLING RESIDENTIAL EMISSIONS

A variety of options for reducing residential emissions were identified in Section 5:

- Optimisation of existing stoves
- Conversion of open fireplaces to optimised stoves
- Information campaigns to encourage best practice in domestic use of solid fuels
- Switching to cleaner fuels
- Adoption of energy efficiency measures
- Prohibition of solid fuel burning

A major problem for analysis is the current limited state of knowledge regarding the residential sector. It is suggested that a major exercise be undertaken to better elaborate the potential for abatement in this area, to collect further data on fuel use and emissions, and to provide monitoring data that allow the results of studies such as this to be validated. Some reduction in PAH emissions will come about as a result of existing policies on other issues (see Table 23). A good example is the increased penetration of energy efficiency measures in response to concerns over global warming. In the absence of information to gauge the extent of implementation of these measures, the costs identified in this report should thus be regarded as a maximum. Estimated costs of the one option for which data are available, and associated benefits of reducing concentrations to 1 ng.m<sup>-3</sup> are shown in Table 44. Annualised (as opposed to total) costs will be of a broadly similar magnitude to the estimated benefits.

**Table 44. Costs and benefits of the control of PAH emissions from domestic sources to attain a limit of 1 ng.m<sup>-3</sup>.**

<b>Risk factor</b>	<b>Cases of lung cancer avoided</b>	<b>Benefit (Euro M/year)</b>	<b>Cost</b>
Low	51	24 (low) 37 (best estimate) 140 (high)	<b>Total</b> cost for attaining a limit value of 1 ng.m <sup>-3</sup> by optimising stoves is estimated at between Euro 2.5 and 3.7 billion.
Best estimate	205	96 (low) 150 (best estimate) 550 (high)	
High	820	390 (low)	

590 (best estimate)  
2200 (high)

## 7.4 COSTS AND BENEFITS OF CONTROLLING TRAFFIC EMISSIONS

Options identified for controlling emissions from vehicles in order to meet air quality objectives were reviewed in Section 5. The results shown in Section 4 demonstrated that significant exceedences of the limit values investigated here could arise, though they would affect a relatively small proportion of the population. In reality, the effect of earlier daughter Directives should lead to a reduction in exceedences for PAHs from those shown here. In the absence of thorough Europe-wide review of the problems of attaining the existing standards, individual countries should account for this in interpretation of the results of this study. As a result of the small population affected by the worst concentrations, the number of cases of lung cancer linked annually to these high levels is low, when compared, for example, to the number of cancers estimated to be linked to the effects of domestic sector emissions. Table 45 shows the level of abatement required in each country to meet proposed limit values in 2010 in 'traffic hot-spots'.

**Table 45. Required abatement in the worst estimated traffic hot-spots for PAHs in each country, including background contributions from other sources.**

Country	Required abatement (%)	
	0.5 ng.m <sup>-3</sup>	1 ng.m <sup>-3</sup>
Austria	68	36
Belgium	62	25
Cyprus	0	0
Czech Republic	41	0
Denmark	0	0
Estonia	21	0
Finland	49	0
France	75	50
Germany	41	0
Greece	0	0
Hungary	59	18
Ireland	17	0
Italy	46	0
Luxembourg	47	0
Netherlands	39	0
Poland	58	15
Portugal	29	0
Slovenia	0	0
Spain	24	0
Sweden	32	0
United Kingdom	32	0

For two countries at least (Austria and France) it is predicted that a limit value of  $1 \text{ ng.m}^{-3}$  could not be met solely through action to address hot-spot emissions of PAHs, in other words that other sources, particularly domestic emissions would also need to be targeted (though the need for this has already been identified above). Similarly, the limit of  $0.5 \text{ ng.m}^{-3}$  is predicted not to be met in cities in six countries – Austria, Belgium, France, Hungary, Italy and Poland, without action on sources outside the traffic hot-spots. This stresses the fact that PAH exposures are a multi-source problem.

The overall picture is mixed, however, with those countries not listed above predicted to be capable of meeting the limit value of  $1 \text{ ng.m}^{-3}$  in traffic hot-spots without any action. The benefits in terms of cases avoided against both the  $0.5$  and  $1 \text{ ng.m}^{-3}$  limits are shown in Table 46. The problem is forecast to be more serious in France than any other country, due to the high population and the large share of diesel in the fuels market.

**Table 46. Benefits in each country of controlling traffic related PAH levels to the limits of  $0.5$  and  $1 \text{ ng.m}^{-3}$ . Benefits quantified using the best estimate of Euro 0.72M/case.**

Limit	$0.5 \text{ ng.m}^{-3}$		$1 \text{ ng.m}^{-3}$	
	Avoided cases/year	Benefits, Euro M/year	Avoided cases/year	Benefits, Euro M/year
Austria	0.5	0.4	0.5	0.4
Belgium	0.8	0.6	0.1	0.1
Cyprus	0	0.0	0	0.0
Czech Republic	0.3	0.2	0	0.0
Denmark	0	0.0	0	0.0
Estonia	0	0.0	0	0.0
Finland	0.3	0.2	0	0.0
France	8.8	6.3	8.8	6.3
Germany	4.4	3.2	0	0.0
Greece	0	0.0	0	0.0
Hungary	0.5	0.4	0.2	0.1
Ireland	0	0.0	0	0.0
Italy	2.4	1.7	0	0.0
Luxembourg	0	0.0	0	0.0
Netherlands	0.5	0.4	0	0.0
Poland	1.8	1.3	0.2	0.1
Portugal	0	0.0	0	0.0
Slovenia	0	0.0	0	0.0
Spain	0.1	0.1	0	0.0
Sweden	0.3	0.2	0	0.0
United Kingdom	0.6	0.4	0	0.0
Totals	21.3	15.3	9.8	7.1

Estimated costs and benefits of meeting the two limit values are shown in Table 47. As in Table 46, results are based on the assumption that measures can be closely targeted at areas where exceedences will need to be addressed. This is reasonably realistic, given the nature of many of the non-technical options for air quality control. Given that exceedences of the standards were

identified for a relatively small area of the set of cities studied, it is probable that the main tool for dealing specifically with PAH exceedences from traffic would be the non-technical options described in Section 5. Technical improvements that are likely to come through, but are as yet not legislated for, would clearly assist in meeting the targets. An attempt to undertake a more detailed costing (beyond the indicative ranges in Table 47) has not been made here for a number of reasons. First, the level of abatement that may be achieved by different options varies considerably between cities according to local conditions – indeed some options will not be suitable to some cities (imagine, for example, a situation where there is no land available for a park and ride scheme close to the main roads serving a city). Second, the costs of abatement options will vary from place to place, according to general details of the local infrastructure and so on. Third, it is more appropriate to use transport models to investigate the cost-effectiveness of the available options, than to extrapolate from existing studies. The use of such models was beyond the scope of this pan-European exercise. Fourthly, individual countries are likely to have better data at their disposal for the assessment of options than was available to this work, including information on plans for meeting the existing daughter Directives. However, it has to be said that the benefits to be derived from abating vehicle emissions appear to be slight, when compared to the costs of some of the measures described in Section 5 (see Table 28).

**Table 47. Costs and benefits across the EU21 of achieving air quality targets for PAHs in cities through reduction of vehicle emissions.**

<b>Risk factor</b>	<b>Cases of lung cancer avoided</b>	<b>Benefit (Euro M/year)</b>	<b>Cost</b>	
<b>LIMIT VALUE: 1 ng.m<sup>-3</sup></b>				
Low	2.5	1.2 (low) 1.8 (best estimate) 6.8 (high)	Costs will range considerably from city to city depending on the severity of problems. Small, localised, exceedences may be addressed using very localised measures such as the use of bus priority systems to encourage the use of public transport. At the other extreme major road pricing schemes in large cities may cost in excess of Euro 100 million/year to run.	
Best estimate	9.8	4.6 (low) 7.1 (best estimate) 26 (high)		
High	39	18 (low) 28 (best estimate) 110 (high)		
<b>LIMIT VALUE: 0.5 ng.m<sup>-3</sup></b>				
Low	5.3	2.5 (low) 3.8 (best estimate) 14 (high)		
Best estimate	21	9.9 (low) 15 (best estimate) 57 (high)		
High	85	40 (low) 61 (best estimate) 230 (high)		

## 8 Conclusions

1. Although there is no direct evidence of human carcinogenicity against any individual PAH compound, the potential for these compounds to harm is established as a result of experimental work on animals, and epidemiological observations of workers exposed to mixtures of PAH compounds.
2. B[a]P, the most commonly monitored of the PAHs makes a reasonably consistent contribution to the carcinogenicity of PAH mixtures from the combustion sources that dominate the PAH emission inventory. It is therefore considered to be a suitable marker, although its contribution may well have been overstated in the past (through limiting the list of PAHs included in assessments). A further factor in its favour as a marker is that it is typically found bound to particles, and available evidence suggests that those PAHs that are bound to particles account for the major share of the overall carcinogenicity of PAH mixtures.

One source of PAH emissions that is not well represented by the use of B[a]P as a marker is wood preservation, for which the lighter fractions, such as fluoranthene dominate. Additional monitoring of fluoranthene would thus be useful.

3. Significant reductions in emission of B[a]P from what might be regarded as the 'traditional sources', particularly traffic, wood preservation, coke and primary aluminium production, will occur by 2010. Emissions from transport and wood preservation are set to decrease as a consequence of the implementation of already agreed EU legislation by 2010. Emissions from coke and aluminium production are predicted to decrease as a consequence of legislative measures, predicted lower production volumes, and (for the aluminium industry), trends away from the use of the more polluting Soderberg process.
4. Domestic consumption of solid fuels, in particular wood, is predicted to give the largest single contribution to total European B[a]P emissions – 90% by 2010. Despite this, the highest exceedences of possible limit values will arise from other sources, such as cokeries, because of the relative concentration of emissions. Only a small decrease in emissions from domestic solid fuel use is predicted between 1990 and 2010.
5. Estimated emissions of PAHs are subject to a high level of uncertainty. Indeed, the air quality working group under AOP-II regarded it as inappropriate to model traffic-related B[a]P, partly for this reason. In the course of this study it has become apparent that there are significant differences between different inventories that are unlikely to arise from real differences in emissions. The problems are clear when considering the numerous factors that affect emissions from the domestic sector which include:
  - System used (open fireplace, older style stove, new optimised stove).
  - Fuel used (wood, coal, peat, and waste materials).
  - Amount of fuel used (recent work at Eurostat suggests a much higher consumption of solid fuels in the home than previously reckoned).
  - Fuel characteristics (e.g. hardwood, softwood, moisture content, size, etc.).
  - Characteristics of the fire (size, airflow, etc.).
  - Deviation from 'best practice' in using solid fuels in the home.

Given the limited availability of emissions data from this sector it is not surprising that there are major uncertainties present. Further work is therefore needed to characterise PAH emissions. It is suggested that this should concentrate on traffic and domestic solid fuel burning, as these have been predicted to give rise to the greatest impacts in this study. Further work on emissions from wood preservation may also be worthwhile, given the difference in the PAH compounds emitted from this source.

6. Modelling work using the EUTREND model suggests that even rural background concentrations of B[a]P are high, as a result of widespread wood burning.
7. The study has considered three types of 'hot-spot' for B[a]P, linked to traffic, domestic and industrial sources. Of these, the highest concentrations are linked to cokeries and the few Soderberg process aluminium works that are still in operation. However, these point sources give rise to only very localised increases in concentration. The traffic and domestic sectors are predicted to give rise to lower peak concentrations, but much greater total exposure of the population because of the large number of small sources. At the present time there is a general lack of data for validation of the results of this modelling work. One particularly important issue relates to the distance between individual point sources (i.e. houses) which varies considerably in the types of village and town where problems might be expected. For example, the average distance between chimneys in a mining village may be less than 10 m, whereas in a more agricultural setting the average distance would be expected to be considerably greater. Results of this study do, however, agree reasonably well with available data.
8. There are very limited options available for reducing emissions from most existing cokeries, given the extent of legislation already in existence for the sector. With BAT in force, available options seem to cover only:
  - Prohibition of operations (without addressing demand issues this seems likely to shift production to less environmentally regulated economies, and so is considered to be against the spirit of the Framework Directive).
  - Major upgrading of plant to a level capable of meeting significantly lower concentration limits. This may not be possible, but needs to be considered.
  - Relocation of plant away from populations (there are numerous problems to this option, though it could be combined with redesign to reduce emissions).
  - Purchase of affected properties.
9. The situation with respect to Soderberg process aluminium works seems more straightforward. Most such plant operating in 1990 have either closed or been upgraded to alternative, less polluting processes, to the extent that this study has identified only four Soderberg plant in the EU21 that may be in operation by 2010. Given overall trends in the industry it is considered likely that these, too, will be closed or upgraded before the Directive comes into force. With BAT in force there are no options for significant PAH control short of major plant modification, plant conversion to alternative processes, or closure.
10. In contrast there are many options available for control of traffic emissions, which are mainly generated by diesel powered vehicles. These include technical measures (e.g. particle traps, catalysts, alternative fuels and drive-trains), economic measures imposed at the national level (e.g. fuel taxes, scrappage subsidies) and local 'non-technical' measures (e.g. encouraging

public transport and cycle use, improving traffic flows, road pricing). Many of these options are likely to be introduced in response to the first daughter Directive or concerns over noise or congestion.

11. There are also many options available for control of emissions from the domestic sector. However, further work is needed to improve understanding of the extent and reasons for problems in this sector. Reliable estimates of cost-effectiveness are available for few of the options available.
12. The best estimate for the unit lifetime risk factor for this study is selected as  $87.10^{-6}/\text{ng.m}^3$  of B[a]P for lung cancer incidence, with a factor 4 uncertainty higher and lower. No factors are readily available for quantification of bladder and skin cancers. From most occupational studies these appear less important, though recent work in Norway found association between B[a]P and bladder cancer. The same study reported no evidence for an association of B[a]P with lung cancer.

Dietary intake of PAH deposited to agricultural land may influence susceptibility to cancers other than lung cancer, though this risk cannot currently be quantified. The exposure pathways relevant to these cancers are likely to be dominated by PAHs generated through some processes for food preparation (such as smoking fish), though clearly some additional intake of PAH will arise from deposition to crops and pastureland.

13. No data are available for quantification of ecological impacts.
14. Comparison of risk factors for PAHs and benzene indicates that a target for PAH of  $0.34 \text{ ng.m}^{-3}$  B[a]P would provide an equivalent overall risk with the standard set in a previous daughter directive for benzene. However, other factors need to be considered – such as the feasibility and costs of meeting targets and the level of uncertainty in the analysis.
15. Estimated numbers of lung cancers attributable annually to PAH exposure from each source in 2010 under a business as usual type scenario are as follows. The figures given are best estimates. As noted in paragraph 12 there is approximately a factor 4 uncertainty both up and down in these :
 

- Cokeries:	4.4
- Soderberg process aluminium works:	1.0
- Traffic:	42
- Domestic:	355
- Others:	8.7
- Total	410

The death rate within 5 years of diagnosis of lung cancer is 90%.

16. The study has applied valuation data agreed from a meeting of European and US economists held in late 2000. A number of factors need to be accounted for:
  - ⇒ Starting values are 1.0 MEURO (best estimate), 0.65 MEURO (low) and 2.5MEURO (high).
  - ⇒ These values are adjusted upwards by a factor of 1.5 to take account of increased Willingness To Pay (WTP) against contracting cancer and costs of treatment.
  - ⇒ Data on the incidence of lung cancer against age shows that roughly 69% of cases occur in those aged over 65, and 31% in those under 65. The recommendation from the

Commission's meeting on valuation was that the death of those of more average age should be valued 1.43 times higher than deaths among the elderly, based on the results of contingent valuation studies. Accounting for this and the variability in age of those affected requires the use of a further factor of 1.13.

- ⇒ It has been assumed in this study that the lag between exposure and effect will be similar to that observed for smoking, 20 years. This introduces a factor of 0.46 (at the preferred 4% discount rate) or 0.67 (2% discount rate, included for sensitivity, providing the upper bound).
- ⇒ Finally account needs to be taken of the survival rate for lung cancer (which is only around 10%). This 10% of cases is valued at 1/3 the cost of fatalities.
- ⇒ This leads to average values per case of Euro 0.72 million, Euro 0.47 million (lower bound) and Euro 2.7 million (upper bound).

It is noted that the USEPA use a considerably higher starting point for costing mortality, around \$6million. However, as noted above, the Commission's meeting of experts did include representatives from both sides of the Atlantic. Further discussions of this issue will, no doubt, continue.

17. The best estimate of total damages from the 410 cases per year across the EU21 is therefore Euro 300 million/year.
18. The limited options for abatement from the industrial sector make it unlikely that the costs of abatement from cokeries and Soderberg plant could be justified on the basis of the benefits of reducing PAH emissions alone.
19. The situation with respect to traffic is far more complex. Many of the measures that could be introduced to reduce PAH levels will follow from legislation on other pollutants, and measures to reduce congestion in towns and cities. Attainment of a 1 ng.m<sup>-3</sup> limit value in areas subject to traffic hot-spots would reduce the number of cancers by an estimated 10/year for the EU21. Going further, to 0.5 ng.m<sup>-3</sup>, would reduce the number of cancers by 21/year. The limited extent of the problem linked to traffic means that it is unlikely that additional measures specifically directed to PAH abatement from traffic could be justified on cost-benefit grounds. The situation may well be different were the analysis widened to include the numerous other impacts related to traffic, such as emissions of other pollutants (PM<sub>10</sub>, VOCs, NO<sub>x</sub>, etc.), noise, congestion and accidents. The need to control traffic emissions to meet either a 1.0 or 0.5 ng.m<sup>-3</sup> limit could be significantly reduced by action on domestic emissions, given the contribution of these sources to background levels.
20. Domestic emissions are forecast to cause 86% of the lung cancers attributable to PAHs in 2010. Although a number of options have been identified for abatement there is a lack of cost-effectiveness data. However, comparison of the benefits of reducing emissions to meet a limit of 1 ng.m<sup>-3</sup> with the costs of stove optimisation suggests that the two are comparable. Other options, such as the use of alternative fuels need to be explored, though feasibility and costs of, e.g., the provision of natural gas to areas currently not served by pipelines are likely to be very site specific. Not surprisingly there are many other factors relevant to this debate, including emission of other pollutants and potential effects on the rural economy of restricting wood burning.

21. Further work is needed to refine methods for developing and reporting PAH emission inventories, and to monitor concentrations in a wider range of locations than has previously been done. Also, consensus is needed on the relative toxicity of different PAH compounds.
22. This study has traced through the pathway of emission, to concentration, impact and abatement for PAHs. Given the level of uncertainty in many of the inputs, results should only be considered indicative, though they provide a much better basis for discussion of the issues relating to exposure to airborne PAH than has existed previously.

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# Appendices

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- Appendix 2      Methodology used for domestic wood combustion
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# Appendix 1

## Methodology used for the application of coal-tar containing products

Coal tar comprises a mixture of numerous hydrocarbon components among which there are many PAHs. Coal tar is usually produced by condensation of coke oven gas. Historically there have been many applications of coal tar. The most relevant include application as fuel, for roof covering, as a pitch binder for road paving asphalts and as a wood preservative in the form of creosote and carbolineum. These applications all bring about some form of PAH release, the most important, historically, being wood preservation.

### ROAD PAVING AND ROOF COVERING

Coal tar has found application as a pitch binder for road paving asphalt. Heating of asphalt during processing or road paving results in some evaporation of PAH, a process highly dependent on the processing temperature. The European Asphalt Manufacturing Association (EAMA) however, has stated that all its members (which cover the whole of the studied area) have stopped using coal tar for asphalt manufacture in the course of the 1990s. The industry has switched to bituminous products that are regarded as being more environmentally friendly, and which have almost no PAH release when heated. It is therefore assumed that this source will be negligible in 2010. There was some concern in the Netherlands about how much PAH was released from existing roads on very hot days. Recent measurements have, however, demonstrated that these emission can be considered negligible. During re-heating of old asphalt for recycling, some emission might occur. Coal tar is not expected to be applied with roofing by 2010, primarily to avoid workers being exposed and also for environmental safety reasons.

### WOOD PRESERVATION WITH CREOSOTE

This section describes the methodology used to estimate the 1990 base level emission from wood preservation with creosote, as well as the 2010 projection for this source. The results for B[a]P, and the more volatile fluoranthene are presented. The emission estimates cover the preservation process of wood with creosote, the storage of treated wood on site, and the emissions during the use phase, i.e. after placement of the treated wood in the soil or water, depending on the application.

### Changes in composition of creosote during 1990 to 2010

For the purpose of this study, the following assumptions have been made regarding the composition of creosote. These figures are average estimates based on available literature:

- up to 1990, exclusively *conventional* creosote (WEI-A) was used with the following concentration of B[a]P: 0.0005 kg/kg creosote (0.05 %, range 0.01-0.10)
- between 1990 and 2010, *modified* creosote (WEI-C, GX, CCO440) is used partially besides the conventional type. The *modified* creosote has the following B[a]P concentration: 0.00002 kg/kg creosote (0.002 %, range 0.001-0.005)
- from the year 2010 onwards, exclusively *modified* creosote is used

### Other assumptions

- it is assumed that 10% of the PAH present in creosote is emitted to air at the preservation site (It is further assumed that this also applies to B[a]P. In practice it is obvious that the less volatile compounds (such as B[a]P) will have a relatively lower emission than the volatile ones (such as for instance fluoranthene). 10% is probably an overestimation for B[a]P but there is a lack of data to confirm this)
- from the creosote wood in the use phase, 0,5% of the PAH is lost annually, of which 30% to air (taking into account the distribution of emissions between water and air, and the relatively low vapour pressure of the B[a]P)
- the average retention of creosote in wood is 80 kg/m<sup>3</sup> (in 1990).
- the average lifetime of creosote wood is 36 years, which is an average for different applications varying from 15 to 75 years (For the year 2010 it is assumed that over the second half of the average lifetime of 36 years (i.e. from 1993 to 2010), modified creosote oil was used, and over the first half (1975-1992) conventional oil)

### Calculation of emissions in use phase

In general, the following formula can be used to calculate the total amount of wood preserved with creosote that is present in a country:

$$\text{total wood preserved with creosote} = \text{average annual input in past} \times \text{average lifetime of wood}$$

For the Netherlands this implies for 1990:

$$3,600,000 \text{ m}^3 = 100,000 \text{ m}^3/\text{year} \times 36 \text{ years}$$

Based on the foregoing assumptions, the emission of PAH can be calculated as follows:

$$\text{emission of substance} = \text{content of substance in creosote} \times \text{total wood preserved with creosote} \times \text{retention of creosote in wood} \times \text{percentage lost annually} \times \text{percentage emitted to air}$$

For The Netherlands (1990) this results in the following B[a]P emission during use:

$$5.0 \cdot 10^4 \text{ kg/kg creosote} \times 3,600,000 \text{ m}^3 \times 80 \text{ kg/m}^3 \times 0.5\%/y \times 30\% = 216 \text{ kg/y}$$

For the calculation of 2010 data, two additional assumptions need to be mentioned. First, it is assumed that the wood in use, i.e. the wood placed during the last 36 years contains 50% modified creosote and 50% conventional creosote. Second, due to reduction in demand, the annual placement of new creosote wood in 2010 will be half the amount placed in 1990.

For the Netherlands (2010) the B[a]P emissions is then as follows:

$$(0.0005+0.00002)/2 \text{ kg/kg} \times 1,800,000 \text{ m}^3 \times 80 \text{ kg/m}^3 \times 0,5\%/y \times 30\% = 56 \text{ kg/y}$$

*calculation of emissions during production*

The emission to air during production is a rough estimate of the emissions at the industrial site due to the preservation process and due to evaporation during storage of the treated wood. The emission is calculated as:

$$\text{emission during production} = \text{use of creosote oil} \times \text{fraction of substance in creosote} \times 10\% \text{ loss}$$

For the Netherlands in 1990, this results in

$$5.0 \cdot 10^4 \text{ kg/kg creosote} \times 9,252,000 \text{ kg/y} \times 10\% = 463 \text{ kg B[a]P/y}$$

For 2010 - assuming full use of modified creosote and assuming that half the amount of creosote oil will be used compared to 1990 - the emission will be:

$$2.0 \cdot 10^5 \text{ kg/kg creosote} \times 4,626,000 \text{ kg/y} \times 10\% = 9.3 \text{ kg/y}$$

The same calculations have been performed for the other countries under consideration. A data quality rating for the base data like usage rates is given in the table below. The same assumptions are used as for the Netherlands. The figures marked with \* were derived from extrapolation in the absence of other data and were based on the population of the specific country. The figures marked with\*\* have a relatively poor reliability.

<b>15 EU Members +6 Accession States</b>	<b>Data quality rating</b>
	<b>1990/2010</b>
Austria	*
Belgium	
Cyprus	*
Czech Republic	*/**
Denmark	**
Estonia	*
Finland	
France	
Germany	
Greece	*
Hungary	*
Ireland	**
Italy	
Luxembourg	*
Netherlands	
Poland	*
Portugal	*
Slovenia	*
Spain	
Sweden	
United Kingdom	
TOTAL EU	*/**
<hr/>	
*	derived from extrapolation
**	low certainty

The next table shows the results for 2010 for B[a]P from this source by country. All emission sources associated with creosote application have been summarised here. It shows that for heavy PAHs like B[a]P this source is basically insignificant, moreover the results are probably an overestimation. This is, however, not the case for more volatile PAHs which are often contained in much higher concentrations than B[a]P in creosote. Fluoranthene is an example of such a compound. Emissions are estimated in the same way as the earlier calculations for B[a]P. It is further assumed that the fraction of fluoranthene amounts to 0.06 kg/kg in conventional creosote and 0.12 kg/kg in modified creosote oil. In spite of reduction measures, the projected emission of fluoranthene in 2010 is still more than 600 tonnes for the area under study. In fact, heavy compounds like B[a]P might not be suitable PAH indicators for this type of evaporative sources at all. The uncertainty of these estimates is however very high, at least a factor of 3.

Country	B[a]P emission (kg)	
	1990	2010
Austria	104	12
Belgium	144	15
Cyprus	11	1
Czech Republic	302	35
Denmark	131	5
Estonia	25	3
Finland	145	10
France	239	26
Germany	957	97
Greece	104	12
Hungary	203	24
Ireland	57	12
Italy	363	33
Luxembourg	6	1
Netherlands	679	65
Poland	737	86
Portugal	108	13
Slovenia		
Spain	517	57
Sweden	320	36
United Kingdom	365	41
<b>Total</b>	<b>5612</b>	<b>595</b>

## WOOD PRESERVATION WITH CARBOLINEUM

This section describes the methodology used to estimate the B[a]P release in 2010 which occurs as a result of treating wood with carbolineum. It is presumed that emission occurs from wood treated in the past and from new application. The methodology covers both new application and evaporation from the existing stock of treated wood.

### Activity levels

The amount of existing treated wood has been estimated for the Netherlands within the framework of the Dutch Emission Registration. Since there are no data available for other countries, the results for the Netherlands have been extrapolated based on an estimated stock of

1.7 m<sup>2</sup> treated wood per capita. This value is for the late 1990s and by evaluation of trend data for earlier years a projected stock of 0.8 m<sup>2</sup>/capita/year is used for 2010. The same approach is used for the amount of freshly applied carbolineum which leads to an annual usage rate of 0.05 kg carbolineum/capita for the late 1990s. Assuming that the application rate in 2010 is half of this rate leads to an estimated usage of 0.3 kg/capita/year.

### Emission factors

Based on engineering judgments and limited measurement results, the following emission factors were derived by the Dutch Emission Registration:  $5.7 \cdot 10^{-7}$  kg B[a]P/mio m<sup>2</sup> wood for existing stock and  $1.4 \cdot 10^{-6}$  kg/mio kg carbolineum for fresh application.

### Results

The table below presents the results for B[a]P for 2010. From these results it is clear that in 2010 carbolineum use is probably not an important B[a]P source. This is primarily due to the low volatility of B[a]P. According to the same method the emission in 2010 for fluoranthene can also be calculated: 300 tonnes, which is significant. These estimates are very rough and have a high uncertainty, conservatively estimated to be at least a factor 3.

Country	B[a]P emission (kg)
Austria	6
Belgium	8
Denmark	4
Finland	4
France	46
Germany	63
Greece	8
Ireland	3
Italy	47
Luxembourg	0.3
Netherlands	12
Portugal	9
Spain	32
Sweden	7
United Kingdom	47
Cyprus	0.6
Czech Republic	13
Estonia	1
Hungary	9
Poland	31
Slovenia	
<b>Total</b>	<b>351</b>

# Appendix 2

## Methodology used for domestic wood combustion

Domestic combustion of wood is now becoming recognised as an important source of the release of both particles and PAH to air. In Europe, wood is used either as a fuel for domestic heating purposes in some areas, or in many homes principally for the cosmetic effect of a natural fire. In order to estimate the PAH release, it is necessary to obtain information on both the amounts of fuel wood used and the average specific emission (emission factor).

### Activity levels

Most of the wood used for domestic combustion is obtained through non-commercial channels. Large amounts of wood find their destination without being noticed by any central register. Therefore, officially reported figures often represent an underestimation of the actual use, and thus, it has always been difficult to make an accurate estimate of wood consumption. Wood consumption is usually not included by countries in preparing national energy balances. There are some statistics of fuel wood use compiled by IEA, FAO and UN/ECE. These are likely to be an underestimation. However, for various reasons, the importance of domestic wood combustion is becoming more and more recognized past years. Eurostat has recently issued a study on the domestic use of renewable fuels in the EU. The resulting statistics are composed of data on biomass obtained from the Member States with annual surveys and studies, financed by ALTENER and Eurostat [Eurostat Publication “Energy Consumption in Households”]. The methodologies used by the Member States are primarily aimed at obtaining the non-commercialised part of wood consumed, and are expected to represent the actual wood usage more realistically than earlier estimates. When comparing the Eurostat data to, for instance, the IEA or FAO statistics, it can immediately be observed that the Eurostat data are significantly higher than previously reported data.

The table below presents the use of wood for domestic combustion according to the best estimate, for all the countries in the studied area. Since no Eurostat figures are available for Norway, Switzerland and the Accession Countries this study has relied on other information and the table might represent an underestimation of the actual consumption in these countries. From evaluation of the Eurostat trend data (time series 1990-1997) it has been assumed that 2010 activity levels will approximate levels in the late 1990s.

Country code (ISO-3a)	Best estimate for 2010 (PJ)	Information source
AUT	117	Expert estimate by IIASA 1998
BEL	9	Eurostat 1999
CZE	11	FAO statistics
DNK	16	Eurostat 1999
EST	12	FAO statistics
FIN	47	Eurostat 1999
FRA	370	Eurostat 1999
DEU	154	Eurostat 1999
GRC	29	Eurostat 1999
HUN	32	FAO statistics
IRL	2	Eurostat 1999
ITA	227	Eurostat 1999
LUX	1	Eurostat 1999
NLD	8	Revised official Dutch expert estimate 1999
POL	36	FAO statistics
PRT	48	Eurostat 1999
SLN		FAO statistics
ESP	88	Eurostat 1999
SWE	43	Swedish expert estimate 1997
GBR	14	AEA technology 1995

## Emission factors

In addition to the problems of estimating wood consumption data, it is difficult to estimate an average emission factor for the emission for PAH from domestic wood combustion. There are many factors which are of great importance to the PAH formation mechanism and hence emissions. These factors include the type, shape and moisture content of the wood and the burning conditions such as size of the fire, temperature and oxygen supply. Many of these conditions vary significantly between fire-places and stoves. A way to proceed in order to estimate generalized emission factors is to base the factors on averages of a large number of measurement results under representative conditions. This has basically been the approach used for the technical emission factor manual [Berdowski, 1996] which has been prepared by TNO to assist countries in undertaking a national emission inventory. In the revised methodology for estimating emission from residential wood combustion which is described here, the emission factors for stoves and fireplaces have been used for all types of non-optimized wood combustion units. For B[a]P the emission factor is 0.17 g/GJ for both fire places and un-optimized stoves. Comparison with recent emission information confirmed the factors listed in [Berdowski, 1996] although it should be noted that uncertainties of the factors is high, easily up to a factor 2.

Several countries in Europe have an obligatory type approval for new wood stoves. This includes a certain degree of optimizing stove construction to minimise emission. In this study we assumed emission regulations for stoves to be already in place in Austria, Norway, Sweden, Denmark, Germany, the United Kingdom, Switzerland and the Netherlands. For the Netherlands TNO-MEP certifies each new wood stove. Based on experience gained with this, a 45% reduction of PAH emission is estimated for emission-optimized stoves. Other experience in the Netherlands suggests that this type of stove penetrates the market at a rate of 4% annually

since entry into force of policy (resulting in an assumption of about 60% penetration in 2010, for the countries mentioned). Furthermore it is assumed that by 2010, as a result of the future CEN and EU measures, optimised stoves have penetrated by 20% in the other EU countries, and there are no optimized stoves yet in the Accession countries. Finally it is assumed that about half of all wood combustion units comprise stoves and the other half, fire places (for which there is very limited potential for optimisation).

### **Spatial distribution**

A default approach for estimating spatial distribution of emissions from domestic wood combustion would be simply to use population density as a proxy. Although this might be valid in general, this method has some serious drawbacks. Experience shows that residential areas where wood burning is significant are usually not densely populated city centers. Instead, wood burning occurs more in sub-urban or in rural regions. There can be various (country-specific) reasons for using wood as a fuel. In absence of other fuels like natural gas, residents may be forced to use fuels like wood or coal. In many homes however, wood is burned as a luxury rather than a necessity. Houses which are fitted with a fireplace or stove tend to be low-rise and often free standing (with possibilities for wood storage) rather than high-rise dwellings. There should also be a supply of wood nearby.

Since domestic wood burning appears such an important source, an alternative method for distributing wood combustion emissions was developed. The general idea has been to exclude urban regions (cities) from the distribution map. This has been accomplished as follows. WHO has compiled a list of cities and towns (including co-ordinates, covering the entire study area), larger than 10,000 inhabitants. It has been assumed that the specific wood consumption (consumption/capita) in these cities is only 10% of consumption outside these cities. With this, the population density map has been corrected for urban areas prior to distributing the wood combustion emissions. Using this "rural population" density map probably gives a more realistic geographical distribution compared to simple population density. It should be noted that even after correction there still remain high wood combustion densities in some urban regions (like for instance the Paris region).

### **Emission factors assumed for coke and aluminium production, and road transport**

On average it is estimated that compared to the situation in 1990, 90-95% emission reduction for B[a]P is achieved in 2010, after implementation of BAT in the iron and steel and non-ferrous metals production sector.

#### **Coke ovens:**

Using the data on coke plants available from the EU IPPC BREF document on the iron and steel production sector and assuming a B[a]P fraction of 23% in reference to the Borneff-6 plus estimating 25% Borneff-6 in total PAH, an emission factor of approximately 5 mg B[a]P/tonne coke could be derived. Two approaches have been followed here: taking the lower value of the range given on page 122 and alternatively, the lower values of the source contributions given on page 123-124.

The TNO-Technical Manual to the UN/ECE – OSPARCOM - HELCOM inventory reports 750 mg B[a]P per tonne coke produced. In the technical Annex of the UN-ECE HM/POP is

stated that reduction above 95% can be achieved using BAT, which would point towards 20-40 mg/tonne coke. Information from a large iron and steel plant in the Netherlands confirms this. Taking all of the above into account a first estimate for the emission factor for coke production after implementation of BAT would be about 20 mg B[a]P/tonne coke produced.

### **Aluminium smelters:**

The B[a]P emission from primary Soderberg-based aluminium smelters with horizontal contact bolts is approximately 101 g/tonne Al, as is reported on the TNO-Technical Manual to the UN/ECE – OSPARCOM - HELCOM inventory. According to the technical annex to the UN/ECE HM/POP Protocols a reduction of more than 90% can be possible for this process. This would indicate 5-10 g B[a]P/tonne Al. This is more or less consistent with the lower ranges which are reported for Soderberg aluminium in the EU IPPC BREF document on non-ferrous metals production (5 g/tonne). Hence an emission factor of 5 g B[a]P/tonne Soderberg Al has been used. Separate anode and pre-baked anodes-based production processes have been disregarded as their contribution is not expected to be significant.

### **Road transport:**

Basically the emission data for road transport have been taken from the earlier work performed by TNO for the EU Priority studies. There is extensive EU policy on vehicle emissions and also the UN/ECE POP Protocol suggests a number of technologies which can be considered as BAT for mobile sources. In the EU Priority studies TNO has assumed a gradual penetration in Europe of technologies (EURO I, II, III and IV). TNO has prepared an additional estimate of the effects of the measures on B[a]P by looking at the effect of the measures for particulate matter and to a lesser extent NMVOC's..

# Appendix 3

## Background data on effects

**Table 1. The degree of evidence for carcinogenicity of various PAH compounds in experimental animals and, and overall evaluations of carcinogenicity to humans evaluated by IARC (1983; 1987). Table taken from the PAH Working Group Position Paper.**

PAH Compound	Number of Rings	IARC	
		Animals	Humans
Anthracene	3	I	3
Phenanthrene	3	I	3
Benzo[c]phenanthrene	4	L	3
Chrysene	4	S	3
Benz(a)anthracene	4	I	2A
Pyrene	4	I	3
Triphenylene	4	I	3
Benzo(a)pyrene	5	S	2A
Benzo[e]pyrene	5	I	3
Dibenz[a,c]anthracene	5	L	3
Dibenz[a,h]anthracene	5	S	2A
Dibenz[a,j]anthracene	5	L	3
Perylene	5	I	3
Anthanthrene	6	L	3
Benzo[ghi]perylene	6	I	3
Coronene	6	I	3
Dibenzo[a,e]pyrene	6	S	2B
Dibenzo[a,h]pyrene	6	S	2B
Dibenzo[a,l]pyrene	6	S	2B
Dibenzo[a,i]pyrene	6	S	2B

**Key:**

(I: inadequate evidence; L: limited evidence; S: sufficient evidence; 2A: probably carcinogenic to humans; 2B: possibly carcinogenic to humans; 3: not classifiable)

# Appendix 4

## Costs of carrying out PAH Measurements

The following data on the costs of measurements in four countries (Austria, Germany, Italy and United Kingdom) were collected by the Mark Hailwood of the Working Group. It is reproduced here partly because of the relevance of the cost data to this exercise, and partly to underline the large variation in monitoring approaches used in Europe, that makes validation of the results of the modelling undertaken here so problematic.

### AUSTRIA: - UBA

**Sample size:** 350 m<sup>3</sup>, 24 hour  
**Sample:** PUF + Filter - no use of PM<sub>10</sub> sample  
**Detection:** GC-MS, 0.05 ng BaP detection limit

**Costs:** DIN PAH (6 WHO) EURO 350  
 EPA16 PAH EURO 500

Costs include material, preparation, analysis.  
 Costs do not include transportation to and from sites.

### GERMANY: - UMEG, Karlsruhe

**Sample size:** 720 m<sup>3</sup>, 24 hour 150 mm filter (equivalent to 80m<sup>3</sup> over 47 mm diameter filter portion)  
**Sample:** PM<sub>10</sub> Filter

The samples for PAH analysis are 47mm discs which are stamped out of the 150 mm filter. Validation experiments have shown that the filter loading is sufficiently uniform from this method to be used. This allows the same 150 mm filter to be used for PM<sub>10</sub>, metals, and PAH.

**Sampler:** Digital DHA-80 with automatic Filter changer  
**Detection:** HPLC Fluorescence, 0.05 ng BaP detection limit  
 Accuracy: +/- 10% at 1 ng/m<sup>3</sup> BaP, +/- 30% at 0.1 ng/m<sup>3</sup> BaP  
 Standard deviation: max. 25% at 0.05 ng/m<sup>3</sup>, less than 10% at values above the detection limit

**Costs:** for 2000 samples PM<sub>10</sub> per year EURO 100  
 for 6000 samples PM<sub>10</sub> per year EURO 40

Costs are the analytical costs based on using PM<sub>10</sub> filter. The sampler is housed in a container (sea container) which is air conditioned to 20°C. The sample may spend up to two weeks in the container once collected. However in winter the loss of particle bound PAH in winter is not detectable and in summer is within the measurement error.

Experiments carried out by UMEG have shown that:

- a) in the cold winter months no loss is expected over 14 days in the sampler for particle bound PAH
- b) for BeP, BbF, BkF, BaP, BghiP and I123cdP for temperatures of up to 35 °C and a storage period in the sampler of 1 week no losses are to be expected.
- c) For BaP in Summer for a storage period of 14 days in the sampler losses can be expected if the sampler is not air conditioned. Experiments carried out by UMEG have shown that samplers exposed to the sun can reach high internal temperatures and losses of up to 50% may occur. Concentrations in the summer months are generally much lower and the measurement uncertainty at these low concentrations is higher

Samples are not bulked. The extraction is carried out using a validated method on a micro-scale using toluene.

### **ITALY: - based on public laboratories**

**Sample size:** 1600 m<sup>3</sup>, 24 hour

**Costs:** 1 Filter sample (20 cm x 25 cm) EURO 16  
Additional cost to a PM10 gravimetric measurement EURO13

150 Filter samples per year and 40 Stations  
individually measured EURO 95 000  
bulked by month EURO 53 000

1 PUF(20 cm x 25 cm x 5 cm) + Filter sample EURO 110  
48 PUF + Filter samples per year bulked by month for five control stations  
EURO 24 000

Costs are materials only

Costs do not include laboratory costs or transportation to and from sites.

### **UNITED KINGDOM: - AEA Technology, analysis undertaken by Harwell Scientifics Ltd.**

**Sample size:** 700 - 1600 m<sup>3</sup> over 2 weeks

The canister (PUF + Filter + isotopically labelled PAH sampling spike) spends two weeks in the field at ambient temperature.

**Analysis:** HPLC UV Fluorescence detector and GC-MS used.

**Costs:**

**Canister Preparation Costs:** PUF + Filter (preparation and raw materials) EUR 80

**Analysis Costs:**

(i.) Samples taken within a three month period are bulked and then analysed for 30 PAH EURO 450 (Extraction and analysis cost only.)

(ii) canisters individually analysed for the EPA 16 EURO 350 (Extraction and analysis cost only.)

**Costs of network operation on a per canister basis:** estimated to be EURO 370

(This cost includes the preparation of the canister, raw materials, data management, transport of canisters, sampler operation, electricity costs, sampler calibration and maintenance visits, reporting, interpretation of results, facilities and computer charges, this estimate does not include analysis costs).

This cost could vary if the size or structure of the network change. This however is an estimation of the present costs on a per canister basis.

**CONCLUSIONS**

As can be seen the costs given vary widely due to the fact that no two methods are sufficiently comparable. However, the following conclusions may be drawn:

1. The analysis of the particulate fraction alone, as compared to the vapour and particulate fractions together, has a much lower cost. This is due to the high solvent cost involved in extracting the PU-foam.
2. The costs incurred in bringing out the filters and returning them to the analytical laboratory have not been assessed in any of the answers, but they could be highly significant. Large distances between sample site and laboratory will increase the costs by several hundred Euro per measurement. The use of PM<sub>10</sub> filters as in Germany (UMEG) allow a higher cost effectiveness of the sampling.
3. The greatest deficit in the comparability is due to the variation in sampling conditions (sampling time, storage before extraction, etc.) rather than analytical conditions. The conditions need to be comparable and standardised (possibly CEN).
4. For measurements to be compared across Europe there needs to be a standardisation of the sampling and analytical methods. The methods described above will not guarantee any high degree of comparability as they are.