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RIJKSINSTITUUT VOOR VOLKSGEZONDHEID EN MILIEU
NATIONAL INSTITUTE OF PUBLIC HEALTH AND THE ENVIRONMENT

RIVM report 481505015

Technical Report on Chemicals, particulate matter and human health, air quality and noise

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May 2000

TME and TNO under contract with the Environment Directorate-General of the European Commission.

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Abstract

The economic assessment of priorities for a European environmental policy plan focuses on twelve identified Prominent European Environmental Problems such as climate change, chemical risks and biodiversity. The study, commissioned by the European Commission (DG Environment) to a European consortium led by RIVM, provides a basis for priority setting for European environmental policy planning in support of the sixth Environmental Action Programme as follow-up of the current fifth Environmental Action Plan called 'Towards Sustainability'. The analysis is based on an examination of the cost of avoided damage, environmental expenditures, risk assessment, public opinion, social incidence and sustainability. The study incorporates information on targets, scenario results, and policy options and measures including their costs and benefits.

Main findings of the study are the following. Current trends show that if all existing policies are fully implemented and enforced, the European Union will be successful in reducing pressures on the environment. However, damage to human health and ecosystems can be substantially reduced with accelerated policies. The implementation costs of these additional policies will not exceed the environmental benefits and the impact on the economy is manageable. This requires future policies to focus on least-cost solutions and follow an integrated approach. Nevertheless, these policies will not be adequate for achieving all policy objectives. Remaining major problems are the excess load of nitrogen in the ecosystem, exceedance of air quality guidelines (especially particulate matter), noise nuisance and biodiversity loss.

This report is one of a series supporting the main report: *European Environmental Priorities: an Integrated Economic and Environmental Assessment*. The areas discussed in the main report are fully documented in the various *Technical reports*. A background report is presented for each environmental issue giving an outline of the problem and its relationship to economic sectors and other issues; the benefits and the cost-benefit analysis; and the policy responses. Additional reports outline the benefits methodology, the EU enlargement issue and the macro-economic consequences of the scenarios.

Technical Report on Chemicals, particulate matter and human health, air quality and noise

This Report has been prepared by RIVM, EFTEC, NTUA and IIASA in association with TME and TNO under contract with the Environment Directorate-General of the European Commission. This report is one of a series of reports supporting the main report: *European Environmental Priorities: an Integrated Economic and Environmental Assessment*. Reports in this series have been subject to limited peer review.

The report consists of four parts:

Section 1: **Chemicals and particulate matter**
Prepared by Winand Smeets and Addo van Pul (RIVM)
in close collaboration with Antoon Visschedijk and Tinus Pulles (TNO)
with contributions from Drs. G.J. Reinds and Dr. W. de Vries (Alterra, The Netherlands).

Section 2: **Human health and air quality**
Prepared by Hans Eerens and Rob Sluyter (RIVM);
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Section 3: **Benefit assessment**
Prepared by D.W. Pearce, A. Howarth (EFTEC)

Section 4: **Policy assessment**
Prepared by D.W. Pearce, A. Howarth (EFTEC)

References
All references made in the sections on benefit and policy assessment have been brought together in the Technical Report on Benefit Assessment Methodology. The references made in the sections 1 and 2 on environmental assessment follows at the end of their section.

Note to the reader:
This 'technical report' provides background information on two issues in the main report: 'Chemicals and particulate matter' and 'Human Health and Air Quality'.
There are four sections.
Section 1 deals with a) emissions and costs of emission abatement of primary particulate matter and persistent organic pollutants (POPs), such as dioxins, and b) emissions and depositions of heavy metals, which are often attached to primary particulate matter, and of some pesticides.
Section 2 deals with air quality in urban and rural areas for a selected number of pollutants:

- primary particulate matter, lead, B(a)P, and benzene and
- secondary particulate matter (from SO_2, NO_x and NH_3), and SO_2, NO_x, of which the emissions are dealt with in the context of acidification and eutrophication (*see Technical Report on Acidification*).

Economic Benefits with regard to human health are dealt with in Section 3, where also cost-benefit ratios (especially for particulate matter) are presented.
Finally, Section 4 deals with a set of possible policy measures to reduce emissions of pollutants (*see section 1*) beyond baseline levels.

Contents

1. Chemicals and particulate matter

1.1 Introduction

This section on chemicals and particulate matter considers the future emission trends to air of primary particulate matter (PM), heavy metals (HMs) and persistent organic pollutants (POPs), and also briefly discusses deposition for some selected HMs (cadmium, copper, lead), and POPs (dioxins/furans, atrazine, endosulfan, lindane, pentacholorophenol). In addition, impacts on forest soils are evaluated for cadmium, copper and lead. Human exposure to selected air pollutants (PM, Benz(a)Pyrene, benzene, lead) is dealt with in a separate section (see section 2). Impacts of HMs on non-forest ecosystems and impacts of POPs have not been assessed.

Primary particulate matter consists of particles emitted from anthropogenic and natural sources such as combustion, industrial processes[1], sea-salt spray and suspended soil dust. Secondary PM is formed by chemical reaction from SO_2, NO_x, and NH_3 gases, condensation of organic vapours emitted from various anthropogenic sources and photochemical reactions. This study uses PM_{10}[2] as main indicator for the effects of human exposure to particulates. Emission trends presented in this section only relate to emissions from anthropogenic origin.

Chemicals are introduced into the environment through human activities. These include the release from production and use, but also the dispersion as unwanted by-products during combustion and industrial processes. Chemicals are dispersed into the air, water and soil, and may lead to unwanted effects on human health and ecosystems.

Among the large number of chemicals entering the environment, HMs and POPs represent two groups that are of particular importance due to their persistent, bio-accumulative and toxic characteristics. HMs and POPs are known to be a threat to human health (blood and organ disorders, carcinogenic effects, birth defects, intellectual development) and the environment (forest ecosystem stress, reproductive impairment). Clearly, there are tens of thousands of chemicals (including pesticides) that could be considered, but this study will focus on emissions of heavy metals (HMs) and persistent organic pollutants (POPs) to air that are subject to EU, UN-ECE and other international agreements and for which a reasonable amount of data exists. The environmental problems around chemicals (or hazardous substances) are presented in detail in the state-of-environment reports brought out by the EEA (EEA, 1998; 1999). The risk assessments of new and existing non-assessed chemicals dealt with in EU regulations are discussed in Box 1.1.

Emission targets have been established for specific HMs and POPs under the auspices of the UN/ECE Convention on Long-Range Transboundary Air Pollution (CLRTAP). According to the protocols on HMs and POPs, countries are obliged to reduce atmospheric emissions of lead (Pb), cadmium (Cd), mercury (Hg), dioxins/furans and polycyclic aromatic hydrocarbons (PAHs) to below a reference year, most probably 1990 for the EU. Emissions of these substances, together with copper (Cu), which is not covered by the CLRTAP Protocol, are used as the main pressure indicators for chemicals in this study. Future emissions of polychlorinated biphenyls (PCBs) and pentachlorophenol (PCP) are only briefly referenced due to the effective control on emissions through current EU regulations. Four agricultural pesticides (atrazine, endosulfan, lindane and pentachlorophenol (PCP)) have been considered in the baseline scenario only.

Three principal emission scenarios have been assessed in this study. Making use of a 1990-2010 timeframe, future trends in emissions under current legislation were assessed in the baseline scenario (BL), while the technology-driven scenario (TD) assumes full penetration of advanced end-of-pipe emission control technologies, such as high efficiency electrostatic precipitators, fabric filters and highly efficient wet scrubbers. The accelerated policy scenario (AP) takes into account the effects of policy action on climate

[1] Non-combustion related emission sources in industry (stack plus diffuse emissions).
[2] PM_{10} are particles with diameter less than 10 μm that can follow the inhaled air into the respiratory system and the lungs.

change and acidification. Advanced end-of-pipe technologies to reduce emissions of primary PM_{10} and selected HMs and POPs are considered in the AP scenario also.

Box 1.1.: The assessment of chemicals

The output of the chemical industry worldwide is almost € 1500 billion per year. A 30% share in this output makes the EU a major player on the global market. Within the EU there is sufficient regulatory legislation to adequately reduce risks associated with chemical substances. Although existing assessment procedures implied in current legislation can always be improved, they should not be regarded as a significant bottleneck in the proper handling of chemical risks. The efficacy of directives and international agreements in substantially reducing chemical risks varies greatly. Whereas risks associated with the introduction of new chemicals can be largely avoided, the degree of manageability of risks associated with existing substances is not sufficient.

Adequate management of chemical risks implies targeted risk reduction measures that are based on risk assessments when an apparent concern has been established for a given chemical. The EU started assessing the risks of the 100,000 existing chemicals in 1993, giving priority to the 2,500 so-called High Volume Production Chemicals (HVPCs; >1,000 tonnes per year). Since then, the risks of some 30-40 chemicals have been assessed. For a few chemicals risks were sufficiently high to warrant proposal of proper risk management programmes to be adopted by the Commission. At this pace it will take ages to assess all HVPCs adequately. Assessment costs vary from € 100,000 for a basic set of toxicity data to an estimated € 5 million for comprehensive toxicity testing of one substance.

Full risk assessment of more HVPCs is prevented due to inadequate toxicity information (for 75% of these HVPCs minimal toxicity data for a preliminary assessment are lacking). In many cases where this information is present, limited or lack of information on emissions and exposure prevents further action.

To overcome these obstacles, a joint EU-wide professional organisation is needed to promote and monitor progress in producing adequate and free access (eco-) toxicity information on existing chemicals, and substances that fall into special categories, such as biocides, pharmaceuticals, etc. A recent study recommended improving the integration of the myriad of directives and regulations. The aim was to clarify definitions, provide clear guidance on the determination and weighing of advantages and implications of risk reductions measures and to develop tools, including voluntary agreements, to speed up the slow chemical-by-chemical approach [Van Leeuwen et al., 1996].

1.2 Methodology

The environmental risks of PM_{10}, selected HMs and selected POPs were evaluated following a Driving forces – Pressure – State – Impact – Response (DPSIR) analysis. The Driving forces and the subsequent *Pressure* in terms of the emissions to air were determined first (PM_{10}, Cd, Cu, Pb, Hg, dioxins/furans, PAHs, benzene and the pesticides atrazine, endosulfan, lindane and pentachlorophenol). Results of such calculations at the national level are presented in section 1.3. Urban emissions are discussed in a separate section on human health and air pollution (see section 2 of this technical report). Section 1.3. also briefly discusses European-scale deposition for some selected HMs (Cd, Cu, Pb), and POPs (dioxins/furans, atrazine, endosulfan, lindane, PCP) (*State*). In addition, exceedances of critical loads for accumulation of HMs (Cd, Cu, Pb) in forest soils are evaluated [3] (*Impact*). Human exposure to selected air pollutants (PM_{10}, B(a)P, benzene, Pb) is dealt with in a separate human health and air pollution section (see section 2.). Impacts of HMs and POPs on non-forest ecosystems have not been assessed.

1.2.1 Emissions

1.2.1.1 Methodology for emission calculation

Pollutant definition
For PAHs only a limited set of indicator components have been studied: benz(a)pyrene, benzo(b)fluoranthene, benzo(ghi)perylene, benzo(k)fluoranthene, fluoranthene, indeno(1,2,3-c,d)perylene. These six are known as the '6 Borneff PAH'. In the case of PCB either all PCB (when dealing with leakage) or six indicator PCB (PCB 28, 52, 101, 118, 153 and 180) have been selected. Emissions of different congeners of dioxins/furans are given in toxicity equivalents (TEQ) in comparison to the most toxic 2,3,7,8-tetrachloordibenzo-p-dioxine (2,3,7,8-TCDD) using the system proposed by the NATO Committee on the Challenges of Modern Society (NATO-CCMS) in 1988. PM_{10} are defined as particles with a diameter less than 10 μm that can follow the inhaled air into the respiratory system and the lungs

1990
Emission estimates for the base year 1990 have their origin in emission inventories carried out earlier by TNO (Visschedijk et al., 1998; Berdowski et al., 1997a,b). Emissions for selected HMs and POPs are based on a European emission inventory carried out within the framework of OSPARCOM, HELCOM and the UN-ECE (Berdowski et al., 1997b). This inventory was based on emission estimates produced by the countries themselves. However, if these data were not available default TNO-estimates were used.

Emissions for PM_{10} are based on a European inventory carried out within the framework of a Dutch research program on PM_{10} (Berdowski et al., 1997a). PM_{10} emissions were estimated using country statistics and a default set of emission factors. Due to information lacking, this study could only make a general distinction between emission factors for Western Europe and Central/Eastern Europe; no further distinction in country-specific emission factors was made.

In the study prepared by TNO, emission estimates for 1990 were partly revised by RIVM[4]. Total EU 1990 emissions for Cu, Cd, Pb, PAHs (and future trends) turned out to be dominated by one single sector in a particular country. Emissions for copper, cadmium and lead were dominated by the 'other transport' sector in Spain, and emissions for PAHs by combustion in 'the residential, commercial and other' sector in France. Since such figures seemed unlikely, it was decided to bring those high figures in line with much lower emissions reported for other countries. PM_{10} emissions for agriculture were revised on the basis of new knowledge on the emissions from livestock stables. Details are presented in Appendix D.

2010
The Baseline scenario (BL) has been reported by the European Environment Agency (EEA, 1999). It was based on the socio-economic and energy scenario described in this study (*see Technical Report on Socio-Economic Trends, Macro-Economic Impacts and Cost Interface* for details of these scenarios). Results of the work done by TNO and

[3] The critical load of a heavy metal equals the load causing a concentration in a compartment (soil, soil solution, groundwater, plant etc.) that does not exceed the critical limit set for that heavy metal.
[4] These revisions are not accounted for in the 1998 State of the Environment report for the EU (EEA, 1999)

methodological aspects have been reported in a separate background document for this study prepared by TNO (Visschedijk et al., 1998).

As mentioned earlier, emission projections prepared by TNO were revised by RIVM for the following substances: PM_{10}, Cu, Cd, Pb and PAHs. Details on these revisions are presented in Appendix D. In addition, RIVM calculated the spill-over effects of policy actions in the field of acidification. These were not reflected in the TNO-calculations.

Effects and costs of possible further emission control options for hazardous substances have been estimated by TNO on the basis of the results of earlier TNO studies performed for the Dutch Government in the framework of the UN/ECE HM and POP protocols (Berdowski et al., 1997c and 1998).

Spatial allocation
For the purpose of atmospheric transport modelling it is neccessary that results of the inventories and projections, calculated on a country level, are spatially distributed. Therefore, national emissions (calculated per detailed source category) have been allocated to point and/or area sources based on stored information in the TNO-databases (Visschedijk et al., 1998). Emissions from power generation and waste incineration have fully been treated as point sources. The major part of combustion and process emissions from industry (SNAP3 and SNAP4) have also been treated as point sources; remaining industrial emissions have been treated as diffuse sources and distributed according to population density. Emissions from residential, commercial and institutional combustion (SNAP2), solvent use (SNAP6), road transport (SNAP7) and other transport (SNAP8) have been treated as area sources and distributed according to population density. Agriculture related emissions (SNAP 10) have been treated as area sources and allocated according to the distribution of arable land.

Results have been prepared for air quality modelling as point and area source emission data per UN/ECE-source category (SNAP90 level 1). The resolution used for area sources was 1^0 x 0.5^0 longitude-latitude grid.

1.2.1.2 Uncertainties in emission estimates

The uncertainty in emissions varies per substance, source category and region or country in Europe. The available data for emissions of PM_{10}, HM and POP do not enable an in-depth uncertainty analysis. However, based on mainly TNO-work an effort was made to give a first order quantitative indication of uncertainties (Berdowski et al., 1997a and 1997b; Wesselink et al., 1998). Uncertainty factors presented apply in principle to the baseyear 1990 and the country level, and should only be considered as an indication of the degree of uncertainty. An uncertainty factor of 4 indicates in this study that there is a 95% chance of the real value deviating by no more than a factor of ¼ (25%) or 4 (400%) from the estimated value.

PM_{10}

Unknown PM_{10} sources
The bulk of the anthropogenic PM_{10} sources are expected to find inclusion in this study. However, some potential sources for which almost no information is available have not been considered. Such sources are resuspension of dust due to the motion of vehicles along the road, agricultural activities (blown-up dust from bare agricultural land areas, land preparation and harvesting), mining and quarrying, and finally, construction sites. Emissions of natural sources have not been estimated in this study. Important natural sources may be blown-up dust from non-cultivated land, sea-salt and biological particles such as pollen grains, fungal spores, bacteria and viruses (QUARG, 1996). Results from the Dutch research programme on particulates show that computed concentrations of PM_{10} (primary plus secondary) explain only 50 (rural) to 75% (industrial areas) of measured concentrations (Bloemen et al., 1998). The observed gap in concentration levels is assumed to be partly explained by unknown sources described above.

Known PM_{10} sources
Knowledge about the emissions of known sources is also limited. For known sources, uncertainties in emission estimates are large. National emission totals for PM_{10} have been calculated by TNO on the basis of a generalised set of default emission factors (one set for Western European countries and one for Central/Eastern European countries). Therefore, when interpreting uncertainty figures it should be realized that uncertainty factors given here not only reflect the uncertainty within a country but also the scatter in emission factors between countries (due to differences in techniques and control measures).

The largest uncertainties exist for emissions from 'industrial processes' and 'stationary fuel combustion in the energy sector and industry' (at a factor 4). Emission estimates for 'road transport' (exhaust emissions), 'waste incineration', 'residential, commercial and institutional combustion' and 'agriculture' (stables) are expected to be more accurate (at a factor 1.5 to 2).

- Unfortunately, *exhaust road transport emissions* (25% of the EU total in 1990) have been systematically overestimated in this study due to methodological reasons (at a factor 1.25 for 1990 to 1.03 for 2010) (see appendix *D* for details). Disregarding this methodological error which is typical for this study only, exhaust emissions to air from road transport are relatively accurate and well known (uncertainty at a factor 1.3).
- *Stationary combustion emissions for the energy sector and industry* (34% of the EU total in 1990) depend highly on the type of emission control applied. However, with a few exceptions no exact information about the type and degree of emission control in distinct EU-countries was available. Therefore default emission factors were used on the basis of information from the United States (EPA, 1995) and Germany (UBA, 1989). It is difficult to give quantitative indications of the representativeness of these emission factors for the 1990 behaviour of sources in individual EU countries. However, an indication may be given of the reliability of emission estimates. A tentative uncertainty at a factor 4 may be assumed. In addition, it should be noted that emissions probably have been overestimated for countries such as Germany and the Netherlands, both situated at the top end of the EU emission control spectrum (ERM, 1996).
- *Emissions from industrial processes* (18% of the EU-total in 1990) highly depend on the type and degree of emission control. Generalised emission factors have been used mainly on the basis of information from the Dutch emission inventory. Emissions are probably underestimated because the Dutch emission inventory (1997 database) underestimates the difuse emissions caused by ventilation of industry buildings. A tentative uncertainty at a factor 4 may be assumed.
- *Emissions from residential and institutional combustion* (18% of the EU-total in 1990) are largely uncontrolled. Uncertainty is therefore less than for other stationary combustion sources. A tentative uncertainty factor of 2 may be assumed. This uncertainty is largely associated with uncertainties in the amount of wood combusted in households in different countries as well as in applied emission factors.
- *Emissions from waste incineration* (4% of the EU-total in 1990) are based on a comprehensive study on dust emissions from Western European countries for the year 1990 (Rijpkema, 1993). An uncertainty factor of 1.5 may be assumed (Berdowski et al., 1997a).

Other pollutants

For heavy metals (HM), a tentative uncertainty at a factor 1.5 may be assumed for North Western European countries. For Central/Eastern European countries, the uncertainty in heavy metals emissions is expected to be larger (at a factor 3.5), and for Southern European countries a value in between is assumed to be applicable (Berdowski et al., 1997b).

The uncertainty in national emission estimates for PAHs within Europe is assumed to range from 2 to 5 depending on the country, mainly because of the uncertainty of domestic (wood) fuel consumption data, and of national activity statistics and composition data related to wood preservation.

For dioxins/furans, the uncertainty in emission estimates for European countries is assumed to range from a factor 5 to 20 dependent on the country (Berdowski et al., 1997b).

The uncertainty of emission figures for PCBs is large due to the fact that they are based on limited number of measurements. An uncertainty factor of 10 may be assumed although the fairly good agreement of calculated concentrations with measurements indicates that the uncertainty of estimates is probably smaller (Baart et al., 1995).

For pesticides, the uncertainty in national emission estimates is also quite large ranging from a factor 2 to 5, mainly explained by incomplete usage statistics, limited knowledge on application methods and the use of emission factors which are partly based on extrapolations from chemically related substances.

1.2.2 Atmospheric transport and deposition

1.2.2.1 Methodology for calculating transport and deposition

The general concept of the atmospheric transport models EUTREND and EUROS centres on the concentration of substances in air being calculated from its emissions and subsequently transported by the mean wind flow and dispersed by atmospheric turbulence. Meanwhile, the substance is removed from the atmosphere by dry and wet deposition and (photo-)chemical degradation.

EUTREND

Heavy metals, dioxins/furans, B(a)P and atrazine were calculated with the EUTREND model (Van Jaarsveld 1995), used in many studies on the deposition of contaminants over Europe and the seas forming its borders (Warmenhoven et al., 1989; Van den Hout et al., 1994; Van Jaarsveld et al., 1997). Recently, the EUTREND model has been used for the calculation of the depositions of heavy metals to the convention waters in the framework of OSPARCOM (Van Pul et al., 1998). The model also participated in the model intercomparison study carried out by EMEP/MSC-E (Sofiev et al., 1996).

In the model the dispersion and advection at a long range are described using trajectories assuming a well-mixed boundary layer, while local transport and dispersion is described with a Gaussian plume model. The latter 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 (temporary) transport of pollutants above the well-mixed boundary layer.

Transport and deposition of particles is calculated separately by the model for five different size-classes separately, each with specific deposition parameters. Particle growth is not incorporated in the model but is implicitly assumed to take place in the lowest size-class ($d < 10m$). The particle size distribution which has to be specified is the distribution of the particles as they are primarily emitted. As the larger particles tend to be removed faster than small particles, the actual size distribution is a function of transport distance and hence also of the effective deposition velocity. Size distributions used in this study, based on measured values in the Netherlands, are taken from Van Jaarsveld et al., (1986).

The deposition velocity is also a function of the roughness of the receptor area. The deposition velocity above grass, which is the dominant land cover, is taken for use in the EUTREND model domain. However, the deposition velocity above forests is considerably larger i.e. typically a factor of 2 to 3 (Ruijgrok et al., 1994). Therefore adjustments using a factor 2 and 3 of the deposition to forests were made to the original calculations of the deposition.

EUROS

The pesticides endosulfan, lindane and PCP were calculated with the EUROS model. EUROS is an Eulerian atmospheric transport model which describes the advection and dispersion of substances in the lower troposphere. This model has been used for acidification and ozone calculations (De Leeuw and Van Rheineck Leyssius, 1990; Van Loon, 1996) and has recently been extended to describe the deposition of persistent organic pollutants (POP) as well (Jacobs and Van Pul, 1996). Part of this model development is carried out in coöperation with MSC-E of the UN-ECE/EMEP framework.

Since many POP are semi-volatile at atmospheric conditions they may be re-emitted from the soil and water surfaces where they have been deposited. Due to the deposition and re-emission cycling of POP, the description of the deposition process generally used for components which only deposit, such as most acidifying components and heavy metals, cannot be applied. Instead, deposition should be considered as a net deposition, i.e. the sum of the deposition and re-emission fluxes. For this reason, a dynamic model which describes the gaseous exchange of POP at soil and sea surfaces (dry-deposition and re-emission) was coupled to the EUROS model (Jacobs and Van Pul, 1996).

1.2.2.2 Uncertainties in depositions

The uncertainties in modelling depositions of HMs and POPs are very large, particularly for POPs. The total uncertainty in the deposition calculations is caused by:

a) uncertainties introduced by the model concept,

b) uncertainties in substance-specific parameters,
c) uncertainties in emissions.

Ad a) This includes all the processes relevant in describing dispersion and deposition. Such general aspects can be tested using the model for substances such as SO_2, for which much more reliable data (emissions and measurements) are available. This type of uncertainty is expected to be relatively small (in the order of ± 30% for the deposition on a yearly basis). The range in the deposition data due to different meteorological conditions is also included in this figure.

Ad b) The choice of deposition parameters has a great impact on the calculated deposition. The dry and wet deposition rates of the HM are highly dependent on the particle size. The uncertainty of the particle size distribution of the emitted compounds could cause a range in the deposition of 30-50%.
For POP, the dry and wet deposition rates are also dependent on the physicochemical properties of the substance and properties of the receiving surface. The uncertainty in the yearly deposition data is estimated at a typical factor of 2.

Ad c) The uncertainties in the emissions are large as has been discussed in section 1.2.1.2.

A comparison between calculated and measured heavy metal concentrations in air (EMEP/CCC and OSPAR/CAMP networks) showed, in general, a good correlation (R^2 = 0.6-0.7) close to the 1:1 line. Practically all calculated depositions are within a factor of 2 of the measurements. However, for Cd and Pb two distinct regions in concentration levels are found: lower values in NW Europe (coastal stations) and higher values in Central Europe. For Cd, this means that the calculations underestimate the measurements by a factor of 2. This was also found in the study on deposition to the Convention Waters of OSPARCOM (Van Pul et al., 1998).

PAH and dioxins/furans do not form part of any monitoring programme in Europe. Therefore very limited measurement data (mostly urban and data for campaigns) are available to check the model results on a European scale. Van Jaarsveld and Schutter (1993) and Van Jaarsveld et al., (1997) have carried out a validation of their EUTREND model calculations for B(a)P and dioxins using soil data and some national monitoring data. They concluded that the agreement between modelled and calculated levels, in general, was fairly good. For B(a)P concentrations were found to be underpredicted in remote areas and overpredicted in industrial areas. Since the same model is used in this study, deviations from the above findings will be mainly due to a difference in emission data.
Of the pesticides only for lindane are a few measurements available from the EMEP/OSPAR network (Nordic countries and locations round the North Sea). This comparison between model calculations and methods showed calculations to approximately overestimate the measurements by a factor of 3. For endosulfan and PCP no data could be found. Since both substances have similar physicochemical properties to lindane, it is expected that the uncertainty in the results will be in the same order of magnitude as for lindane, i.e. where the uncertainty in the emissions is not taken into account. Previous calculations for atrazine showed the EUTREND modelled concentrations to be seemingly in reasonable agreement with measured data (within a factor of 2 of 3) in the Netherlands and Northern Germany (Baart et al., 1995).

1.2.3 Critical loads

1.2.3.1 Methodology for calculating critical loads

The critical loads for cadmium, lead and copper were calculated with both a steady-state and simple dynamic approach as described in detail by De Vries and Bakker (1998). Critical loads are computed on the basis of either:

- critical dissolved metal concentrations, since these criteria are good indicators of ecotoxicological effects. Using this steady-state approach implies that adsorption and complexation descriptions are not needed, and that critical load is mainly dependent on hydrological and vegetation data.
- a simple dynamic approach in which the accumulation up to a given critical metal content in the soil is accepted in a finite period (100 years) as preferred to an infinite period (steady-state approach). This approach, which implies the use of metal content present, may be regarded as one of the methods to derive target loads.

At the Bad Harzburg workshop it was recommended to calculate the target loads using the simple dynamic approach for a time frame of 50-100 years and the steady-state critical loads for infinity.

In this study all critical load maps were calculated with the simple dynamic approach using a time frame of 100 years.

1.2.3.2 Uncertainties in critical loads

Various sources of uncertainty in deriving critical loads exist, namely critical limits, calculation methods and input data. Uncertainties due to differences in critical limits can be very large. Uncertainties in calculation methods due to assumptions, such as equilibrium partitioning in a homogeneously mixed system, may give rise to a high uncertainty in certain situations. The uncertainty in data, either by spatial variability or because of lack of knowledge, can be quantified through an uncertainty analysis. Such an analysis, which also gives insight into which parameters are main determinants of the uncertainty in the resulting critical load, has been performed for cadmium and copper (Groenenberg 1999, in prep.). This uncertainty analysis was carried out for critical loads calculated with (i) a steady-state model, using background concentrations in the soil solid phase as critical limits and (ii) a simple dynamic model, using effect-based critical concentrations in the soil solution, including an acceptable net accumulation in the soil. Results showed that parameters describing the adsorption are generally of importance for both cadmium and copper, and both types of models and critical limits. Additionally, complexation plays a dominant role for copper, whereas hydrological parameters are important for cadmium, especially when using a dynamic model combined with a critical limit for the soil solution.

1.3 Results and analysis

1.3.1 Emission scenarios

1.3.1.1 Outline emission scenarios

The primary objectives of the emission calculations performed are:

- to assess future trends in emissions of primary PM_{10}[5] and selected HMs and POPs based on existing EU and ECE policies, and to compare these baseline trends with emission targets insofar as such targets have been set,
- to assess spill-over effects of policy action in the field of climate change (Kyoto targets) and acidification (commission proposal for a EU National Emission Ceilings Directive 1999c),
- to address further policy measures for emissions control of primary PM_{10} and selected HMs and POPs.

Three principal emission scenarios have been assessed in this study (see Box 1.2). Making use of a 1990-2010 timeframe, future trends in emissions under current legislation were assessed in the baseline scenario (BL), while the technology-driven scenario (TD) assumes full penetration of advanced end-of-pipe emission control technologies, such as high efficiency electrostatic precipitators, fabric filters and highly efficient wet scrubbers. The accelerated policy scenario (AP) takes into account the effects of accelerated policy action on climate change (Kyoto targets) and acidification (commission proposal for a EU NEC Directive, 1999c). Advanced end-of-pipe technologies to reduce emissions of PM10 and selected HMs and POPs are considered in the AP-scenario also[6]. Assessed control options are presented in detail in section 1.3.1.2.

Calculations have been performed at the country level. A description of projected baseline changes to basic socio-economic parameters such as population, GDP growth and energy consumption is presented in *Technical Report on Socio-Economic Trends, Macro-Economic Impacts and Cost Interface.* The scenarios BL and TD are based on the 'pre-Kyoto Business-as-Usual' energy scenario (BAU). The AP-scenario is based on the 'post-Kyoto no-trade' energy scenario. For this 'post-Kyoto no-trade' energy scenario it was assumed that the provisions of the Kyoto protocol are met assuming no-trade in GHG emissions. Details incl. costs concerning assumed control measures for the attainment of Kyoto protocol targets and NEC-targets are evaluated in the context of technical reports climate change and acidification respectively.

[5] Particles with diameter less than 10 μm (<0.01 mm) that can follow the inhaled air into the respiratory system and the lungs

[6] For the macro-economic feedback only costs involved with PM emission control have been taken into account.

The 'post-Kyoto no-trade' energy scenario differs substantially from the 'pre-kyoto BAU' scenario. For the EU total primary energy supply for the 'post-Kyoto no-trade' energy scenario is about 10% decreased when compared to BAU (6700 PJ); consumption of coal and oil (for energy purposes) falls by 46% (3100 PJ) and 16% (4200 PJ), respectively, and consumption of gas (for energy purposes) is more or less unchanged. The use of other fuels such as waste and biomass rises about 30% (700 PJ).

Under the AP-scenario, spill-over effects from policy action on climate change and acidification were analysed separately from the effects of full application of advanced abatement technologies for control of PM_{10} and selected HMs and POPs. Results of the spill-over calculations are presented as separate scenarios in this background report (see box). The Spill-Over Kyoto Protocol scenario (SO-KP) considers spill-over effects of Kyoto protocol targets only. The Spill-Over scenario (SO) considers effects of Kyoto Protocol targets as well as NEC emission targets for acidification as specified in the commission proposal (1999c).

Box 1.2.: Emission scenarios

The **Baseline scenario (BL)** is based on the Business-As-Usual socio-economic scenario presented in Technical background report 11, assuming the continued implementation of existing EU policies as of August 1997. All measures or policies agreed upon after that date are not included in the Baseline scenario. Furthermore, spill-over effects resulting from the continued post-1990 implementation of current policies in the field of acidification have not been considered in this scenario.

The **Technology Driven scenario (TD)** assesses maximum feasible emission reductions in the year 2010, assuming full application of advanced end-of-pipe emission control technologies for particulates, HMs and POPs, against the background of the BAU-scenario.

The **Spill-Over Kyoto Protocol scenario (SO-KP)** has been developed to assess the spill-over effects of policy action on climate change i.e. Kyoto Protocol targets, assuming no trade in GHG emissions. Spill-over effects have been assessed by repeating BL-calculations using a 'post-Kyoto no-trade' energy scenario, keeping all other socio-economic parameters unchanged.

In addition, a **Spill-Over scenario (SO)** has been developed to assess spill-over effects of policy action on climate change as well as on acidification. This scenario incorporates Kyoto Protocol targets as well as proposed EU

1.3.1.2 Control measures

Control measures included in the various scenarios will be overviewed here.

Hazardous chemicals are largely emitted in solid form adsorbed onto particles and thus may be effectively controlled by dust arresters such as electrostatic filters, fabric filters and scrubbers. This applies to most heavy metals. However, a substantial amount of mercury is emitted in the vapour phase, just like organic pollutants such as dioxins/furans and PAHs. To minimize the emissions of these partly gaseous pollutants, special techniques should be installed in addition to dust arresters.

Dust emissions may be reduced by improved operation of the combustion or production process, or by cleaning of the flue gas. Cleaning of exhaust gases by some type of dust arrester is common practice in coal combustion in power plants and industry, and also in many industrial production processes. Oil-fired installations in power plants and industry are, in general, not equipped with any dust cleaning device as is the case for firing devices in the 'residential, commercial and institutional' sector and motor engines. Dust emissions by municipal waste incineration are generally effectively controlled within the EU, although in some EU countries[7] significant amounts of municipal waste are incinerated without an any type of emission control (Berdowski et al., 1997a,d). However, the situation in these countries is improving thanks to current EU legislation.

It should be noted that measures to reduce emissions of acidifying compounds, such as desulfurization processes and low-S fuels, also have substantial side-effects on emissions of particulates and adsorbed chemicals.

[7] France, Italy, Spain, United Kingdom

Baseline scenario (BL)

BL: stationary sources

- Large Combustion Plants Directive (EC, 1988, 88/609/EEC); emission limits are 50 mg/Nm_3 for dust for new (post-1987) combustion plants > 500 MWth and 100 mg/Nm3 for new combustion plants between 50 and 100 MWth.
- The majority of utility and industrial combustion plants in the EU already complied with these dust emission limit values in 1990[8] (Berdowski et al., 1998). Thus, there is no legal obligation to reduce dust emissions further. Nevertheless, it may be expected that dust emissions from large combustion installations will decline substantially in the period 1990 to 2010 due to spill-over effects of SO_2-related control measures (SO_2 emission standards). Such spill-over effects are not included in the baseline scenario.
- Municipal Waste Incineration directives for existing installations (EC, 1989b, 89/429/EEC) and for new installations (EC, 1989a, 89/369/EEC): general emission limit values of 30 mg/Nm^3 for dust and 0.2 mg/Nm^3 for mercury have been applied. For dioxins/furans a limit value of 0.1 ng I-Teq/Nm^3 has been used in line with the Commission proposal for the amendment of current waste incineration directives[9] (EC, 1999a).
- EURO-CHLOR emission reduction programme for the chloro-alkali industry.
  The organisation of chlorine producers has developed a programme of voluntary measures to control mercury emissions which implies a specific emission of mercury of 1.5 g Hg/tonne chlorine-producing capacity by 2010 (Berdowski et al., 1998).
- Decreases in S-content of heavy fuel oil in oil refineries: a 40% reduction in the 1990 emission factor has been assumed (Visschedijk et al., 1998).

BL: mobile sources

- Compliance with EURO-3 particulates emission standards (phase-in from 2000) has been assumed for passenger cars, light-duty vehicle and heavy-duty vehicles. A lifetime of 10 years has been assumed for all vehicles.

Technology Driven scenario (TD)

TD: stationary sources

- High performance 4-field electrostatic precipitators have been assumed for coal and biomass combustion in the 'Public power, cogeneration and district heating' and 'Industrial combustion' sectors, with an assumed particle concentration in the flue gas of 20 mg/m^3 (Visschedijk et al., 1998).
- Electrostatic precipitators have been assumed for heavy fuel oil combustion in the 'Public power, cogeneration and district heating' and 'Industrial combustion' sectors.
- Combustion systems with optimised burning rates have been assumed for coal and biomass combustion in the 'Residential, Commercial and Institutional combustion' sector, with an assumed overall abatament efficiency of 25% (UN/ECE, 1998b; Hulskotte et al., 1999).
- For reducing dust emissions from industrial processes[10], many different control measures such as high performance electrostatic precipitators, fabric filters, and highly efficient wet scrubbers combined with waste gas collection systems have been assumed. In addition, specially designed techniques have been assumed for the control of gaseous emissions (Visschedijk et al., 1998; Berdowski et al., 1997c; 1998).
- For municipal waste incineration, emission standards have been assumed to be in line with the proposal for the amendment of current EU waste incineration directives: emission limit value of 10 mg/Nm^3 for dust [11] , 0.05 mg/Nm^3 for mercury and 0.1 ng I-Teq/Nm^3 for dioxins/furans (EC, 1999a).
- A full switch to PAH-free techniques for wood preservation has been assumed.

[8] With the exception of 'industrial combustion' and 'industrial processes' in southern Europe (Spain, Greece, Italy and Portugal) which in 1990 did not fully comply with requirements of the Large Combustion Plant Directive for dust.

[9] Strictly speaking, the emission limit value for dioxins/furans of 0.1 ng I-Teq/Nm^3 should not be accounted for in the Baseline because this standard had not yet been decided on by the EU per August 1997.

[10] Non-combustion related emission sources in ferrous, non-ferrous, cement, glass, chloro-alkali and the oil refining industry.

[11] TD-scenarios for waste incineration are by mistake based on a dust standard of 30 mg/Nm^3 (=BL-value) although the stricter TD-value of 10 mg/Nm^3 should have been used. However, effects on presented results are negligible.

TD: mobile sources

- Compliance with EURO-4 particulates emission standards (phase-in from 2005) has been assumed for passenger cars, light-duty and heavy-duty vehicles
- Particulates emission standards in compliance with EURO-2 for heavy duty vehicles have been applied for vehicles in the off-road transport sector.

Spill-over Kyoto Protocol scenario (SO-KP)

In addition to control measures listed for the BL scenario, the SO-KP scenario incorporates spill-over effects from policy action on climate change i.e.:

- Kyoto targets for reduction of GHG emissions, assuming no trade in emissions.

Spill-over scenario (SO)

In addition to spill-over effects from climate change, the SO-scenario incorporates spill-over effects from acidification (commission proposal for a National Emission Ceilings Directive, 1999c) i.e.:

- Limit sulfur content in heavy fuel oil to 1% (EC, 1999b).
- Spill-over effects are caused by the continued (post-1990) penetration of flue gas desulfurization techniques (FGD) on coal combustion in the 'Public power, cogeneration and district heating' and 'Industrial combustion' sectors. Spill-over effects have been calculated on the basis of IIASA estimates for the penetration of FGD technology in the separate EU countries in the years 1990 and 2010 (see Technical report on acidification); with particle concentration levels in the exhaust gas -without and with FGD - of resp. 50 and 20 mg/Nm^3 (EEA/EMEP, 1996).
- A realistic turnover rate of 10 years has been modelled for passenger cars, light-duty and heavy-duty vehicles. Thus, half the vehicle fleet is anticipated to satisfy EURO-4 emission standards (phase-in from 2005) in 2010; the other half is anticipated to comply with the less stringent EURO-3 standards (phase-in from 2000).
- Particulates emission standards complying with EURO-2 for heavy duty vehicles have been applied for vehicles in off-road transport sector.

Spill-over effects caused by by an EU limit of 0.1% for the sulphur content of gasoil for stationary sources (directive on sulfur in liquid fuels) is not reflected in the AP-NT scenario, because of a lack of information on the relationship between S-percentage for gasoil and particulates emissions. However, these effects, based on preliminary calculations using available relationships for heavy fuel oil, are expected to be rather small.

Accelerated Policy scenario (AP)

The AP scenario projects emissions, taking into account the effects of policy action on climate change and acidification. High performance technologies to reduce emissions are considered also (see list of control measures mentioned above for the TD-scenario).

1.3.2 Emissions and costs EU

This section on emissions and costs starts with a brief general analysis of the formation and emissions to air of primary PM_{10} and selected HMs and POPs. Next, main anthropogenic source sectors in 1990 are assessed. Finally, results for the calculated emission scenarios are discussed.

Emission tables for the base year 1990 and different scenarios are presented for the EU-region in Appendix A-1. For PM_{10}, emissions are also presented by country (see Appendix B-1). Figure 1.1. illustrates the differences between scenarios for the EU-region in terms of percentage emission reduction in 2010 compared to 1990. A further analysis of results is presented in figures 1.2. to 1.4.. These figures illustrate for the EU resp. the contribution of sectors to emission totals in the year 1990 and 2010, the importance of various source categories for overall emission trends in the period 1990 to 2010 and the sectoral emission changes in this period)

Origin of emissions

Heavy metals are present in trace amounts as a natural element of fossil fuels and biomass, and raw materials such as iron, zinc, copper and lead ores. Consequently, combustion and industrial processes are, in principle,

important potential emitters of heavy metals. Furthermore, metals are used in a variety of products such as pigments, batteries, fuel additives and fertilisers. These products end up in the waste stream and may ultimately be disposed of by incineration, giving rise to air emissions. Emission rates of heavy metals are determined by the content of trace elements in the fuels or wastes combusted or the raw materials processed, by the applied combustion or production technology, and by the efficiency of the emission control equipment (dust abatement and specific techniques for capturing gaseous heavy metals).

The heavy metal content (expressed in terms of energy content of the fuel) for lead, copper and mercury is on average one order of magnitude higher in coal than in oil. Cadmium content of coal and oil are more comparable (EEA/EMEP, 1996). Heavy metal content of natural gas is negligible.

Dioxins/furans are emitted from thermal processes involving organic material and chlorine-containing substances as a result of incomplete combustion or chemical reactions. Dioxins/furans are emitted for the larger part in the vapour phase, but also partly adsorbed onto dust particles. Approximately 80% of dioxins/furans is emitted in the gaseous phase at temperatures above 300 ^{o}C, while 90% is adsorbed onto dust particles at temperatues below 70%.

Emissions of PAHs are the result of incomplete combustion of fossil fuels, waste materials and other fuels such as biomass, and the use of PAH-containing coal-tar products for wood preservation.

1.3.2.1 EU: base year 1990

Source categories used for this study are consistent with level 1 of the UN/ECE-source category classification (SNAP90; EEA, 1995; EEA/EMEP, 1996). Only the treatment of refineries is different from SNAP90; i.e. emissions for refineries have been fully accounted for in SNAP-sector 4 'production processes'. The following source categories have been distinguished in this study (with corresponding SNAP90 codes in parentheses and deviations from SNAP90 in italics):

- Public power, cogeneration and district heating (01)
- Residential, Commercial and Institutional (RCO) combustion (02)
- Industrial combustion *(excl. combustion in petroleum industries)* (03)
- Production processes (*incl. combustion in petroleum industries)* (04)
- Extraction and distribution of fossil fuels (05)
- Solvent use (06)
- Road transport (07)
- Other mobile sources and machinery (08)
- Waste treatment and disposal (09)
- Agriculture (10)

The contribution of various UN/ECE-source categories to total EU emissions in 1990 is illustrated in figure 1.2. and will be briefly discussed below for the various compounds.

- **Particulate matter (<10 μm)**

PM_{10} emissions in EU countries are caused by different source categories, the most important sectors being combustion in the energy and industrial sectors (SNAP1/3 – 34% of EU-total), combustion in road transport (24%), combustion in households and services (SNAP2 – 18%) and industrial production processes (SNAP4 –18%).

Emissions from combustion in stationary sources is mainly due to coal (70%). Oil and other fuels (such as biomass) contribute 19% and 10%, respectively.

Emissions from industrial production processes are mainly due to the ferro and non-ferro industry (43%) and oil refineries (43%).

In interpreting the PM_{10} results of this study, it should be realised that available information on non-exhaust emissions by road transport is very scarce. The magnitude of non-exhaust PM_{10} emissions in this study forms only 5% of total road transport emissions (see Appendix D). Non-exhaust is specified here as tyre, brake and road wear. Some information is available for these sources, although very scarce. However, in this study, no account is taken of resuspension of dust due to the motion of vehicles along the road.

- **Cadmium**

Cadmium emissions in the EU are mainly (38%) due to industrial production processes (40%); other significant sources are combustion in the energy sector and industry (25%) and road transport (19%); minor sources are waste incineration (8%) and combustion in households and services (5%).

Emissions by combustion in stationary sources are caused by the combustion of oil (42%), coal (35%) and other fuels, such as biomass (23%). The relative high contribution of oil to total emissions compared to other heavy metals is caused by the relative high cadmium emission factors for oil combustion.

Emissions by industrial production processes are almost completely (95%) due to the ferrous and non-ferrous metal industry.

- **Copper**

About half the copper emissions come from mobile sources (48%) i.e. road transport (26%) and other transport (22%); other significant sources are combustion in the energy sector and industry (25%) and industrial production processes (26%).

Emissions from stationary combustion sources are mainly due to combustion of coal (68%). Oil and other fuels (such as biomass) contribute 17% and 15%, respectively.

Emissions from industrial production processes are almost entirely caused by the ferrous and non-ferrous metal production industry (98%).

- **Lead**

Lead emissions are largely (80%) caused by road transport i.e. lead in gasoline; industrial processes represent the other significant source (12%).

Emissions by industrial production processes are almost entirely caused by the ferrous and non-ferrous metal industry (90%).

- **Mercury**

Mercury emissions are mainly caused by industrial production processes in the non-ferrous metal (17%), cement (15%), chloro-alkali (11%) and iron and steel industries (3%). Other significant sources are combustion in the energy sector and industry (32%), and waste incineration (17%).

Emissions from stationary combustion sources are mainly due to combustion of coal (66%). Other fuels (such as biomass) and oil contribute 30% and 4%, respectively.

- **Dioxins/Furans**

Emissions of dioxins and furans (PCDD/Fs) are mainly caused by the waste incineration sector (41%). Other sources are combustion in the energy sector and industry (26%), industrial production processes (21%) and combustion in households and services (10%).

Emissions by stationary combustion sources are largely caused by the combustion of other fuels (such as biomass) (45%) and coal (38%). Emissions from oil are less important (17 %).

Emissions by industrial production processes arise mainly from the iron and steel industry (76%), non-ferro industry (12%) and other industrial processes (11%).

- **Polycyclic Aromatic Hydrocarbons (PAHs)**

Emissions of Polycyclic Aromatic Hydrocarbons (Borneff 6 PAHs) are due to solvent use (39%), combustion in households and services (26%), road transport (19%) and industrial production processes (12%).

Emissions due to solvent use are determined primarily by wood preservation with PAH-containing coal-tar products. Emissions may occur during the impregnation process as well as during storage, handling and use of the impregnated wood. The most widely used coal-tar products are carbolineum and creosote.

Emissions from stationary combustion sources are primarily caused by the combustion of other fuels such as wood (62%) and coal (37%). Emissions from oil are negligible (1%). Furthermore, emissions from stationary combustion

are mainly caused by combustion in households (84%). This is the result of using small firing installations in households that are less optimized than installations used in the energy sector and industry; this leads to more incomplete combustion and thus higher PAH emissions.

About half the emissions from industrial production processes are caused by non-ferrous metals production (52%), mainly in the aluminium industry. Other major industrial sources are the iron and steel industry (23%), primarily coke production, and asphalt road paving companies (26%).

- **Polychlorinatedbiphenyls (PCBs)**

Emissions of Polychlorinatedbiphenyls (PCBs) are mainly (about 80%) caused by the leaks and spills from closed electrical equipment such as transformers and capacitors. The other major source is re-emission from contaminated water and soil. The formation of PCBs in high temperature processes (stationary combustion and waste incineration) is relative unimportant in the baseyear 1990.

- **Pesticides**

Pesticides atrazine and endosulfan are mainly used in the agricultural sector. Pentachlorophenol and lindane are also used by industry for various applications.

1.3.2.2 EU: Baseline scenario for 2010 (BL)

Scope

The Baseline scenario (BL) assesses future trends in emissions under current legislation. The BL is based on the business-as-usual socio-economic scenario (see Technical Report on Socio-Economic Trends, Macro-Economic Impacts and Cost Interface); assuming the continued implementation of existing and proposed EU and ECE policies as of August 1997. All measures or policies agreed upon after that date are not accounted for in the Baseline.

Emission targets have been established for selected heavy metals (HM) and persistent organic pollutants (POP) in the framework of the United Nations Convention on Long Range Transboundary Air Pollution). According to the UN/ECE-protocols on HM and POP (UNECE, 1998a,b) countries are obliged to reduce atmospheric emissions of lead, cadmium, mercury, dioxins/furans and polycyclic aromatic hydrocarbons below the levels in a reference year, most probably 1990 for the EU.

The BL presented does not incorporate spill-over effects from the continued post-1990[12] penetration of control options in the field of acidification, such as flue gas desulfurization (FGD) techniques on coal boilers in the energy sector and industry (effecting emissions of PM_{10}, HM, POP) and the reduction of the sulfur content in heavy fuel oil to 1% (effecting PM_{10}). Furthermore, the BL only reflects current emission standards laid down in EU directives and ECE protocols[13]; more stringent *national* policies for emissions control of particulates, HMs and POPs are not accounted for. This methodology was decided on because detailed national data on a sector basis were not always available. The sectors 'power generation', 'industrial combustion', 'industrial processes' and 'residential, commercial and institutional combustion' are estimated to have already complied with current international standards in the base year 1990 (Berdowski et al., 1998). Consequently, no future improvement in the emission has been modelled for these categories under the BL. For other major sectors, i.e. 'road transport' and 'waste incineration', 1990 compliance with EU/ECE standards was estimated to be incomplete, making modelling of future reduction in emission factors under Baseline conditions necessary.

The chosen methodology of unchanged emission factors for stationary combustion sources and industrial processes should be considered to possibly lead to an overestimation of future emissions in specific countries with an advanced control strategy for particulates and hazardous substances. More stringent national, regional or local policies such as permitting requirements and national emission reduction agreements with sectors, may lead to reductions in emissions that go far beyond what may be expected based on EU/ECE requirements only.

Assessment and trends

Under current policies, substantial emission reductions are expected by 2010 (compared to 1990) for PM_{10} (-40%), lead (-60%), dioxins/furans (-31%), and to a lesser degree for mercury (-9%). In addition, emissions of PCBs, PCP

[12] Spill-over effects effectuated in the year 1990 are already accounted for in the 1990 emission inventory.

[13] Possible effects of the general EU-IPPC requirement to use BAT for major industrial activities have not been taken into account.

and lindane should be almost negligible in 2010 due to current EU regulations[14]. With such reductions, the EU is likely to meet emission stabilisation targets for these HMs and POPs, as established under the UN ECE CLRTAP Protocols. However, the achievement of emission stabilisation targets for cadmium (+5%), copper (+1%) and PAHs (+5%) is not ensured under baseline conditions[15].

The projected downward trend in PM_{10} emissions results primarily from lower transport emissions and stationary combustion emissions due to respectively stricter emission standards[16] and reduced coal use. The phasing out of leaded gasoline explains the substantial reduction in lead emissions expected in 2010. The marked improvement in emission levels of dioxins/furans is explained by the application of efficient flue-gas cleaning technologies in 2010. Reduced coal use and reduced emissions from the chloro-alkali industry, which has adopted an emission abatement programme for mercury emissions, are expected to bring about lower mercury emissions by 2010. Under the BL, small increases in the emissions of cadmium and PAHs are expected due to growth in road transport (for Cd) and higher use of wood fuel in households (for PAHs).

Below, emission changes under current legislation are analysed in more detail in terms of sectoral contributions, and for main sectoral trends in terms of underlying driving forces (i.e. socio-economic developments and control policies). The relevance of various sectors for overall EU-emission trends under baseline conditions is illustrated in Figure 1.3. The change in emissions per sector in the period 1990 to 2010 is presented in Figure 1.4. The contribution of various sectors to total EU-emissions is illustrated in Figure 1.2.

- **Particulate matter (<10 um)**

Emissions of PM_{10} are expected to decrease substantially by 2010 compared to 1990 (-40%). The downward trend is explained by lower emissions from *road transport* and *stationary combustion* sources (i.e. combustion in the energy sector and industry, and combustion in the 'residential, commercial and institutional' sector). Absolute emission reductions anticipated for both sectors are comparable.

Road transport emissions are anticipated to decrease by 70% by 2010 compared to 1990, in spite of the high increase in volume of passenger and freight transport (37% increase in road transport fuel use). This marked drop in emissions is a result of EU emissions standards for road vehicles being sharpened up.

For *stationary combustion* source*s* an emission reduction of 32% is expected; this is due to an anticipated sharp decrease in coal use (-40%). A downward trend in coal use is projected for all stationary combustion categories i.e. energy sector, industry, and households plus services. The use of natural gas (with almost negligible emissions) will increase, oil use will more-or-less stabilise. Although the contribution of other fuels (such as biomass with high emission factors for particulates and also HM and POPs) to total emissions is low, it should be considered that for the sector 'residential, commercial and institutional combustion' the effect of reduced coal use is partly compensated by increased biomass use.

- **Cadmium**

Cadmium emissions are anticipated to more or less stabilise in the period 1990 to 2010 (5%). The assessment of sectoral trends shows that the projected small increase in Cadmium emissions (5%) is explained by a large increase in emissions for *road transport* that is largely counterbalanced by a decline in emissions for *waste incineration*. Emissions from *road transport* are expected to increase by 50% in the 1990-2010 period due to the increase in the volume of transport. For *waste incineration*, a decline in the emissions of 83% is projected with the introduction of advanced flue-gas cleaning technologies needed for compliance with EU waste incineration directives.

It should be noted that emissions from road transport reported by TNO and used for this study are largely uncertain. According to the TNO-inventory, the contribution of road transport in total cadmium emissions greatly varies from country to country (from 3% for the UK to 65% for Denmark; 19% for the EU). Based on the UK-inventory, which is well-founded using a survey of cadmium content of various fuels, it can be doubted whether road transport really is a major source of cadmium. If UK-results would be extrapolated to all the other EU-

[14] Emissions of atrazine and endosulfan are expected to stabilise.

[15] Due to new insights, baseline results in this study were revised for PM_{10}, Cd, Hg, Cu, PAHs and therefore differ from (EEA, 1999, see Appendix D)

[16] New post-2005 EURO-4 emission standards for freight and passenger road transport have not been taken into account in the Baseline scenario; these standards were decided upon in 1998 and have been incorporated in the TD- and AP-scenarios.

countries road transport would no longer be a major source of cadmium (only 2%). Anticipated emission trends for cadmium emissions within the EU under Baseline-conditions would then change from an expected increase with 5% to a decrease with about 5%.

- **Copper**

Emissions of copper are projected to stabilize by +1% in the 1990 to 2010 period. The assessment of sectoral trends shows the anticipated large increase in copper emissions from road transport to be counterbalanced by a decline in the emissions from *combustion in the energy sector and industry.* Road transport emissions are expected to increase by 52% due to the increase in the volume of transport. Emissions from *combustion in the energy sector and industry* are expected to decrease by 31% due to an anticipated sharp decrease (-40%) in coal use.

- **Lead**

Emissions of lead will show a sharp decline by 2010 compared to 1990 (-60%) due the phasing out of leaded gasoline.

- **Mercury**

Emissions of mercury are expected to decrease slightly in the 1990 to 2010 period (-9%) due to anticipated emission reductions for industrial production processes and for combustion emissions in the energy sector and industry. Emissions for industrial production processes are anticipated to drop by 9% due to a decline in the emissions from the chloro-alkali industry, which has adopted an emission abatement program for mercury emissions. Emissions for combustion in the energy sector and industry will decline by 18% due to the anticipated decrease in the use of coal.

- **Dioxins/Furans**

Emissions of dioxins/furans are anticipated to show a marked decrease (31%) by 2010 compared to 1990 due to a large decline in emissions in the *waste incineration* sector. Emissions from *waste incineration* are expected to decline by 96% in the 1990-2010 period due to requirements prescribing an emission limit value of 0.1 ng TEQ dioxins/furans/Nm3 for municipal waste incineration. Advanced technologies for flue gas cleaning should be installed on waste incinerators comprising a combination of dedustment, wet scrubbing and the injection of active carbon in the flue gas in an entrained flow reactor followed by a fabric filter. The entrained flow reactor also serves to remove other gaseous substances such as PAHs and gaseous metals (mainly mercury).

- **Polycyclic Aromatic Hydrocarbons (PAHs)**

Emissions of PAHs are expected to increase slightly (+5%) in the 1990 to 2010 period. This anticipated small increase in emissions is caused by the anticipated large increase in the combustion of other fuels such as wood at 44%. The contribution of biomass residential combustion to total PAH emissions is anticipated to increase from 15% in 1990 to 28% in 2010.

- **Polychlorinatedbiphenyls (PCBs)**

Production and use of PolychlorinatedBiphenyls (PCBs) in EU countries was already tightly controlled in the base year, 1990. Remaining emissions in 1990 were mainly due to accidental releases from PCBs used in old closed electrical equipment. Current EU regulations on disposal of PCBs ensures that almost all PCBs still in use in old equipment will be de-contaminated by the year 2010. Therefore, PCB emissions will be almost negligible in 2010 compared to 1990. Remaining PCB emissions in 2010 will be mainly due to re-emission from contaminated waters and soils, and also due to formation of PCBs in high temperature stationary combustion processes.

- **Pesticides**

Overall pesticide use – measured by mass of active gradient - appears to have been decreasing in most EU countries over the past two decades (Thyssen, 1999; EEA, 1999). This trend may likely continue in the near future. However, consumption in terms of mass does not necessarily reflect the environmental burden, as more active and more specific substances are being developed and penetrate on the market (EEA, 1999).

Emissions of four pesticides (pentachlorophenol, lindane, atrazine and endosulfan) have been studied in more detail in this study.

The use of Pentachlorophenol (PCP) has already been strictly controlled by the EU. Unfortunately, emission trends for PCP shown in this study do not consider settled EU-directives imposing tight restrictions on the use of PCP. It is expected that due to these strict regulations emissions of PCP will be largely reduced in 2010.

Restrictions on the use of lindane imposed by the UN/ECE-POP-protocol have also not been taken into account in the Baseline. This was decided upon because the POP-protocol was settled in 1998 and therefore did not fit the definition of existing EU policies used for this study i.e. policies agreed upon as of August 1997. Nevertheless, it is expected that due to protocol requirements emissions in the EU will be almost negligible in 2010.

No change in emissions of atrazine and endosulfan is anticipated in the near future.

1.3.2.3 EU: Technology driven scenario for 2010 (TD)

The ***Technology Driven scenario (TD)*** has been developed to assess future emissions assuming full penetration of advanced end-of-pipe emission control technologies for particulates and hazardous chemicals, such as high-effciency electrostatic precipitators, fabric filters and highly efficient scrubbers. Results of this scenario for the year 2010 are compared with the Baseline, illustrating the magnitude of the remaining technical potential for further emission control. Control strategies causing changes in the structure and levels of energy consumption and economic activities have been excluded from this scenario. The relevance of various sectors for overall emission trends under TD conditions is illustrated in Figure 1.3.. The change in emissions in the period 1990 to 2010 per sector is presented in Figure 1.4. The contribution of various sectors to total EU-emissions is illustrated in Figure 1.2.

The full application of advanced control technologies would significantly reduce emissions of all pollutants studied in comparison to baseline results (see Figure 1.1: TD compared to BL). By 2010, 1990 emission levels of PM_{10}, lead and dioxins/furans would be reduced by about 70% (30-60% for BL); emissions of Hg and PAHs would be approximately 50% (10% for BL) while emissions for cadmium and copper would be reduced by 40% and 25%, respectively (0% for BL). These results clearly demonstrate the substantial technical potential remaining for the further control of emissions from hazardous substances.

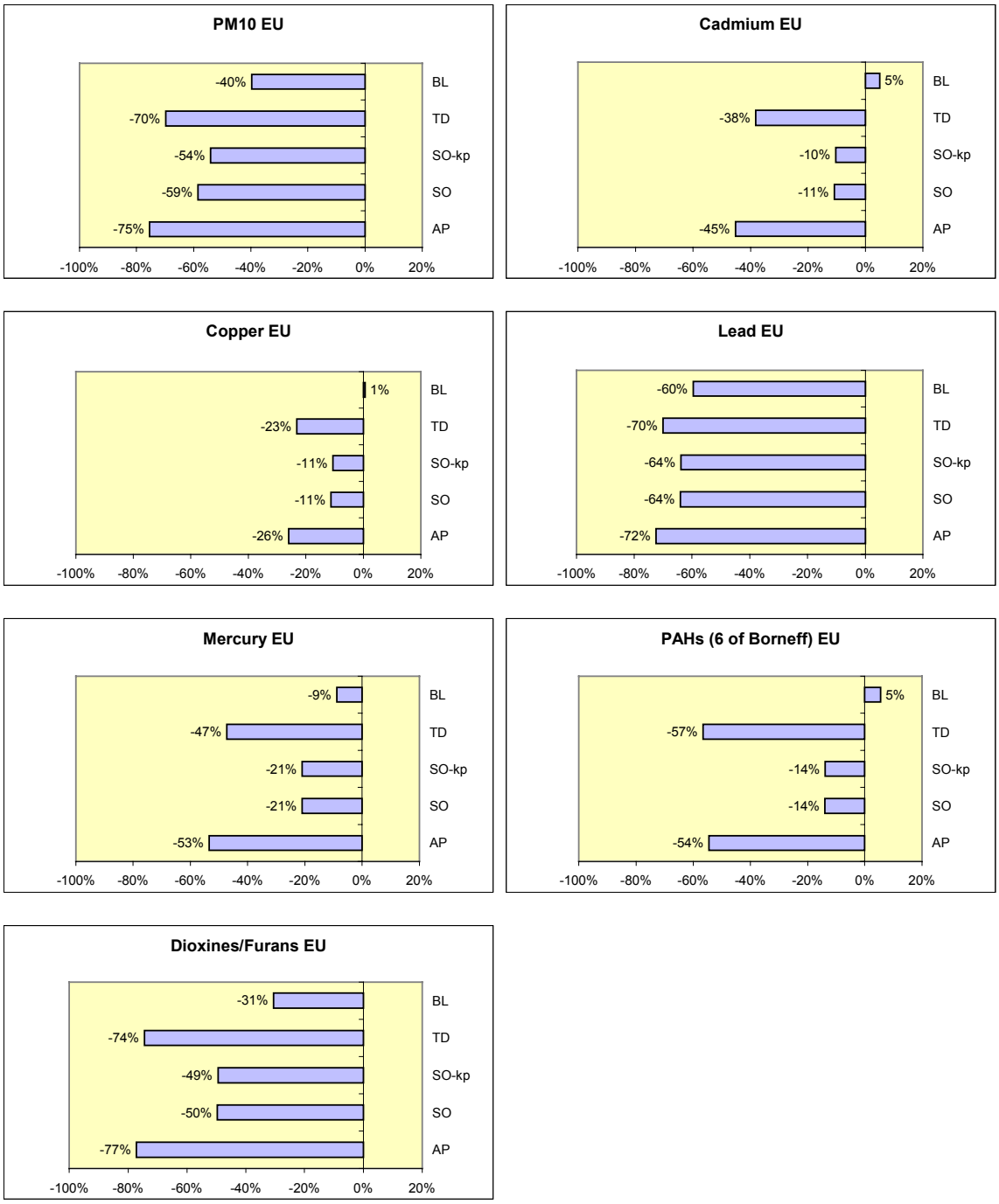

Figure 1.1 Emission changes (%) for the EU for PM_{10}, selected HMs and selected POPs in 2010 compared to 1990 under the Baseline (BL), Technology Driven (TD), Spill-Over Kyoto Protocol (SO-KP), Spill-Over (SO) and Accelerated Policy (AP) scenarios.

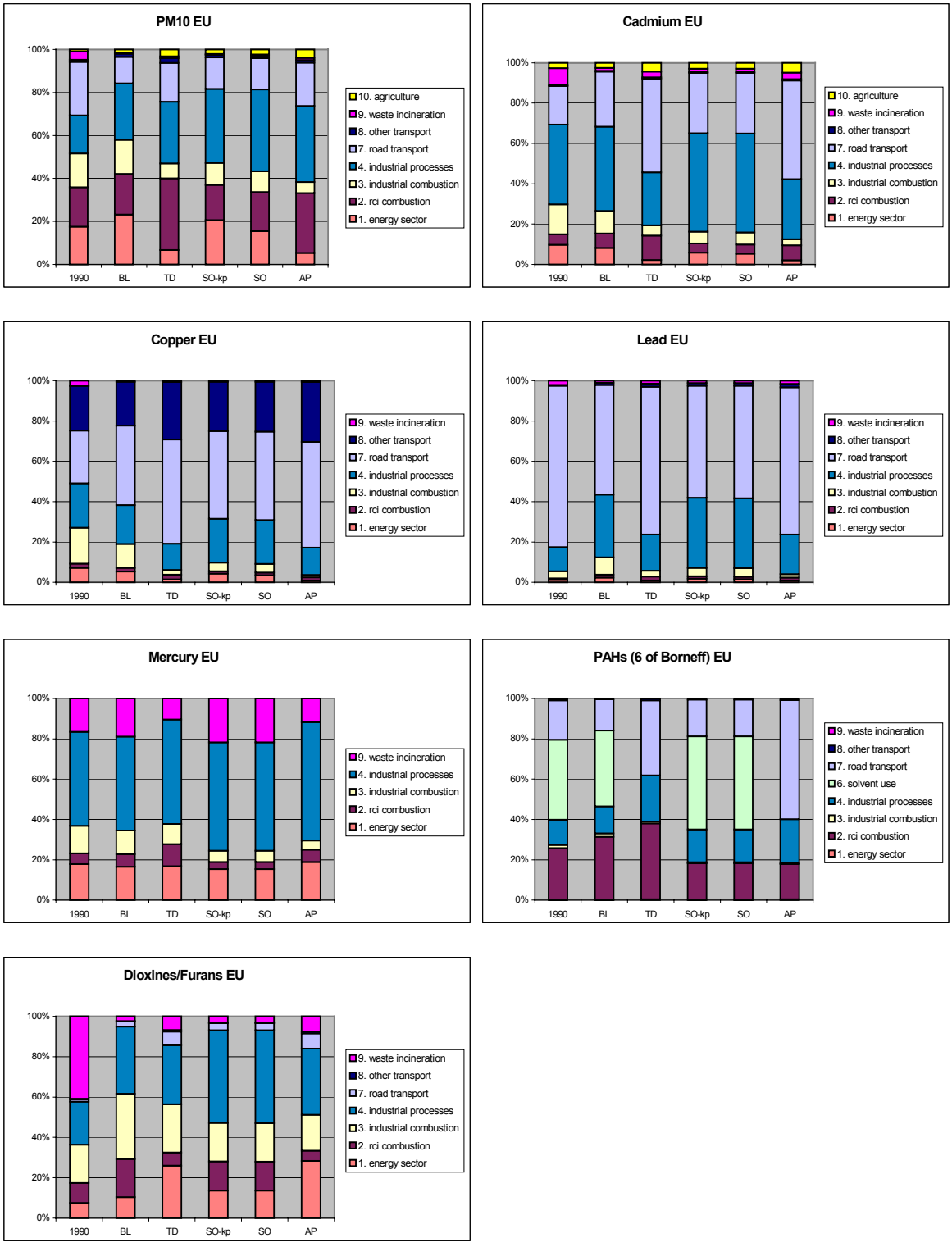

Figure 1.2. Contribution (%) by UN/ECE source category (SNAP90 level 1) to EU total emissions for the year 1990 and 2010 (%) under the Baseline (BL), Technology Driven (TD), Spill-Over Kyoto Protocol (SO-KP), Spill-Over (SO) and Accelerated Policy (AP) scenarios.

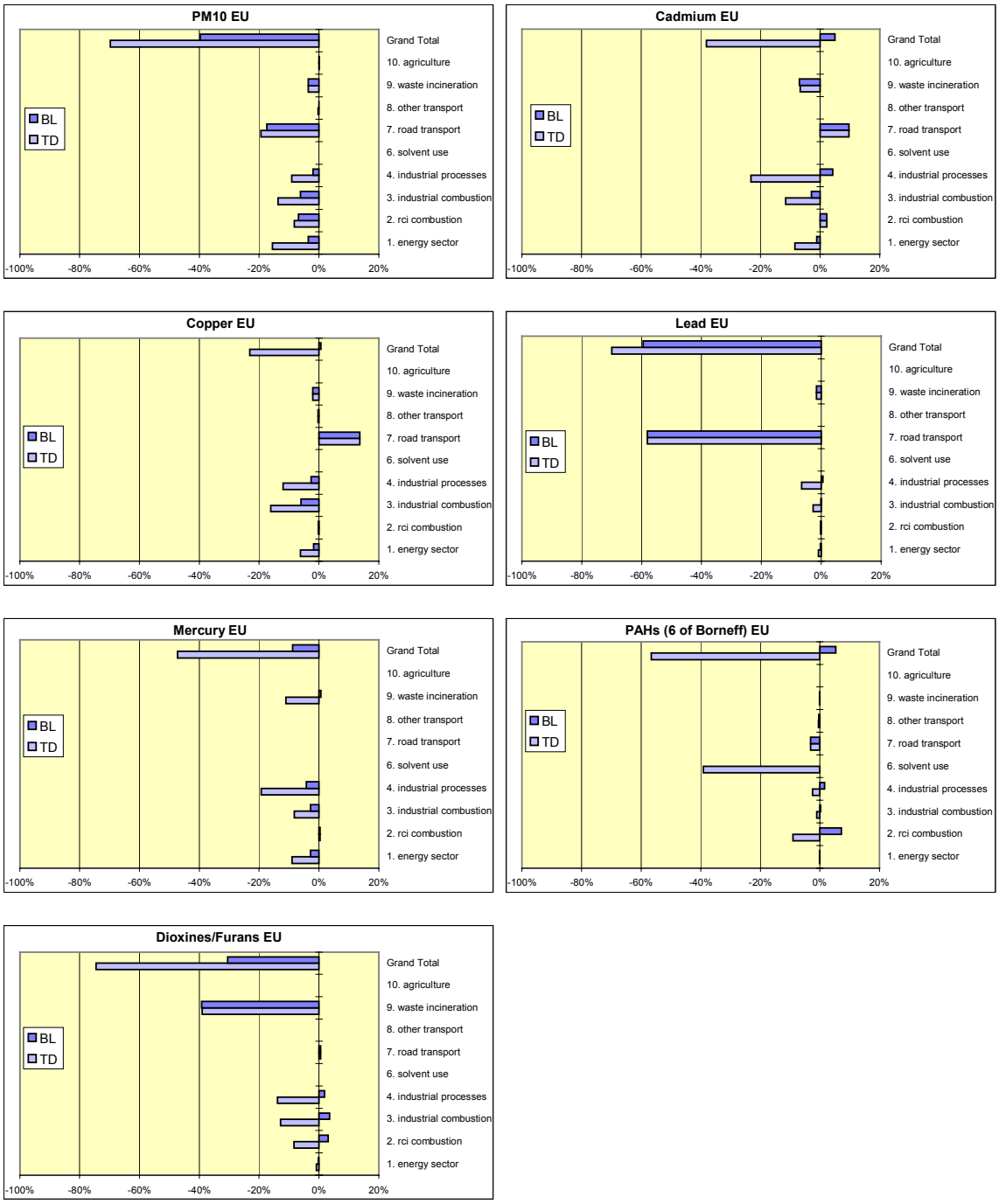

Figure 1.3. Emission changes by sector for the EU in the 1990 to 2010 period compared to the overall EU emission total in the reference year 1990 (%) under the Baseline (BL) and Technology Driven (TD) scenarios.

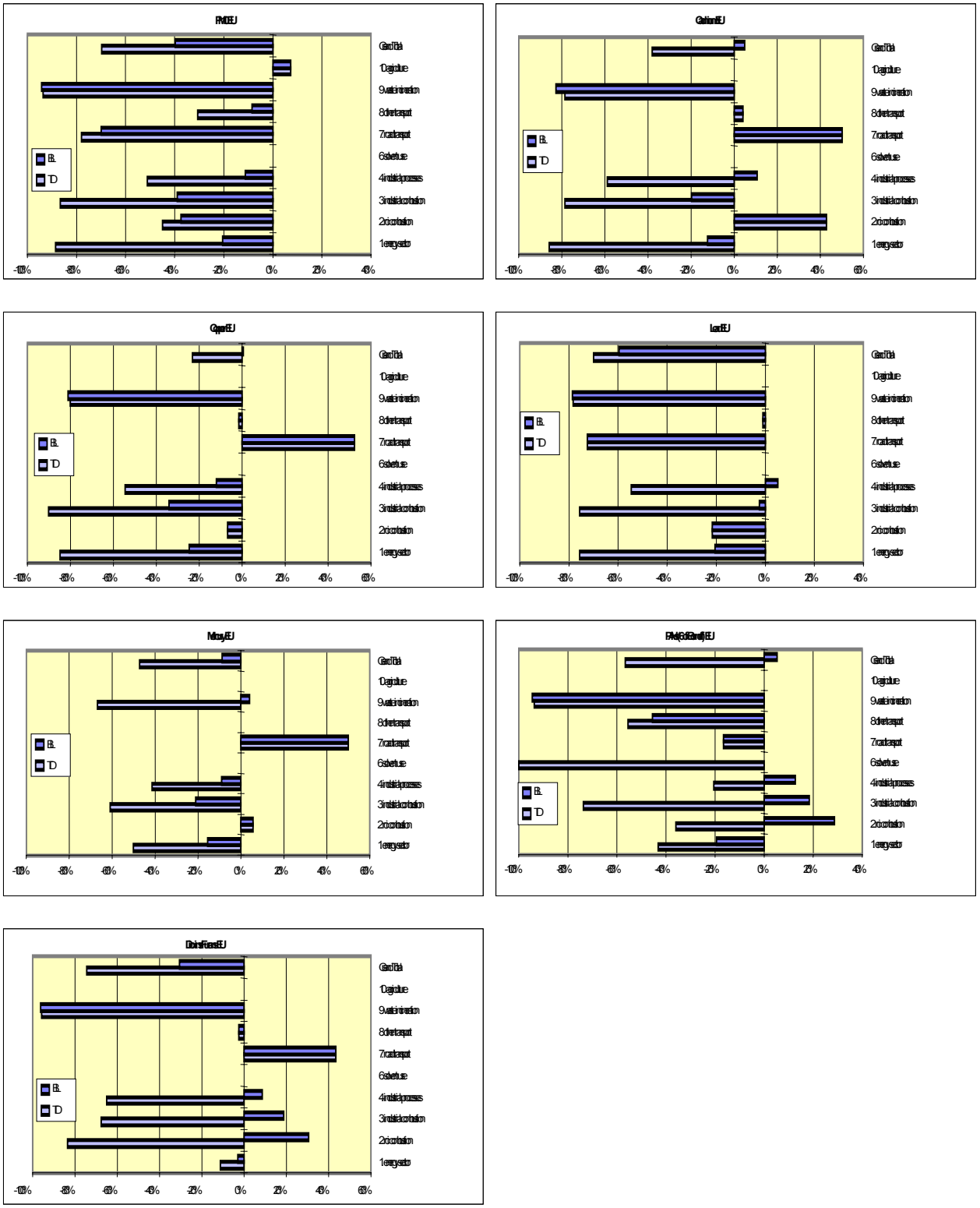

Figure 1.4. Emission changes by sector for the EU in the period 1990 tot 2010 (%) under the Baseline (BL) and Technology Driven (TD) scenarios.

1.3.2.4 EU: Spill-over scenarios for 2010 (SO-KP, SO)

Under the AP-scenario, spill-over effects from policy action on climate change and acidification were analysed separately from the effects of full application of advanced abatement technologies for control of PM_{10} and selected HMs and POPs. Results of such calculations are presented as separate 'spill-over' scenarios (see box). The **Spill-Over Kyoto Protocol (SO-KP)** scenario considers spill-over effects of Kyoto protocol targets only. The **Spill-Over (SO)** scenario considers effects of Kyoto Protocol targets as well as national emission targets for acidification proposed by the commission (1999c). In addition, the SO-scenario incorporates new EURO-4 emission standards for particulates for motor vehicles to be introduced from 2005, as adopted by the Council and the European Parliament in 1998.

Results for the SO-KP scenario demonstrate that emissions of all hazardous substances studied are significantly lowered due to spill-over effects from climate action, i.e. the expected large decline in the consumption of coal and oil[17]. Compared to 1990 emission levels, estimated spill-over emission reductions due to Kyoto targets would be significant (see Figure 1.1: SO-KP compared to BL), i.e. in the 10% to 20% range for fine particulates, cadmium, copper, mercury, PAHs and dioxins/furans, and only 5% for lead.

Results for the SO-scenario demonstrate that emissions of all substances studied are significantly lower due to spill-over effects from action on climate change and acidification (see Figure 1.1: SO compared to BL). For PM_{10}, estimated spill-over emission reductions are for about 75% due to climate change policies; the balance due to acidification measures incl. EURO-4 particulates emission standards for vehicles. For HMs and POPs, estimated spill-over effects are almost completely explained by climate action. With such reductions, emission stabilisation targets would be achieved for all studied substances, including cadmium, copper and PAHs, for which the achievement of targets was not ensured under baseline conditions.

Estimated spill-over emission reductions due to acidification measures for PM_{10} are for about half caused by the continued penetration of flue gas desulphurisation technologies; about one-quarter stems from the use of low sulfur heavy fuel oil and the balance from EURO-4 particulates emission standards for vehicles.

1.3.2.5 EU: Accelerated Policy scenario for 2010 (AP)

Assessment and trends

Under the AP-scenario, spill-over effects from policy action on climate change and acidification were analysed first. Results have been presented as separate spill-over scenarios in section 1.3.2.3. Results demonstrate that emissions of all studied hazardous substances are significantly lowered due to these spill-over effects (see table 1.1 and 1.2.). Costs of control measures for climate change and acidification are dealt with in the context of Technical Reports Climate Change and Acidification respectively.

The **Accelerated Policy (AP)** scenario shows the lower limit for future emissions, recognising spill-over effects of policy action on climate change and acidification, as well as the full application of advanced abatement technologies for PM_{10}, full control of gaseous PAHs and dioxins/furans in industry and the penetration of PAH-free wood preservation techniques. Such advanced control technologies for PM_{10}, HMs and POPs are included under the AP scenario because the assessment of air concentration levels (see section 2 on human health and air pollution) demonstrates that even with such far-reaching measures, stringent 2010 target-concentration levels for PM_{10} of 20 µg/m3 will still be exceeded in most countries. The contribution of various sectors to total EU-emissions is illustrated in Figure 1.2.

The SO-scenario (see section 1.3.2.3.) reflects spill-over effects of action in the field of climate change and acidification. Compared to this scenario, full application of advanced control technologies for control of PM_{10}, HMs and POPs is expected to significantly reduce emissions of all substances studied (see Figure 1.1: AP compared to SO). By 2010, emission levels for PM_{10}, Pb and dioxins/furans compared to those for 1990 would be reduced by about 75% (50-60% for the SO scenario); emissions for Hg and PAHs would be approximately cut in half (15-20% for SO), while emissions for Cd and Cu would be reduced by approximately 45% and 25%,

[17] The full trade variant was not estimated for chemical risk, but emission reductions can expected to be comparable to the no-trade variant because coal and oil consumption is similar in both variants

respectively (10% for SO). These results clearly demonstrate the substantial technical potential remaining for the further control of emissions from hazardous substances, also under post-Kyoto conditions.

Measures and costs

Emission control technologies for PM_{10} and selected HMs and POPs evaluated in the AP scenario may be distinguished into devices for catching dust in the off-gas and special designed methods for capturing chemicals emitted in the gaseous phase (Visschedijk et al., 1998; Berdowski et al., 1997c,1998).

Technical measures to reduce emissions of PM_{10} and adsorbed HMs and POPs from stationary combustion sources evaluated in the AP scenario comprise high-efficient 4-field electrostatic precipitators (ESPs) for coal and biomass combustion in power plants and industry, ESPs for oil combustion in power plants and industry, and improved stoves for coal and biomass combustion in households. For industrial processes advanced dust cleaning methods, such as fabric filters, high performance electrostatic precipitators, and high efficient wet scrubbers have been assumed. (Visschedijk et al., 1998; Rentz et at, 1996). For mobile sources, 100 percent implementation of EURO-4 emission standards has been assumed under the AP scenario.

Techniques evaluated to reduce gaseous emissions of PAHs in the metallurgical industry have included 'waste gas collection systems and vertical contact bolds' for the Soederberg-process in the aluminium industry, and 'highly sealing oven doors, suction hoods and electrostatic tar filters' for the production of cokes . Furthermore, a 100% shift to PAH-free techniques for preservation of wood is taken into account.

For the control of gaseous emissions of dioxins/furans in the ferrous and non-ferrous industry, techniques such as 'high efficiency wet scrubbers' have been studied (Berdowski et al., 1997c).

Total PM_{10} reductions for the AP scenario compared to the Baseline and costs for advanced dust abatement technologies are summarised in Table 1.1. It should be noted that uncertainties in emission reductions and cost estimates are high: a factor of 2 to 4. Side-effects on particle-bound HMs are also presented. Costs of measures to reduce PM_{10} emissions from transport (i.e. EURO-4 standards) are evaluated in the context of acidification (see Technical Report on Acidification). No cost estimates have been produced for the optimisation of stoves in households. Control of PM_{10} will also have substantial side-effects on particle-bound emissions of PAHs and dioxins/furans. However, for PAHs and dioxins/furans no information was available to distinguish such effects from the emission reductions due to the control of gaseous emissions. Therefore side-effects of PM_{10} measures on PAHs and dioxins/furans are not included separately in Table 1.1.

Total reductions in PAHs and dioxins/furans for the AP scenario compared to the Baseline (side effects from dust control measures and effects from specially designed techniques for reducing gaseous emissions) and costs for control of gaseous emissions of these substances are summarised in Table 1.2. No cost estimates have been produced for the shift to PAH-free wood-preservation techniques.

The largest reduction of PM_{10} emissions can be achieved through further control of the emissions from major industrial production processes for which the existing Integrated Pollution Prevention and Control (IPPC) Directive contains only a general requirement to use the Best Available. An appropriate action could be to extend the IPPC-directive with binding emission standards for major industrial production installations as already existent for other major source categories such as large new (post-1987) combustion plants (>50 MWth) in the energy sector and industry, waste incineration plants and mobile sources. The PM_{10} emission reduction that could be achieved by upgrading of industrial production sites with the most advanced control technology amounts to about 7% of 1990 emissions. Such advanced measures for PM_{10} control will also reduce emissions of heavy metals, dioxins/furans and PAHs. Compared to 1990 emission levels, substantial emission reductions are expected for those substances ranging from about 7% for lead to 28% for cadmium.

Looking to cost effectiveness, further reduction of PM_{10} emissions may also be achieved in the sectors on 'Public power, cogeneration, and district heating' and 'Industrial combustion'. About 85% of the emission control potential for these two sectors mentioned in Table 1.1 could be obtained by the replacement of dust arresters on coal and biomass-fired combustion installations with high performance arresters. The remaining 15% could be obtained by the installation of dust arresters on oil-fired installations, which currently do not have such technologies installed. Such measures will also reduce emissions of HMs and POPs.
Compared to 1990 emission levels, emission reductions are anticipated for HMs ranging from 1% for lead to 7% for cadmium.

Results for the AP-scenario presented in this study suggest that emission reductions for PM_{10} which may result from the installation of optimised burning systems in the residential, commercial and institutional sector, are rather low. However, it should be noted that presented emission reductions only relate to coal and biomass/wood combustion. A further reduction of emissions from oil combustion has not been considered. Furthermore, uncertainties in emissions from households are rather high due to large uncertainties in the quantity of biomass/wood burned in residential stoves and in related emission factors. It should also be realised such measures are important for reducing emissions at the local city level.

Finally, it should be considered that the use of available alternative techniques for the preservation of wood, which are not based on impregnation with PAH-based product, could reduce PAHs emission substantially.

Total costs for further emission control in the context of the Accelerated Policy scenario are about 14% lower than total costs in the TD scenario; this is due to the switch to less polluting fuels in the Accelerated Policy scenario (coal to gas).

Table 1.1. Emission reductions for PM_{10} for the EU in the AP scenario compared to the Baseline, associated costs, and side-effects of advanced PM_{10} control technologies on emissions of Heavy Metals.

Sector (SNAP90)	PM_{10}	PM_{10}	PM_{10}	PM_{10}	Cd	Cu	Pb	Hg
	Reductions ktonne/yr		Direct costs M€/yr	M€/ktonne	Side effects PM-control[f] Tonne/yr			
	SO[d]	CT[e]	CT	CT				
Public power, cogeneration and district heating	197	134	270	2.0	7	36	55	8
Residential, commercial and institutional combustion	103	17	n.e.	n.e.	0	0	0	0
Industrial combustion	147	73	190	2.5	7	42	171	6
Industrial processes	0	186	1050	5.5	54	136	1165	37
Transport	48[a]	32[c]	n.e.[b]	n.e.[b]	0	0	0	0
TOTAL	496	443	1510	3.4	68	214	1392	51

n.e.: not estimated

The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in calculated figures. See section.1.2.1.2. of this technical report for a discussion on uncertainty in emission estimates for the base year. See Technical report on Climate Change and on Acidificaton for costs of control measures for climate change and acidification respectively.

[a] Reduction includes effect of 50% penetration of EURO-4 standards compared to 100% penetration of EURO-3 assumed under the Baseline scenario.

[b] Costs of EURO-4 emission standards are part of acidification (*see Technical report on Acidification*)

[c] Reduction refers to 100% penetration of EURO-4 standrds compared to 50% penetration of EURO-4 assumed under spill-over.

[d] Spill-over (SO) reductions due to policy action on climate change (Kyoto targets no trade) and acidification (NEC-directive) (=SO minus BL).

[e] Control Technologies (CT) reductions due to advanced dust control technologies (=AP minus SO)

[f] Side-effects of advanced PM_{10} control technologies on emissions of heavy metals (= AP minus SO)

Table 1.2. Emission reductions for dioxins/furans and PAHs for the EU in the AP scenario compared to the Baseline and related costs

Sector (SNAP90)	Dioxins Furans	Dioxins Furans	PAHs	PAHs	PM_{10}	Dioxins furans and PAHs
	Reductions				Direct costs	
	SO[d] g I-Teq/yr	CT[c] g I-Teq/yr	SO tonne/yr	CT tonne/yr	CT M€/yr	CT M€/yr
Public power, cogeneration and district heating	24	23	2	3	270	0
Residential, commercial and institutional combustion	349	368	964	420	n.e.	n.e.
Industrial combustion	774	335	78	11	190	0
Industrial processes	0	948	0	231	1050	90[a]
Solvent use	0	0	0	2211	0	n.e.
Transport	0	0	30	0	n.e.[b]	0
TOTAL	1147	1673	1076	2875	1510	90

n.e.: not estimated

The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in calculated figures. See section 1.2.1.2. of this technical report for a discussion on uncertainty in emission estimates for the base year. See Technical report on Climate Change and on Acidification for costs of control measures for climate change and acidification respectively.

[a] Costs of specially designed techniques for capturing gaseous emissions of dioxins/furans and PAHs in metallurgical processes

[b] Costs of EURO-4 emission standards are part of acidification (*see Technical report on Acidification*).

[c] Control Technologies (CT) reductions due to 1) advanced dust control technologies with side-effects on particle-bound PAHs and dioxins/furans, and 2) techniques designed specifically for the control of remaining gaseous emissions.

[d] Spill-over (SO) reductions due to policy action on climate change (Kyoto targets no trade) and acidification (NEC-directive) (=SO compared to BL)

1.3.3 Emissions and costs in accession countries

In general, emissions from stationary combustion sources are more important in accession countries than in the EU15, and emissions from transport and waste incineration are less important. Overall results for the ten accession countries together are presented in Appendix A-2 for the base year 1990 and for the BL and the TD scenarios. For PM_{10}, emission tables are also presented by country (see Appendix B-2). Other scenarios (SO-KP, SO and AP) have been developed for the EU only.

Under current policies, substantial emission reductions are expected by 2010 with respect to 1990 for all substances studied except cadmium, for which a small increase in emissions is anticipated (see Table 1.3). Emission reductions for PM_{10}, lead, and mercury will be similar to reductions expected for the EU; emissions of copper and PAHs will likely decrease more and emissions of dioxins/furans will decrease less.

It should be noted that the Baseline scenario considers only emission control requirements of the second UN/ECE sulfur protocol only (UNECE, 1994a); requirements of the UN-ECE protocols on HMs and POPs and the Gothenburg protocol have not been accounted for in the Baseline. This was done so because these protocols were settled in 1998 and thus do not fit with the definition of existing EU policies used throughout this study i.e. existing policies in place as of August 1997.

Table 1.3. Emission trends for primary PM_{10}, selected HMs and selected POPs for accession countries under the Baseline (BL) and Technology Driven (TD) scenarios in % change in 2010 compared to 1990

Substance	1990	BL	TD
	tonne	%	%
Primary PM_{10}	2×10^6	-41	-79
Lead	4×10^3	-58	-71
Copper	1×10^3	-30	-58
Mercury	80	-12	-49
Cadmium	160	+4	-19
Dioxins/Furans (I-Teq)	2.5×10^{-3}	-9	-44
PAHs	1.5×10^3	-16	-40

The projected downward trend in emissions for PM_{10}, copper and mercury results primarily from lower combustion emissions due to reduced coal use and the implementation of the second UN/ECE Sulphur Protocol (PM_{10}, copper, mercury). The European wide elimination of the use of leaded gasoline explains the large reductions in lead emissions. For cadmium, positive effects of reduced coal use and S-protocol requirements on emissions will be undone by the increase in the use of fuel oils in households. As a result, a small increase in the emissions of cadmium is anticipated. The anticipated decline in emissions of PAHs and dioxins/furans is explained primarily by reduced coal use in households.

No specific assessment has been made in this study of the consequences of enlargement on emissions, and the associated costs. Enlargement could increase energy consumption and transport in accession countries, leading to higher emissions of hazardous substances. On the other hand, enlargement implies that accession countries have to comply with emission standards specified in EU directives, of which the large combustion plant directive, the vehicles directive, the waste incineration and fuel directives are the most important. Implementation will put emission reductions somewhere between the results for the Baseline and the Technology Driven scenarios (see Table 1.3).

It is also important to note that current emission requirements of the UN/ECE protocols on HMs and POPs are as stringent as requirements of EU regulations, except for transport. Studies of the impact and costs of these HM and POP protocols (Berdowski et al., 1997c,1998) indicate the total annual costs for upgrading industrial installations and waste incineration plants to comply to emission standards for particulates and HM at about € 100 million per year (capital investment: € 1 billion). Similar cost estimates have been reported for smaller coal-fired combustion installations in the energy sector and industry[18]. In addition, costs for upgrading waste incineration plants to emission limit values for dioxins/furans are estimated at € 10 million per year (capital investment: € 100 million). Such figures are low compared to costs of acidification measures and costs of measures to reduce emissions from transport; both of these are evaluated in the context of acidification (see Technical Report on Acidification).

Summarising, it may be concluded that due to reduced coal use and emission control requirements of the UN/ECE-protocols (second sulfur protocol, protocols on HMs and POPS, Gothenburg protocol), large emission reductions for PM_{10}, HMs and POPs can be expected for accession countries. For the near future, a major improvement in emission control technology can be expected in accession countries for all relevant major source sectors. The reduced emission of hazardous substances in the accession countries will contribute to a significant improvement in the environmental situation in these countries, but also in the EU15. Especially the particulate matter reductions are important for attaining EU policy objectives.

1.3.4 Emissions other countries

BL- and TD-scenarios have also been calculated for non-accession countries. Results can be found in *Appendix A-3 and B-3*. Results will not be further discussed here.

[18] It has been estimated that compliance with S-protocol requirements will also lead to compliance with HM-protocol requirements for about 80% of the coal-fired and 100% of the heavy fuel oil-fired power-generating capacity in the energy sector and industry (TNO, 1998). Costs of such S-protocol measures have not been included here.

1.3.5 Depositions and critical loads

The deposition of heavy metals and POPs were calculated with atmospheric transport models using the above emissions for 1990 and 2010 Baseline scenario.

The transport from substances emitted to the atmosphere and the deposition to the European region was calculated using RIVM atmospheric transport models (EUTREND/EUROS). The following substances were considered: the heavy metals (HM) cadmium (Cd), copper (Cu) and lead (Pb) and the persistent organic pollutants (POPs) dioxines/furans, atrazine, endosulfan, lindane, pentachloorphenol (PCP). The uncertainty in the modelled depositions of these substances is quite large. For dioxins/furans and pesticides, in particular, the measurement data available are very limited for checking the model results on a European scale. Therefore these model calculations should be considered as indicative of the yearly deposition, with a typical first-guess accuracy, which is as large as the uncertainty in the emissions.

Critical loads were calculated only for the heavy metals. For POP the critical load approach is still under discussion. This means that the impact of the deposition of POP could not be indicated. The deposition as such was used in evaluating the environmental impact of POP deposition.

BL depositions

All results shown and discussed here are for 1990 and 2010 Baseline scenario. It should be noted that original preliminary emission data prepared by TNO (Visschedijk et al., 1998) have been used for the atmospheric transport modelling. For the EU, such data are partly inconsistent with the final national emission data presented in section 1.1.2.1.

Heavy metals
The deposition of cadmium and lead for 1990 and 2010 are shown in Appendix C (Figures C.1 a,b,c,d). For cadmium the deposition in the EU is expected to more-or-less stabilize in the 1990 - 2010 period due to stabilising emissions in the EU and accession countries (see *Appendices A and B*). No attention should be given to the moderate increase in Cd depositions for EU countries, illustrated in the maps shown in *Appendix C*. These maps do not account for the expected decreaese in Cd emissions from waste incineration. For lead, large decreases in deposition levels are anticipated for the EU due to the decrease in transport emissions. High depositions are found in areas with a large traffic intensity, although the deposition in industrial areas becomes more important in the future. Despite stabilizing EU emissions, EU depositions for copper will decreaese because of the anticipated emission reductions in the rest of Europe.

Persistent Organic Pollutants
The deposition calculations for selected persistent organic substances are very uncertain, as discussed above. Therefore the results should be considered as indicative.

The deposition of dioxin/furans in 1990 and 2010 is depicted in Appendix C (Figure C.1 e,f). For dioxins/furans, high deposition areas are largely associated with large point sources. The deposition in the EU is estimated to decrease by about 20% due to the control of emissions from waste incinerators.

For the pesticides atrazine, endosulfan, lindane and PCP only 1990 deposition maps were calculated. The depositions in the non-agricultural area for lindane and PCP are depicted in Appendix C (Figure C.2 a,b). The deposition of pesticides is highest near the areas of application. However, large amounts of pesticides are also transported out of the application areas and deposited in countries where these pesticides are not used at all. The use of lindane is severely restricted under the UN/ECE POP protocol.

BL exceedances of critical loads for forest soils

For the assessment of the impact of the deposition of heavy metals on ecosystems the critical load method is used. The critical load is defined as*: a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge* (Nilsson and Grenfelt, 1988). Critical loads for cadmium, lead and copper were calculated for European forest soils according to De Vries and Bakker (1998). At the UN/ECE-Working group of the Effects Workshop in Bad Harzburg in 1997, this method was discussed and accepted as the method with which

critical limits and loads for heavy metals in terrestial and aquatic ecosystems can be calculated. According to the above definition the critical load of a heavy metal equals the load causing a concentration in a compartment (soil, soil solution, groundwater, plant etc.) that does not exceed the critical limit set for that heavy metal. Since this is a newly accepted method, it has not been used frequently yet and no thorough uncertainty analyses and validations have taken place. Therefore, the critical load and exceedence maps should be considered as preliminary maps; they are shown here to demonstrate the potential of the method.

For cadmium and lead the critical loads for forest soils (95-percentile of the forested area) over Europe are depicted in Appendix C (Figures C.3. a,b,c,d). All maps are shown for the organic (top) layer, generally the most sensitive layer in terms of effects. The deposition was corrected for the higher deposition rate for forested areas, i.e. the calculated deposition multiplied by a factor 2. The exceedances were calculated by subtracting the critical loads from the deposition loads.

For cadmium, small areas of Poland are found where the critical load was exceeded in 1990[19]. In the EU and the rest of Europe almost no exceedances were observed. Exceedances in Poland are expected to increase in 2010 due to the projected high increase in emissions from residential combustion
For lead, large areas in the EU were found where the critical load was exceeded in 1990. Such exceedances are also observed in the rest of Europe, although to a lesser degree than in the EU. Exceedances will decrease significantly in 2010 throughout Europe due to the phase-out of leaded gasoline for transport. However, critical loads will still be exceeded in 2010 in large parts of the EU.

No exceedances of critical loads have been found for copper.

[19] High depositions for Poland are explained by the relatively high emissions in this country compared to other comparable countries. No special analysis has been made in this study on the relaibility of the Poland estimates

1.4 Conclusions

Table 1.4. Emission trends for primary PM_{10}, selected HMs and selected POPs for the EU under the Baseline (BL), Technology Driven (TD), Spill-Over Kyoto Protocol (SO-KP, climate policies only), Spill Over (SO, climate plus acidification policies), Accelerated Policy (AP) scenarios in % change in 2010 compared to 1990.

Substance		1990	2010				
			BL	TD	SO-kp	SO	AP
			%	%	%	%	%
PM_{10}	*Mtonne*	2.6	-40	-70	-54	-59	-75
Cadmium	*Ktonne*	0.2	5	-38	-10	-11	-45
Copper	*Ktonne*	1.5	1	-23	-11	-11	-26
Lead	*Ktonne*	16.4	-60	-70	-64	-64	-72
Mercury	*Ktonne*	0.2	-9	-47	-21	-21	-53
PAH's (Borneff 6)	*Ktonne*	5.6	5	-57	-14	-14	-54
Dioxins/furans	*kg I-Teq*	6.0	-31	-74	-49	-50	-77

The following main conclusions are drawn from the environmental assessment for particulates and selected HMs and POPs (see table 1.4. for a summary of results). For PM_{10} emissions from road transport it should be considered that emissions due to the uncontrolled resuspension of dust have not been included in this study because of lack of knowledge. As a consequence, the decline in PM_{10} emissions from mobile sources may be overestimated. Conclusions on human exposure to selected air pollutants (PM_{10}, Benz(a)pyrene, benzene, lead) are dealt with in a separate section (see section 1.2).

- Under current policies, emission stabilisation targets for lead and dioxins/furans will be met in the year 2010. However, meeting the stabilisation targets for cadmium, copper, mercury and PAHs is not ensured under baseline conditions.
- Chemical risks can substantially benefit from the implementation of climate change policies, and policies in the context of acidification. Compared to 1990 emission levels, spill-over emission reductions due to climate action would be approximately 15% for PM_{10} and in the 5 to 20% range for selected HMs and POPs. For PM_{10}, an additional emission reduction of about 5% would result from policy action on acidification. Effects of acidification measures for HMs and POPs are negligible. Recognising such spill-over effects, emission stabilisation targets will be ascertained for all HMs and POPs studied, but achievement of stringent 2010 target concentration levels for PM_{10} remains a problem.
- It should be noted that uncertainties in baseline emissions, further emission reductions and costs are large for PM_{10} and studied HMs and POPs (at a factor 2 to 4).
- Promising technical options for further reduction of PM_{10} emissions are the further control of emissions from combustion in the energy sector and industry, and from major industrial processes. Additional direct costs for such measures come to € 1.5 billion per year. The largest reductions can be obtained in major industrial processes. Emission reductions that may result from advanced control of emissions from the sector transport (EURO-4 2005 standards) and residential combustion (optimised combustion) are estimated to be relatively small. However, it should be realised such reductions may be important for reducing emissions at the local city level.
- Advanced measures for PM_{10} control will have substantial side-effects on emissions of HMs and POPs. Compared to 1990 emission levels, substantial emission reductions for HMs, ranging from about 8% for lead to 35% for cadmium are anticipated.
- The use of alternative techniques for the preservation of wood which are not based on impregnation with PAH-based products would reduce PAHs emissions by 40% compared to 1990 emission levels.
- For accession countries, substantial emission reductions for HMs and POPs are expected under current policies. Such reductions will contribute to a significant improvement in the environmental situation in these countries, but also the EU.
- Under current policies, critical loads for lead on forest soils will still be exceeded in large parts of the EU. No exceedances of critical loads on forest soils are anticipated for cadmium and copper.

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2. Human health and air pollution

2.1 Introduction

2.1.1 Overview

Urban Stress- The Problem
By the year 2010, close to 7 billion people (1990: 5.3) will be living on this planet, and about 52% (43%) of them are expected to live in urban areas. 470 (290) metropolitan cities (above 1 million inhabitants) will inhabit 1.4 (0.8) billion people or 40% (35%) of the world urban population. 67 (61) of these cities will be located in Europe. Europe is one of the most urbanised continents and today some 70% of its population (560 million) is urban (UN/ESA, 1997), while urban areas (with a population density above 100 inhabitants per km^2) account for some 25% of the EU's territory (EEA, 1999).
Important limits to a Europe-wide assessment of the urban stress problems still exist. The information base for that purpose is largely unbalanced in that there is far more information on air quality issues and noise than on, for example, household waste (especially disposal problems), water use and waste water, or the extent of derelict and contaminated land. The present chapter therefore presents a partial view of this integrated issue which touches the majority of Europe's citizens. An increasing body of work on local sustainability indicators arises, developed for measuring progress on implementing Local Agenda 21; but a common analytical framework for effective comparisons is yet to be adopted. The urban stress information in this technical report is therefore limited to the air pollution problem.

Assessment of the state of urban air pollution
Urban areas have high concentrations of man-made air pollution sources, such as power generation, motor vehicle traffic, residential heating and industry. Urban air pollution not only poses a threat to human health and the urban environment, it also makes a significant contribution to regional and global atmospheric pollution problems. Most pollutants contributing to climate change, tropospheric ozone, and acidification originate from urban emissions. On the other hand, regional background levels of SO_2, NO_x, and O_3, for example, can accentuate urban air pollution and associated health risks.
As stated above a marked increase in urban population world-wide can be noticed, which will lead both to an increase in the emissions of pollutants, and to an increase in the number of people exposed directly to the effects of these pollutants.
Urban air pollution is experienced in most of the urban agglomeration above 0.5 millions, which makes it a world-wide problem and an issue of global concern.
The actual occurrence and frequency of increased concentrations depend, in addition to the size and average emission in the city, on local orography (e.g. flat terrain, basin, valley) and climate (e.g. average wind speed, frequency of calm weather conditions, occurrence of inversion layers) conditions.
What is to be considered as 'high concentration' is dependent on the pollutant, the type of exposure to be expected and the health or damage effects associated with this exposure.
To have accurate knowledge on the status and trends in urban air pollution, the emission sources and potential impacts have to be assessed to guide policy makers on the (inter)national and urban scale. The following assessments should therefore, ideally, be carried out:

1. assess the level of air pollution and its sources (such as energy, industry, transport, agriculture etc.)
2. identify and assess major impacts of air pollution and investigate cause effects relationships in order to set priorities for air pollution reduction strategies.
3. identify and evaluate the feasibility, effectiveness as well as environmental and socio-economic impacts of alternative reduction strategies.
4. estimate and assess, to the extent feasible, the socio-economic impacts and costs of inaction.

In this report the above mentioned items are being described, at the extent possible, in the form of Urban Air Quality Indicators.

2.1.2 Human health Indicators

As a starting point the World Health Organisation Air Quality Guidelines (WHO-AQG's) has been taken, extended and or replaced where possible by some newly (proposed) EU limit values (WHO 1997, EU, 1998, 1999).
In Table 2.1 an overview of the AQG's of relevant pollutants is presented. Even if these and all other air pollutants in a city are below their AQG values, it is still not possible to guarantee a healthy urban air quality. The air of a city consists of thousands of components and for many of these the effects and their combined influence on human health and ecosystems are still unknown. Some national limit values, as a precaution, are therefore even lower than these WHO AQG values.

Table 2.1: threshold values (derived from EU- and WHO- AQ values) used in this study

component	percentile	Value	derived annual average
NO_2	annual average	40 µg/m^3	40 µg/m^3
NO_2	99.9-1h	200 µg/m^3	50 µg/m^3
SO_2	max-24h	125 µg/m^3	24µg/m^3
PM_{10}	96-24h	50 µg/m^3	20 µg/m^3 (ref1)
PM_{10}	annual average	20 µg/m^3	20 µg/m^3(ref1)
O_3	Days with 8h >120 µg/m^3	20	20
Pb	annual average	0.5 µg/m^3	0.5 µg/m^3
Benzene	annual average	5 µg/m^3	5 µg/m^3
B(a)P	annual average	1 ng/m^3	1 ng/m^3

The effects on material and buildings aswell as the effects on flora and fauna in and around a city have not been taken into account in this study.

2.1.3 Exposure indicators

The actual exposure of the urban population to air pollutants is difficult to estimate. Next to estimating the spatial distribution and time variation of the pollutant concentration, the location and physical activity level (in relation with inhalation volume rate) of the population should be known. Since detailed data about the activity and actual location of the population is not available, the description of exposure has been limited here to the description of ambient air concentrations in relation to population density. Indoor environment exposure contributes dominantly to the exposure of the population; however, as mentioned before, an assessment of this problem is not within the scope of this report.
Recent some epidemiological studies (PEACE, APHEA) have shown that quantitative assessment of urban air pollution effects on human health is possible. In this report, however, we have limit ourselves to describe exposure in relation to air quality guidelines (AQG) as given in Table 2.1. The pollutants used are representative for three major air pollution exposure situations that may occur in European cities: 1) winter-type smog, 2) summer-type smog and 3) high annual average concentration levels.
The choice of pollutants for which air pollution levels were calculated was dependent on the following pre-conditions:

Air Quality guidelines should have been set or proposed;
Availability of emissions (and scenarios) needed as input for modelling;
Exceedance of guidelines expected based on earlier calculations, monitoring results or expert judgement.

Based on the above presented considerations the following components were distinguished in the study:

Benzene	annual average, exposure above 2 and 5 µg/m^3
B(a)P	annual average exposure above 1 ng/m^3
NO_2	annual average, exposure above 40 µg/m^3
SO_2	daily maximum, exposure above 125 µg/m^3
PM_{10}	annual average, exposure above 20 and 40 µg/m^3
PM_{10}	anthropogenic contribution, annual average, exposure above 20 and 40 µg/m^3
Secondary aerosols	= the contribution of SO_x, NO_x and NH_3 to PM_{10}
Pb	annual average, exposure above 0.5 µg/m^3
O_3	number of days with a 8 hour average above 120 µg/m^3, the summer average of the daily max 8 hour concentration

Notice that because secondary aerosols are a part of $PM_{2.5}$ and $PM_{2.5}$ is a part of PM_{10}, the concentration of secondary aerosols is always lower than $PM_{2.5}$ which, in turn, is lower than PM_{10}.

Spatial levels

The indicators for all components is presented at the following spatial levels:

EU15.
Accession countries.
Member State (MS) by MS.

Next, for each spatial level, the following sub-division has been made:

Large agglomerations (>750.000 inhabitants).
Cities between 100.000-750.000 inhabitants
Regional (remaining areas).

Indicators

For the components mentioned the following indicators are computed (annual emissions in appropriate units):

Air concentration	annual average in μg/m^3 or ng/m^3, weighted by population, i.e. larger cities contribute more to the EU15 average than smaller ones.
Exposure above targets	number of people exposed to levels above the targets[20].
Share of agglomerations in total exposure	the share of large agglomerations in the total exposure, expressed in % of the population in the EU15 that are exposed to levels above the target value. For example, a value of 78% indicates that 78% of the people that are exposed to values above the target are living in large agglomerations. This indicator shows whether the associated component is a typical urban problem or not. Obviously, this indicator is defined at the level of large agglomerations only.
Maximum Exceedance	average highest level of exceedance in a 2 by 2 km^2 grid expressed as a factor compared to the target value([concentration-target value]/target value). The addition of 'average' refers to the fact that we do not compute the grid with the maximum exceedance, but we compute (or assess) the average of all the 'maximum exceedance' grid cells in all cities which are covered by the indicator involved (depending on the spatial level). This indicator is the closest we can get to so called 'hot-spots' (i.e., busy streets).

It should be emphasised that the output of the RAINS model on ozone (see Technical Report on Acidification) is used to compute the values of the ozone indicators here. In short, the method is as follows: the RAINS model computes AOT60 and AOT40 for all areas (at an EMEP grid). This output is directly translated into regional number of days above the 8 hour average of 120 μg/m^3. For those grids that apply to cities and large agglomerations, the calculated regional results are interpreted as background concentrations and used as an input for the computation of ozone concentrations and exposures in cities and large agglomerations as needed for the human health assessment presented here.

Target levels:

It should be emphasised that target setting is an ongoing process and therefore some of the targets above are slightly outdated. Of special importance is that the European environmental ministers have adopted the following new targets:

NO_2: 200 μg/m^3 should not be exceeded for more than 18 hrs/yr
(derived annual average = 50 μg/m^3)

SO_2: 125 μg/m^3 should not be exceeded for more than 3 days a year.

PM_{10}: 50 μg/m^3 should not be exceeded at more than 35 days/yr (derived annual average = 30 μg/m^3)

O_3: no limit value has been adopted. The value in table 2.1 is based on the proposed target value.

[20] Exposure is calculated as the fraction of gridcells of 4 km^2 in a city above target level. Next the calculated fraction is applied to the number of inhabitants living in the city. The calculation method will therefore not distinguish exceedances with a spatial dimension less then 4 km^2, the socalled “hot spots” (e.g. a busy street or an industrial plant).

2.1.4 Study outline

Prime goals of this research were:
1. To calculate ambient air pollutant concentrations in selected cities in the European Union and compare calculated levels to WHO-AQGs or EU AQ limit values;
2. To estimate the number of people living in cities in the European Union exposed to exceedances of WHO-AQGs/EU limit values;
3. To estimate ambient air pollutant concentrations in selected cities in the European Union compared to guidelines and estimate the number of people exposed to those guidelines for various future scenarios[21].

The list of cities selected for this study is presented in section 2.2.
Table 2.1 presents the selected pollutant and the threshold values (air quality guidelines) to which pollutant levels were compared. The models used calculates annual average pollutant concentrations *(see also section 2.3.2*), so annual average thresholds needed to be derived from the actual thresholds as being defined by the EU and WHO respectively. In the final model calculation antrophogenic PM_{10} are be used to describe the consequence for PM_{10} (action levels defined in case PM_{10} are substantially influenced by natural sources).

Calculations have been performed for all selected cities on a 2x2 km resolution resulting in the spatial distribution of the average urban contribution to the total concentration field (when the rural background concentration is added to the urban contribution, the resulting concentration is often called *urban background* concentration).

Unless otherwise stated, the urban background concentration is meant throughout this report when referring to pollutant concentrations in cities. Air pollutant concentrations on hot-spots like busy streets or near industrial sites can be much higher, depending on the pollutant.

In Chapter 2.2 all methods used are described. This relates to the selection procedure of cities and processing of emission and meteorological data, needed to feed the dispersion model Eutrend, which characteristics are also described. The methodology used to calculate regional background concentrations and urban contribution to the concentration field is presented. Algorithms for the conversion of annual means into percentiles are described. Results of this study for the base year and future scenarios are presented in Chapter 2.3. For further background reading on this study reference is made to (Eerens and Sluyter, 2001).

2.2 Methods

2.2.1 Emissions

More information on the collection and processing of national emissions including scenarios, can be found in section 1.1 on chemicals and particulate matter. For the purpose of calculating urban contributions to the total air pollution field, emissions have been attributed to cities. The methodology is described in Section 2.2.3.
Emission input to the Eutrend model is divided into four source categories:

Traffic
Refineries and power plants
Households
Agriculture

Emissions available were grouped into ten SNAP categories. Anthropogenic source categories distinguished in this study are as follows with corresponding SNAP90-codes added in brackets and deviations from SNAP90 indicated in italics:

Public power, cogeneration and district heating	(SNAP-01)
Residential, Commercial and Institutional (RCO) combustion	(SNAP-02)
Industrial combustion *(excl. combustion in petroleum industries)*	(SNAP-03)
Production processes (*incl. combustion in petroleum industries)*	(SNAP-04)
Extraction and distribution of fossil fuels	(SNAP-05)

[21] Within a scenario, measures taken with respect to Climate Change (see Technical background Report 2), Acidification, Eutrophication and Tropospheric Ozone (see Technical background Report 4) are of major importance to the results presented in this report (i.e. spill over effects).

Solvent use	(SNAP-06)
Road transport	(SNAP-07)
Other mobile sources and machinery	(SNAP-08)
Waste treatment and disposal	(SNAP-09)
Agriculture	(SNAP-10)

The SNAP categories have been reallocated into four groups representing the four EUTREND categories:

Group *households*: SNAP04
Group *refineries & power plants*: SNAP01+SNAP03+SNAP06+SNAP09
Group *traffic*: SNAP07+SNAP08+SNAP05
Group *agriculture*: SNAP02+SNAP10

Eutrend uses standard emission heights for area sources. Using additional information on emission heights available through this inventory (Smeets and van Pul (1999), the effect of changing the emission heights on resulting concentrations has been modelled and validated against measured concentrations. Correction factors have been deduced and used to correct concentrations calculated by Eutrend.

Eutrend model calculations are based on the prerequisite of one surface roughness (roughness length for grass). To compensate for the higher surface roughness of cities, resulting concentrations have been corrected.

2.2.2 Selected cities

Starting point of the city selection process was the group of cities studied before as part of the Europe's Environment programme (Dobris study; Sluyter et. al. (1995)). All pan-European conurbation's with a population > 500.000 inhabitants were selected and described as part of the Europe's Environment programme, and if no such city existed in a country, the capital city (105 cities in total). As a result detailed information on the 105 cities is available (size, population, emissions, monitoring results).

The final selection for the purpose of this study was made on basis of the availability of recent air quality monitoring results from AIRBASE[22] and in addition literature (air quality reports on the cities) available at RIVM.

It is not trivial to define a city, especially the city boundary. This is an important issue because existing (environmental) data is often available at the level of administrative units (municipalities) rather than on the city level. A conurbation can be made up of several municipalities. Modelling urban air quality, as has been carried out within this study, requires a city definition (and data available) on the basis of the physical boundary level (built-up area).

In the framework of the Europe's Environment studies, effort has been devoted in collecting information on the size of the conurbation's and number of inhabitants of cities. This was done by sending questionnaires to all municipal authorities of the selected cities and through literature research (ref Dobris). The variations found in city size and number of inhabitants between the various sources proved to be quite large and this poses a problem when:

attributing (sub)-national emission amounts to urban areas (top-down approach).
calculating the number of inhabitants exposed to exceedances of certain air quality levels;

In order to homogeneous the city data set as much as possible for population figures, area and by this emission estimates, the following procedures were followed for estimating the number of inhabitants and the urban area:

Population
The estimated number of inhabitants for the conurbation in 1990 as collected within the Europe's Environment studies was taken as starting point. If data were missing or were to variable between various sources, UN

22 AIRBASE is the pan European air quality information system operated by the European Topic Centre on Air Quality, under contract to the European Environment Agency. The system can be accessed on-line through the Internet (http://www.etcaq.rivm.nl).

population statistics have been used (ref). UN population projections have been used for 1995, 2000 and 2010 scenario calculations.

Urban area
Using a GIS, an overlay has been built between a coverage containing city centre co-ordinates and radius of the city (estimations from the Europe's Environment database) and a polygon coverage containing the land use category 'built-up area'. All polygons falling at least 1% within the radius were labelled as urban. The resulting (vector)-coverage has been re-sampled to a 2x2 km grid. The total urban area for any of the selected cities has been estimated by counting the 2x2 grids. Note that no predictive information is available on city size for 2010.

Table 2.2: EU-15 cities selected, number of inhabitants in 1990, population projections for (1995), 2000 and 2010 (population x1000), area of the city and conurbation as used in the Europe's Environment study, area of the conurbation after performing a GIS operation.

City	Country	Pop.	Pop.	Pop.	Pop.	Area City	Area Conur	Area GIS
		1990	1995	2000	2010	(km2)	(km2)	(km2)
Amsterdam	*The Netherlands*	1053	1108	1149	1171	162	583	356
Athens	*Greece*	3070	3093	3103	3117	350	101	256
Barcelona	*Spain*	2913	2819	2819	2819	91	457	324
Berlin	*Germany*	3288	3317	3337	3347	330	880	928
Birmingham	*United Kingdom*	2301	2271	2271	2271	-1	150	780
Brussels	*Belgium*	1148	1122	1122	1123	-1	402	284
Cologne	*Germany*	2855	2984	3067	3108	120	112	376
Copenhagen	*Denmark*	1345	1326	1326	1326	80	670	540
Dublin	*Ireland*	916	911	913	944	115	115	356
Helsinki	*Finland*	872	1059	1163	1252	105	242	56
Lisbon	*Portugal*	1658	1863	1971	2178	-1	100	228
London	*United Kingdom*	10570	10567	10567	10567	-1	1580	2008
Luxembourg	*Luxembourg*	73	74	75	75	-1	22	60
Lyon	*France*	1266	1319	1359	1392	-1	150	116
Madrid	*Spain*	4172	4072	4072	4072	-1	100	288
Manchester	*United Kingdom*	2282	2252	2252	2252	-1	280	760
Marseilles	*France*	1230	1234	1343	1261	-1	75	156
Milan	*Italy*	4603	4251	4251	4251	-1	160	160
Oslo	*Norway*	697	702	706	710	200	400	432
Paris	*France*	9334	9523	9638	9694	105	1200	1392
Rome	*Italy*	3710	3678	3678	3678	125	63	108
Stockholm	*Sweden*	1490	1545	1582	1615	188	333	800
Stuttgart	*Germany*	2485	2608	2688	2728	-1	200	264
Vienna	*Austria*	2055	2060	2072	2098	190	211	388

2.2.3 Urban emission processing

Urban background air quality levels have been calculated on a city-by-city basis by using the Eutrend model (see section 1.2.2.3.2). In order to calculate the urban contribution to the total concentration field, emission estimates were needed on a city-by-city basis. Emissions have been estimated by using a top-down approach. General information on the emission inventories used is given in section 2.2.1).

POINT SOURCES
By assumption, only those emissions from point sources within or nearby a city are considered important. All point sources within the radius of the city plus 5 km were selected[23].

AREA SOURCES
The total emission values of the area sources within the urban areas and total area sources were supplied by TNO (see also section 2.2.1). The urban emissions were re-allocated to the 2x2 km grid.

[23] The radius of cities is calculated as the square root of the conglomeration area divided by π

Total area sources, needed to calculate background concentration fields, have been re-allocated to 1x0.5 degree longitude-latitude grids.

The 1x0.5 degree emission file includes the 2x2km urban emission files. In order to avoid urban emissions being used in the calculations twice (once for background modelling, once for urban modelling), the urban emissions (within a NUTS3 region) have been subtracted from the emissions used for background modelling. If the subtraction lead to a negative emission estimate, the emission amount was set at zero.

The surface of the 1x0.5 degree areas differs throughout Europe, because the latitude-longitude projection is not equivalent. The area to which the background emission applies thus differs depending upon the latitude. The correct areas were calculated and used while attributing emissions to the input file used for background modelling.

2.2.4 Meteorological data

Meteorological data available consists of series monitored on point locations (meteorological stations). In order to use meteorological data as input to the Eutrend model, point data first has to be converted to data fields. Interpolation of meteorological data within this project was performed using an inverse distance weighting procedure to grids with a resolution of 1°x0.5° lat/lon. Most emphasis in the method is put on selecting the most representative stations in the vicinity of the site under consideration.

For each interpolated grid point of the resulting meteorological data fields, the ten most nearby stations were used for interpolation after a check on availability of data on precipitation, temperature, dew point temperature and wind speed.

Stations with missing data for one of the parameters were deleted from the station set. The remaining stations of the ten which satisfied the data availability criteria were assigned to four quadrants (N, E, S, W) around the grid points and sorted depending on the distance from the grid point to the station location.

A maximum of four stations were selected for interpolation, one from each quadrant. If no stations remained from the selection procedure, interpolation could not be performed and the parameter values were assigned 'non valid'. If only one station remained from the selection procedure, the parameter values from this station were assigned to the grid point. If two or more stations remained, a last check was performed. Stations, which were separated from the grid point by more than two times the average distance of the other remaining stations to that point, were deleted.

Before interpolation, the data were screened in order to remove unrealistic outliers. Screening was performed on precipitation, temperature and precipitation data. Finally, grid point interpolation was performed for each 6 hour time step.

Eutrend divides Europe into 50 meteorological regions. For the centre point of each region, 96 hour backward trajectories were calculated for 6 hour time steps[24]. Each trajectory point was combined with the corresponding interpolated meteorological data. The trajectories were uses as input for the Eutrend meteorological pre-processor, which calculates long-term statistics for all 50 regions. These are used as input in EUTREND (Van Jaarsveld, 1989 and Potma, 1993).

2.2.5 Modelling

EUTREND model
The long range transport model EUTREND is used for the calculation of the annual average air concentration of the gases SO_2, NO_x and benzene and for the aerosol mass concentrations of primary (PM_{10}, $PM_{2.5}$) and secondary particulates (calculated as the sum of SO_4, NO_3, NH_4), lead, Cadmium and benz(a)pyrene over Europe on the basis of the earlier mentioned emissions inventory. The EUTREND model is an European version of the Operational Priority Substances (OPS) (Van Jaarsveld, 1995). This family of models can be characterised as Lagrangian models in which the transport equations are solved analytically. Contributions of the various sources are calculated independent of each other using backward trajectories, local dispersion is introduced via a Gaussian plume formulation. Average concentrations are not determined from sequential (e.g. hourly) calculations but from concentrations calculated for a limited number of meteorological situations (classes) using a representative meteorology for each of the classes. Meteorological data is taken partly from the Numerical Weather Prediction model of the European Centre for Medium

[24] ECMWF wind fields were used for this purpose.

Range Weather Forecasts (ECMWF) in Reading (UK) and partly from observations at meteorological stations all over Europe.
Dry deposition, wet deposition and chemical transformation are incorporated as first order processes and independent of concentrations of other species (Van Jaarsveld, 1995; Asman and Van Jaarsveld, 1992).
For the aerosol calculations five particle size classes are used in the EUTREND model, each characterised by a (monodisperse) particle size with corresponding properties calculated by the semi-empirical model of Sehmel and Hodgson (1980) which gives similar results as the more theoretical model of Slinn (1983). Concentrations and depositions are calculated for each of these classes and weighted with the percentage of the total particle mass appointed to the individual classes. Such an approach is especially useful for the modelling of primary-emitted particles because they usually cover a broad range of particle sizes, often including a significant fraction of large particles. Particle growth is not incorporated in the present model but is implicitly assumed to take place in the lowest size-class (d < 1 μm). In earlier support of the European Commission (*ref ivm-report, NILU-report, us-dutch symp*) the EUTREND model has been used to calculated the primary and secondary contribution of these sources to the PM_{10} and $PM_{2.5}$ concentration in Europe.

Sea Salt calculation
As part of its research on acidification RIVM has developed a method to determine the concentration of base-cations in ambient air from rain water measurements (Draaijers et al. 1996), With some slight modifications this method has been applied to calculate the concentration of sea salt aerosols in ambient air.
Earlier studies have found (e.g. Woolf et al., 1987) that the typical diameter of sea salt aerosol, that is produced at the ocean surface by the bursting of air bubbles (a minimum wind speed of 3 to 4 $m.s^{-}1$ is required), is 1-2 μm (although extending to sizes greater than 10μm). Therefore long-range transport of sea-salt aerosol can be expected. Using scavenging ratios, the air concentrations of these generated sea salt particles can be calculated from precipitation concentrations. Ambient air concentrations derived this way will reflect the large scale background situation.
The scavenging ratios were derived (Draaijers et al. 1996) from simultaneous measurements of base-cations concentrations in precipitation and surface-level. This approach is based on the premise that cloud droplets and precipitation efficiently scavenge particles resulting in a strong correlation between concentrations in precipitation and the surface-level air (Eder and Dennis, 1990). Scavenging ratios have been found reasonably consistent when averaged over one year or longer (Galloway et al., 1993). For this reason annual mean precipitation concentration has been used to infer annual mean air concentrations. The scavenging ratio (SR) is defined as:

$$SR = [C]_{rain} * Rho/[C]_{air} \quad [1]$$

Where $[C]_{rain}$ denotes the concentration in precipitation (mg/l, ~mg/kg), $[C]_{air}$ the concentration in ambient air (in ug/m^3) and Rho the density of air, taken as 1200 g/m^3. For the typical size range of sea salt particles the following relationship between the scavenging ratio and mass median diameter (MMD, in um) can be derived from data of Kane et al., (1994):

$$SR = 188 * e^{(0.227*MMD)} \quad [2]$$

Rearranging equation [1] and [2] gives a simple empirical model describing the relationship between air concentration at one hand and precipitation concentration and MMD at the other hand (Draaijers et al. 1996):

$$[C]_{air} = ([C]_{rain}*1200)/(188 * e^{(0.227*MMD)}) \quad [3]$$

Precipitation concentrations will reflect atmospheric concentrations of the entire atmospheric column from cloud top to surface level and thus will reflect the large scale 'background' situation. A strong correlation with surface level air concentrations will only be present in well-mixed conditions at sufficient distance from sources. Close to sources surface level air concentrations usually will be considerably higher. As a consequence, near the coast the contribution of sea salt to the total suspended matter *in the surface level* air will be underestimated by using the method described above. On the other hand rain occurs mainly during western wind circulation's, leading to an *overestimation* of the yearly average concentration.

Soil resuspension contribution to fine particulates
Soil dust production is connected to aeolian transport of soil grains. From a physical point of view, the particles motion initiated by wind is controlled by the forces acting on them. For a particle at rest, these forces are the weight, the interparticle cohesion forces and the wind shear stress on the surface. The first ones are size dependent; the last one depends on the transfer of the wind energy to the erodible surface which is controlled by the presence

of roughness elements on the surface. All together determine the minimum threshold friction velocity (defined as the square root of the ratio of surface stress to air density), u_{*t}, required to initiate particle motion (Marticorena and Bergametti (1996).
The fine suspended particles constitute the dust flux, which is referred as the vertical flux. Since dust production experiments can hardly be performed in wind tunnels, the physical processes of dust emission are not yet well identified and described.

Composition
The chemical composition of soil dust will reflect the contributions of elements present in the earth's crust, provided other contributions from e.g. anthropogenic or volcanic sources are negligible. However, this premise does not hold for all constituents of the soil dust aerosol. Some trace components are considerably enriched compared with their crustal abundance. For this purpose, a so-called enrichment factor (EF(x)) has been defined (Rahn, 1976):

$$EF(x) = (x)/(Ref)_{aerosols} / (x)/Ref)_{source}$$

where x is the element under consideration and Ref a reference element. An appropriate choice of reference elements is required, as well as a tabulation of elemental compositions of source materials. Elements that are useful as reference elements for crustal material include silicon, aluminium, iron, and titanium (Chow et al., 1994; Warneck, 1986). All these elements are abundant in rocks. A determination of the contribution of the crustal component to atmospheric aerosol is difficult as a result of imprecise knowledge of the composition of material resulting from the wind erosion of soils. Usually a surrogate composition is used. Possibilities include the average composition of crustal rocks, bulk soil, or the aerosol-size fraction of the soil. In most cases, globally averaged rock or soil is used. The elemental composition of the soil-derived fraction of aerosol, however, has been found to deviate appreciably from that of average crustal rock or average soil. The relative abundance of the major elements have been found to differ individually by factors of about three (Warneck, 1986). The EF(x) values should therefore only be regarded as order of magnitude estimates of crustal sources. The enrichment factor express the fraction of elemental mass for particles that are suspended in air to the fraction of elemental mass for all soil mass particles. The fraction of elements in suspended particles, however, will not be the same as the elemental fraction in the whole soil because the greatest part of the mass for suspended particles will be for sizes < 10µm. It is known that the composition of soil varies as a function of size. Sand particles range in size from 50µm to 2mm. The dominant composition of sand is quartz, which is in general chemically inactive. Alkaline mass, for example, may be assumed completely associated with the silt and clay fractions of the soil (this is, particles < 50µm). It follows that a simple estimation of the enrichment factor would be the inverse of the fraction of silt and clay mass to the whole soil mass. since, by definition, silt plus clay fractions are equal to 1 - sand fraction, the enrichment factor computed for the soil texture is simply $(1-s)^{-1}$, where s is the sand fraction of the soil mass. (Gillette et al., 1992).

Method of calculation
Based on the average concentration of soil/rock specific components Ca, corrected for its abundance in Sea Salt aerosol, an estimate of the contribution of resuspended dust has been made. Based on measurements a ratio of respectively 0.65, 0.23, 0.35 for PM_{10}/TSP $PM_{2.5}$/TSP and $PM_{2.5}/PM_{10}$ has been used. The Ca concentrations has been calculated with the infer method as described at the previous paragraph, the scavenger coefficients are derived from Erisman et. al., 1996.

The average concentration of the Ca indicator components in seasalt and soil and/or crustal material (handbook chemistry and Physics, 1995) is given in the table below, in the table also some other elements that can be used for the estimation of resuspended soil is presented.

Table 2.3 The average concentration of elements in soil and crustal material

Element	Mw	Seawater[1]	Soil	Crustal material	Ratio for resuspension (used)	Ratio for Sea salt contribution (used)
		(mg/kg)	(g/kg)	(g/kg)		
Si	28.09	0.02-40/2.2	330	277-311	3.3	15600
Al	26.98	0.002	71.3	77-81	13.5	17.10^6
Fe	55.85	0.002-0.02	38	34-50	26.3	$1.7.10^6$
Ca	40.08	407	13.7	26-42	49.3*	84.4
Mg	24.31	1252	6.3	21-33	73.3	27.4
Na	22.99	10560	6.3	24-32	66	3.25
K	39.09	392	13.6	21-30	57.8	87.7
Mn	54.94	0.0002-0.01	0.85	0.67-0.95	1180	$3. 10^6$
Ti		0.001		4.4-5.7	227	$33. 10^6$
Br		67		0.0024	400.10^3	513
Sr		0.0079		0.37	2700	$4.3\ 10^6$
S		904		0.26-0.35	3850	38
Cl	35.5	19000		0.145	7000	1.81
V	50.94	0.0025	0.1	0.098-0.135	10000	10^7

[1]At a salinity of 3.5 %

* To derive at PM_{10} values the factor 32 has been used (49.3 multiplied by 0.65, the ratio between TSP en PM_{10})

Validation: Cross-examination of the result has been used to estimate the validity of the results. The average uncertainty is estimated to be not more than a factor two.

2.2.6 Concentration calculation methodology

This study focuses on urban background air pollutant concentration fields; pollutant levels found in areas not directly influenced by sources like traffic and industry representative for levels to which citizens are at least exposed. Urban background levels are described as being composed of two parts: a regional background contribution and an urban contribution.

REGIONAL BACKGROUND CONTRIBUTION

The regional background concentrations in the vicinity of the selected cities has been calculated by running the EUTREND model (see section 2.2.5) The following procedure was followed:

1. For every city, four co-ordinates (North, East, South and West direction) were defined 10 km outside the city boundary by using an automatic GIS procedure,
2. The Eutrend model was used to calculate the annual average background concentrations for each of the four locations and for the four source categories as defined in section 2.2.1,
3. The four concentrations were averaged to give the annual average regional background concentration contribution per source category.

Appendix G presents the calculated annual average background concentrations for each EU country and component

URBAN CONTRIBUTION

The contribution of the city itself to the urban background concentration level was calculated using the following procedure:

Eutrend was used to calculate the annual average concentration for the four source categories and every 2x2 km urban grid cell using the urban emissions input files (see also section 2.2.3 on emissions);
The output (concentration values) have been corrected per source contribution and city to take into account differences in roughness length in cities and non standard emission heights for emissions from households[25] (Eutrend does not take into account the high roughness length in cities and emission heights, see also section 2.2.1);

[25] We assumed that the higher the population density in a city the larger the variation in building height, resulting in increased roughness and average height of household emissions. Increased roughness results in decreased contribution from lower sources (e.g. traffic) and increased contribution from higher sources.

The concentration values calculated for the gridcells have been averaged over all gridcells in a city and the four source category contributions have been summed up resulting in the annual average urban contribution.

SCENARIO CALCULATION
For each source category the relative change in emission per country (compared to the 1990 situation) has been scaled separately for the regional background and the city background contribution. The sum of the eight contributions is considered to be representative for the new situation. The following scenario's for 2010 has been calculated according to the prescribed procedure:

Baseline (Policies in place and in pipeline)
AP-NT (Accelerated Policies-No Trade)[26]: Spillover effects from Climate Change (see Technical Report on Climate Change) and Acidification (see Technical Report on Acidification) policies will be maximal.
AP-FT (Accelerated Policies-Full Trade) As scenario 2, but assuming that a large share of the CO_2 emission reduction will be obtained by trading in emission rights. The result is that lower spillover effects from climate change policies can be expected.
TD (Technology Driven): For human health, spillover effects of TD measures taken for acidification and Eutrophication abatement are important, especially for transport (catalysts). The measures as mentioned above are fully applied in the different sectors to maximally reduce PM-emissions (and related components, such as HMs and dioxines).

CALCULATION OF THE EUROPEAN POPULATION EXPOSURE
The exposure of the European population has been derived as follows:

1. The calculated average regional background per country is assumed to be representative for the exposure of the population living in rural communities and cities with less then 100.000 inhabitants.
2. The calculated average, population weighted, concentration for cities with over 750.000 inhabitants per country is considered to be representative for the population in a country living in agglomerations with more then 750.000 inhabitants.
3. The population of a country living in cities between 100.000 and 750.000 inhabitants is taken as the average concentration of the regional background and the concentration in cities with more then 750.000 inhabitants.
4. The number of inhabitants per category is presented in table 2.4.

[26] For fine particulates two variants within the AP-NT scenario's have been calculated, first the effects with only “spill-over” measures and a second calculation with some specific measures to reduces the emission of primary particles (see chapter chemical risk).

Table 2.4: Population per European country for 1990 and 2010, divided in three categories.

	1996				*2010*			
Country	total millions	regional	urban pop 100-750	urban pop >750	thousands	regional	urban pop 100-750	urban pop >750
Albania	3,3	3,04	0,24	0	3,8	3,43	0,32	0
Armenia	3,5	1,91	0,42	1,21	3,9	2,15	0,33	1,422
Austria	7,7	4,81	0,85	2,055	8,2	5,17	0,93	2,10
Azerbaijan	7,2	4,89	0,52	1,751	8,5	5,66	0,64	2,17
Belarus	10,3	5,74	2,87	1,65	10,1	5,06	3,10	1,90
Belgium	10,0	6,37	2,45	1,148	10,5	6,85	2,53	1,12
Bulgaria	8,7	5,78	1,75	1,19	8,0	5,00	1,77	1,19
Bosnia	4,3	2,96	1,35	0,00	4,4	2,48	1,93	0,00
Croatia	4,5	3,17	0,50	0,85	4,4	2,73	0,55	1,13
Czech	10,3	7,80	1,31	1,20	10,0	7,47	1,33	1,24
Denmark	5,1	3,18	0,62	1,345	5,4	3,41	0,64	1,33
Estonia	1,6	1,02	0,55	0	1,3	0,81	0,53	0,00
Finland	5,0	3,13	0,98	0,872	5,4	3,04	1,10	1,25
France	56,7	33,44	10,50	12,79	62,5	37,79	11,32	13,36
Georgia	5,5	3,53	0,66	1,277	5,5	3,27	0,74	1,47
Germany	79,4	32,20	12,69	34,482	86,9	37,81	11,77	37,31
Greece	10,1	5,04	1,10	3,948	11,1	5,84	0,95	4,30
Hungary	10,4	4,73	1,20	4,43	9,3	2,64	1,22	5,45
Iceland	0,3	0,10	0,15	0,00	0,3	0,13	0,18	0,00
Ireland	3,5	2,41	0,17	0,916	3,8	2,64	0,19	0,94
Italy	56,7	34,65	7,41	14,68	56,8	35,32	7,55	13,90
Latvia	2,7	1,54	0,22	0,921	2,2	1,11	0,21	0,92
Lithuania	3,7	2,25	1,49	0	3,6	2,04	1,57	0,00
Luxembourg	0,4	0,30	0,08	0,00	0,5	0,38	0,08	0,00
Moldova	4,4	3,22	1,15	0,00	4,6	3,18	0,47	0,94
Netherlands	15,0	7,26	5,59	2,1	16,8	8,49	6,02	2,26
Norway	4,2	3,02	1,22	0,00	4,5	3,25	1,27	0,00
Poland	38,1	22,34	7,56	8,22	39,4	22,06	8,43	8,89
Portugal	9,8	6,57	1,58	1,658	10,2	6,07	1,91	2,18
Romania	23,2	16,09	5,06	2,054	22,0	14,31	5,53	2,18
Russian	148,3	77,81	40,44	30,04	141,1	69,50	40,81	30,75
Slovak r	5,3	4,57	0,69	0,00	5,4	4,66	0,79	0,00
Slovenia	1,9	1,51	0,41	0,00	1,9	1,43	0,44	0,00
Spain	39,0	20,13	11,00	7,834	40,2	21,05	11,47	7,65
Sweden	8,6	5,44	1,66	1,49	9,4	6,00	0,97	2,39
Turkey	56,1	24,21	18,74	13,15	74,6	25,72	25,86	23,04
Ukraine	51,9	31,08	11,03	9,78	49,0	27,28	11,39	10,36
United Kingdom	57,4	1,37	37,73	18,31	59,4	2,28	38,60	18,48
Yugoslavia	10,2	8,22	0,77	1,162	10,5	8,53	0,71	1,267
Total	784	407	195	183	815	406	206	203

2.2.7 Conversion of annual means into percentiles

Apart from the long-term WHO-AQG for annual means, also short-term AQG exist. The specific statistic which must be evaluated to check for short-term exceedances depends on the compound in question. Because EUTREND only produces annual averages functional relations between annual averages and short-term statistics, like 98-percentiles or maximum values, are desirable.
For nitrogen-oxides a further complication is the fast chemical reactivity of these compounds with ozone. Due to this, dispersion calculations by EUTREND are performed only for the conserved quantity NO_x. Because AQG exist for NO_2 a conversion between the annual means of NO_x to NO_2 is necessary.

Below, for all compounds used in this study, where relevant, empirical relations between long- and short-term statistics are discussed.
CONVERSION ALGORITHMS

annual averaged NO_x → annual averaged NO_2
The following conversion methods was chosen:

BUWAL, an empirical powerlaw relation between NO_x and NO_2 derived from measurements in Germany and Switzerland.

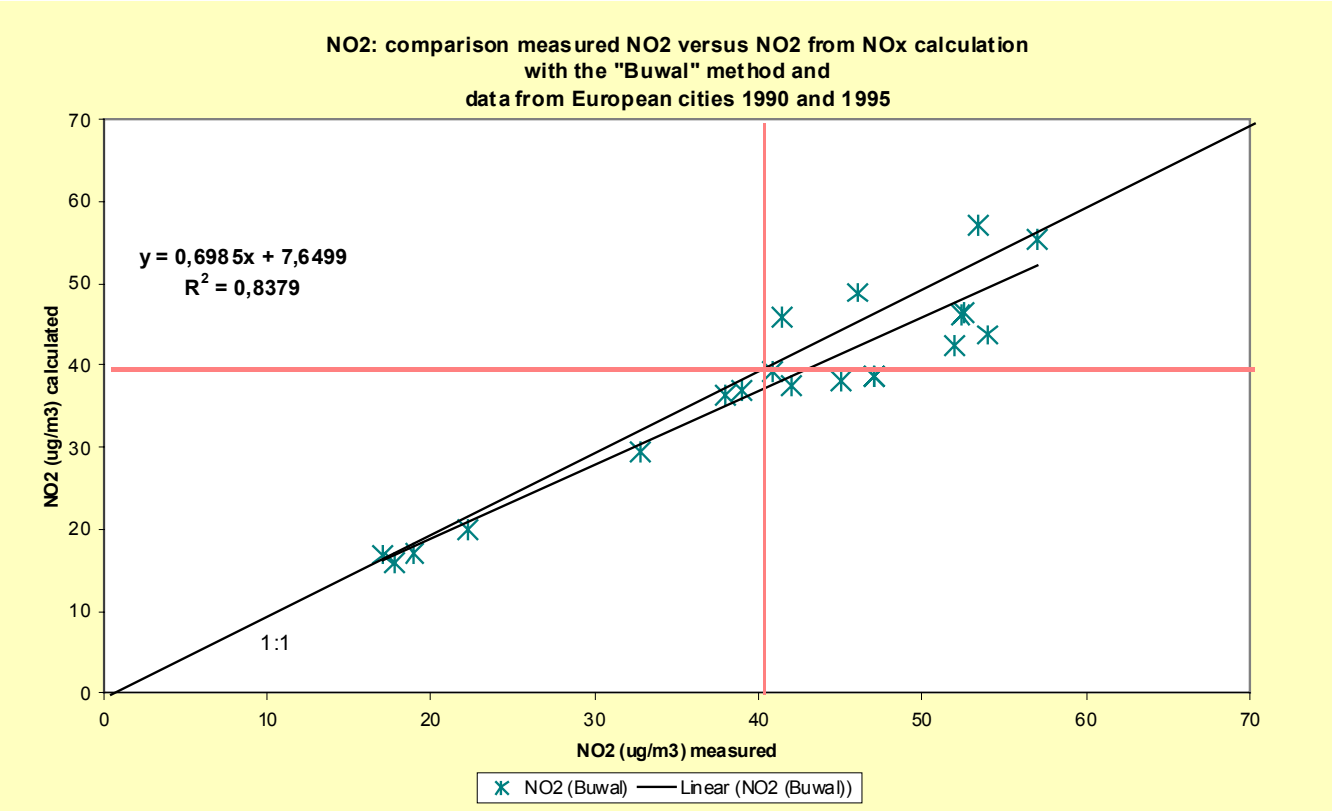

Figure 2.1: Conversion from NO_x to NO_2. Shown are the data collected in this project together with the best powerlaw fit.

Figure 2.1 shows the available measured annual averaged NO_x and NO_2 data collected in this project (from Amsterdam, London, Lisbon and Paris).The black line is the best powerlaw fit for the data. The correlation coefficient of 0.84 is high enough to ensure that the conversion can be done in a reliable way and does not introduce to large uncertainties in the data.
The relation reads:

$$\text{annual average } (NO_2) = 0.055*NO_x + 55*(1-e^{-0.0117*NOx})$$

annual averaged NO_2 → annual maximum of hourly NO_2
Figure 2.2 shows a selection of the complete set of data on annual averaged NO_2 versus hourly, annual maximum values. A linear relation between the 1hr. maximum and the annual average is investigated by a regression analysis on these data. The full data set has not been used because the quality of some data is doubtful or not representative. Especially the measurements from the Eastern European cities showed a clearly different relation between the annual mean and hourly maximum values. Of the 65 available measurements, 52 have been selected for the regression.

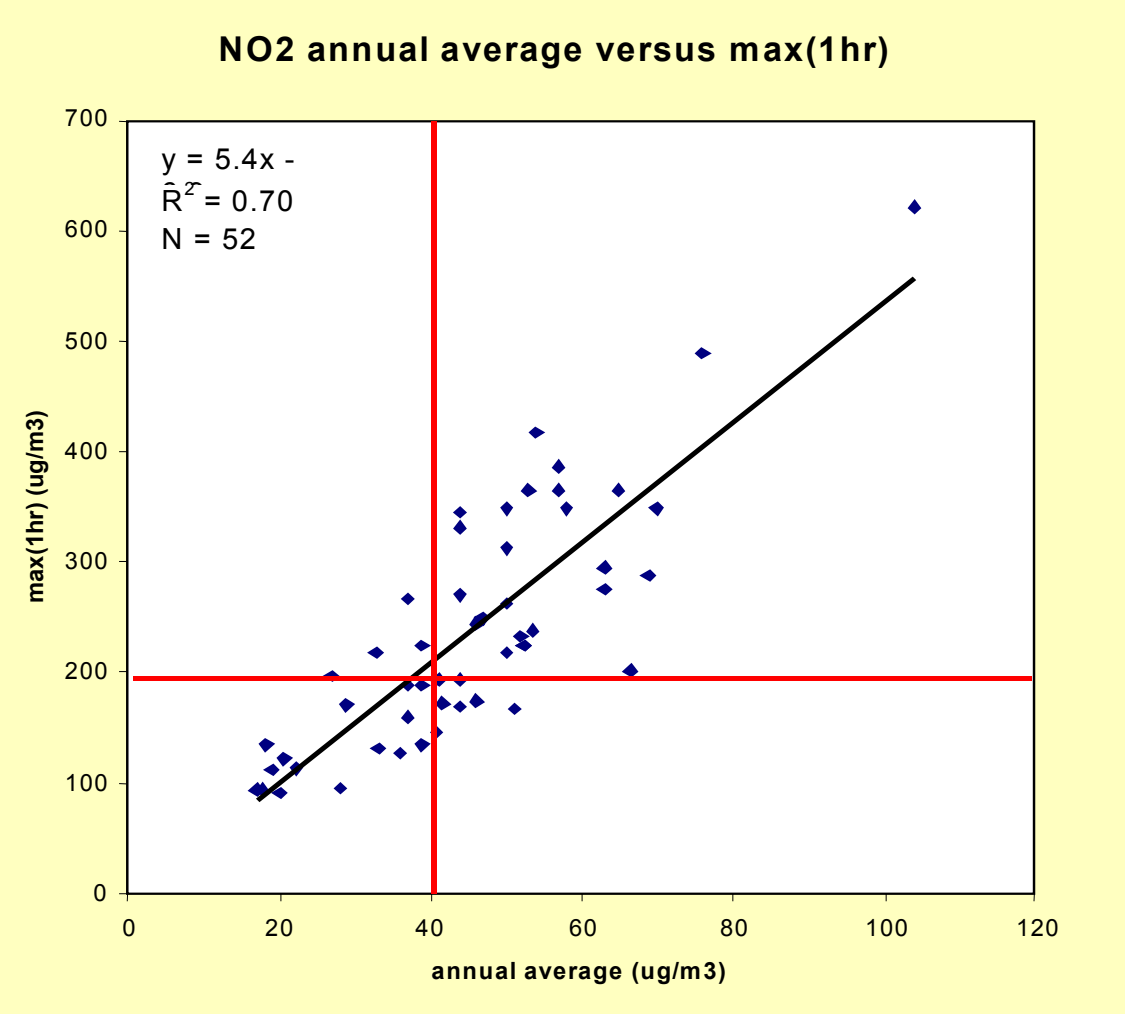

Figure 2.2: Annual average NO_2 versus annual hourly maximum value for a selection of all available data. The best-fit linear relation is shown as a solid line as well as the parameters of the fit

The regression results read:

$$[NO_2 \text{ max 1 hr}] = 5.4[NO_2 \text{ annual average}] - 6.9\ \mu g/m^3,$$

with a correlation coefficient of 0.70. The annual average corresponding to the maximum hourly value AQG of 200 $\mu g/m^3$ equals 38 ± 8 $\mu g/m^3$. The max 1 hr value corresponding to the annual mean AQG of 40 $\mu g/m^3$ equals 210 ± 45 $\mu g/m^3$. This shows that the short-term AQG is only marginally more stringent than the long-term AQG. In practise this implies that both statistics have to be checked for compliance.
.

If we use for the short-term AQG, not the hourly maximum, but instead the eight largest hourly value in a year (8th max)[27], the relation between long-term and short-term AQG changes. The 8th max value has been derived from the maximum and the annual mean by assuming a lognormal distribution for the hourly averages.
Using the same data set as before the linear regression result reads:

$$[NO_2 \text{ 8th max}] = 4.1[NO_2 \text{ annual average}] - 4.6\ \mu g/m^3,$$

[27] In the final EU directive the 200 ug/m3 is allowed to be exceeded 18 times, underpinning the conclusion that the annual average has become the most stringent one.

correlation coefficient of 0.77 for N=52. Now, the annual average corresponding to the short-term AQG of 200 µg/m^3 equals 50 ± 8 µg/m^3. The 8th maximum corresponding to the annual mean AQG of 40 µg/m^3 equals 159 ± 28 µg/m^3. This illustrates that for this choice for the short-term AQG the long-term AQG is more stringent

SO_2

For SO_2, apart from the guideline for the annual average of 50 µg/m^3, a short-term guideline exists for 24-hour averages[28] of 125 µg/m^3. In order to investigate the relation between these two parameters a linear regression has been calculated on the measured urban concentrations gathered in this project. *Figure 2.3* shows these parameters for a selection of the available data together with the regression line.
The fit yields:

$$[SO_2 \text{ max 24 hr}] = 4.7[SO_2 \text{ annual average}] + 13 \text{ µg/m}^3,$$

with a correlation coefficient $r^2 = 0.71$, for N = 64. Also here, a number of stations, mainly Eastern European, have not been included in the analysis. These stations showed a clearly higher ratio short/long term statistic. The annual average corresponding to the maximum 24 hr value of 125 µg/m^3 equals 24 ± 4 µg/m^3. The maximum 24 hr value corresponding to the annual mean AQG of 50 µg/m^3 equals 250 ± 28 µg/m^3. This shows that the short-term AQG is exceeded earlier than the long-term AQG.

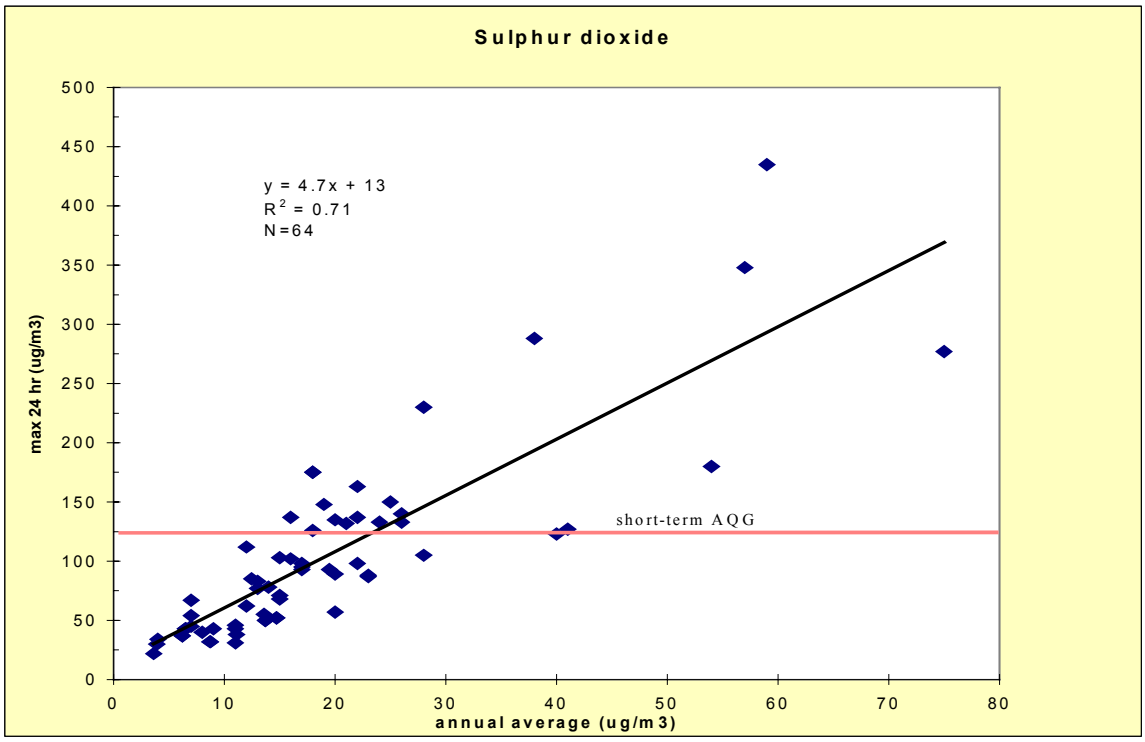

Figure 2.3: Annual average SO_2 versus annual 24-hour maximum value for a selection of all available data. The best-fit linear relation line is shown as a solid line accompanied by the parameters of the fit.

PM_{10}
For PM_{10} the long-term AQG equals 20 (40)µg/m^3 and the short-term AQG, calculated as the maximal 24 hr average, equals 50 µg/m^3 (respectively 7 and 35 exceedances allowed). Using experimental PM10 data collected in this project a linear relation between 24 hr maxima and annual means is investigated by linear regression. A limit number of measurements is available for PM_{10}, so a few stations are added in the analysis in cities that are not further included in this project. Data from Warsaw and Istanbul are not included due to their strongly deviating ratio short/long term.
The regression results based on 11 measurements read:

[28] In the final EU directive the 125 ug/m3 is allowed to be exceeded 3 times a year, as a consequences the conclusion with regard to SO_2 are to stringent.

$[PM_{10}$ max 24 hr] = 2.8[PM_{10} annual average] + 3.8 µg/m^3,

with a correlation coefficient $r^2 = 0.91$.
The annual average corresponding to the maximum 24 hr value of 50 µg/m^3 equals 17 ± 3 µg/m^3. The maximum 24 hr value corresponding to the annual mean AQG of 20 µg/m^3 equals 60 ± 12 µg/m^3. This shows that the short-term AQG is exceeded earlier than the long-term AQG, though for the 20 µg/m^3 not by far. The corresponding annual mean for the 7 and 35 allowed exceedances of the 50 µg/m^3 can be read from Figure 2.4:
20 and 29 µg/m^3 respectively.

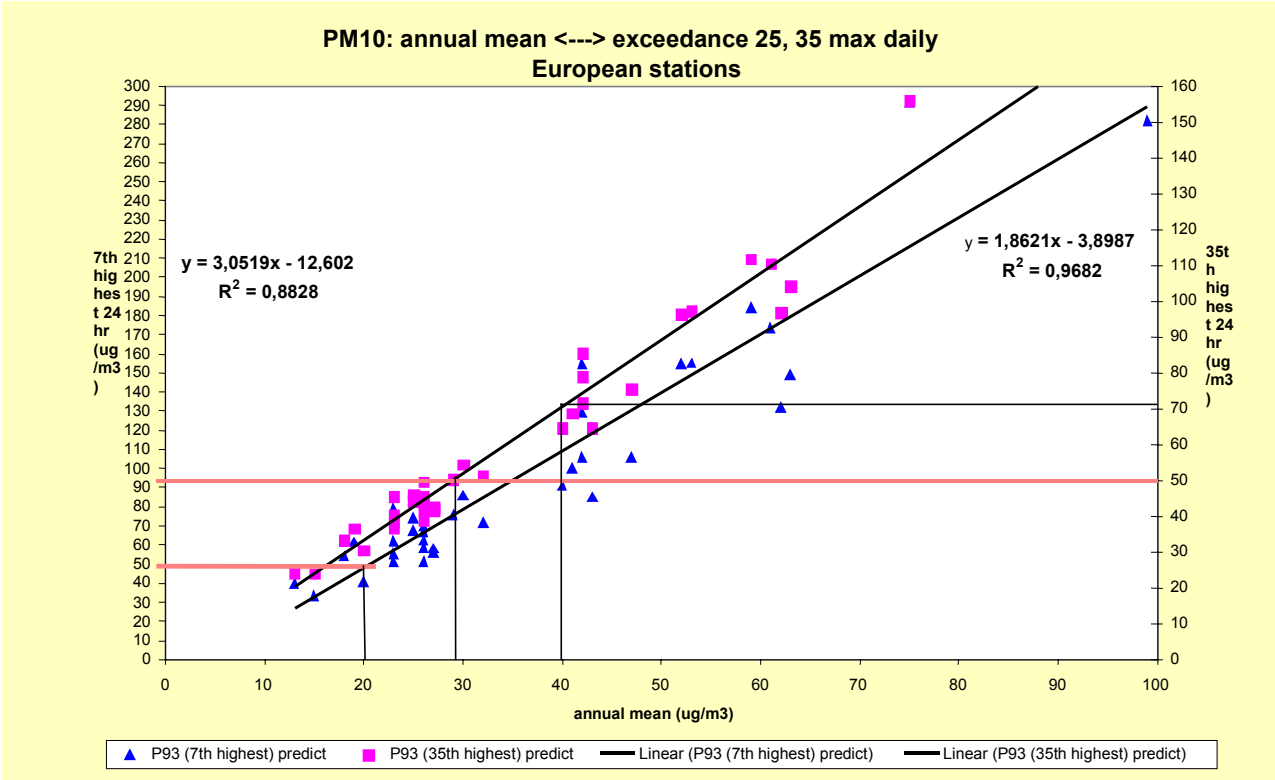

Figure 2.4: Annual average PM_{10} versus annual 24-hour 25th and 35th maximum value for a selection of all available data. The best-fit linear relation line is shown as a solid line accompanied by the parameters of the fit.

2.3 Conclusion and Results

2.3.1 Main trends

Europe is one of the most urbanised continents and today some 70% of its population (560 million) is urban (UN/ESA, 1997) Important limits to a Europe-wide assessment of the urban environment still exist.
Most of the problems are similar in urban and rural areas but they are largely more pronounced in the former because of greater population densities, attracting the services such as transport hubs and commercial services.
The downwards trend in the concentration of and exposure to air pollutants over de past 10 years is expected to continue, although exceedances of the limit values is still expected by 2010 in the baseline scenario.
Accelerated policies can reduces the concentration further but even then PM_{10} and Ozone turnout to be persistent problems wherefore the solution requires a time horizon beyond 2010.

2.3.2 Urban Air Pollution: Road Transport Takes the Lead

Air pollution on the urban scale is the source of a range of problems both within cities as well as outside as emissions from cities lead to an increase of the regional background concentration levels of many pollutants. These problems include damage to flora and fauna, decomposition of materials, buildings, historical monuments, weather and climatic changes and most important health risks mostly associated with inhalation of gases and particles. Most of these issues are not covered in this report.
Health effects which arise from exposure to air pollution can be classified as: irritation and annoyance, loss of organ functions (e.g., reduced lung capacity), morbidity and mortality. Some of these effects can be acute and reversible, while others develop gradually into irreversible chronic conditions. The respiratory system and the eyes are the main organs affected by air pollution, while systemic effects (cardiovascular problems, carcinogeneses etc.) may also be evoked. The population in rural areas is also affected, although in a lesser extent, as the urban pollution contributes to the enhancement of the regional air pollution charges.
Many historic monuments and buildings are affected by air pollutants and in particular the sulphur compounds, especially those made from marble, calcareous sandstone, or other materials susceptible to damage. Many of these objects are situated in heavily or moderately polluted areas and thus are subject to serious deterioration. Examples from the UNESCO cultural heritage list are the Acropolis in Athens, Cologne Cathedral, and whole cities, such as Cracow and Venice.
Past and present situation:
Although air quality in Europe and particularly in the large European urban areas has improved in recent decades, nearly 350 million people in the EU are still experience exceedance of the air quality target values for at least one pollutant every year and an average of 180 million people are exposed at levels above the target value for at least 6 pollutants.
There are considerable improvements acquired in the ambient concentrations of sulphur dioxide (SO_2), lead and particulates over the last decade. The main sources of SO_2 and particulates in the past being industry and energy production from coal and heavy fuels combustion, emission reduction mostly relied on the emergence of new clean energy sources and more efficient combustion technologies. In a similar way, lead levels in the atmosphere were controlled by reducing its content in the fuels.
However, the levels of so-called 'photochemical pollutants' (NO_x, NMVOC, CO and O_3) remain high in most European cities (EEA, 1998a). Exceedances of the short term WHO air quality guidelines are recorded in the majority of the large European cities. Road traffic emissions is the dominant source category for this new form of air pollution.
National and EU-level regulations aiming at automobile emissions reduction, such as introduction of catalytic converters (EC Directive 91/441/EEC), or unleaded petrol (EC Directive 85/210/EEC), resulted in considerably lower vehicle emission factors. The continuous expansion of the vehicle fleet, however, wrestles these improvements.
NO_x and NMVOC emissions, in contrast to the clear and continuous downward trend in the SO_2 emissions per capita, increase until about 1990. Their decline from that year on is much smoother than the SO_2 emission decrease. It is important noticing the SO_2 emission decrease in the ten accession countries starts much later than in the EU15+EFTA4 countries, while for NO_x and NMVOC approximately the same trend is recorded for both country groups. For NO_x in particular, the reduction of the emissions in the accession countries is faster than in EU15+EFTA4 countries as a result of the relatively more recent renewal of vehicle fleet.

In line with decreasing emissions, SO_2 concentrations exhibit decreasing trends; although many European agglomerations experience exceedances of the short-term WHO air quality guidelines for winter-smog in 1995 (EEA,1998a).
Comparing statistical monitoring results in the case of particulates is complicated since very often only the smaller particulate fraction (e.g. PM_{10}) is being sampled in view of the fact that this fraction represents with ozone the most serious threat for human health. Furthermore, particulate matter concentrations are also influenced by natural sources. Therefore, the situation in a given urban area depends on its actual geographic location and the characteristics of its land cover. A comparison of several annual average concentrations of particulate matter monitored in European cities over the last decade shows mainly downward trends (EEA, 1998c) despite the fact that, in 1995, the short term WHO air quality guidelines was exceeded in the majority of the large European cities (EEA, 1998a).
As a result of the EC Directive for the unleaded petrol coming into force, lead concentrations have dropped sharply after 1986 in the majority of the European cities. In 1995, no city experiences exceedance of the long term WHO air quality guidelines.
Controlling air pollution from road traffic (e.g. NO_x, NMVOC and indirectly O_3 levels) is identified as the single biggest and most complex issue (CEC-DGXI-The European AQ Management Project). A variety of methods include improving public transport, diverting traffic from city centres by building ring roads, reducing car use by means of parking policies or stimulating bicycle use, have been used for dealing with the problems caused by the ever increasing number of car on the roads of Europe with varying degrees of success. The result is a modest and statistically disputable downward trend in NO_x and O_3 levels from 1990 to 1995.
By 2010, expectations to improve the situation
Future trends of the main air quality indicators have been computed by the aid of the EUTREND model for the year 2010. The models results are summarised in table 2.5 Most indicators are above their target in 1990 but the policies in place and pipeline are expected to improve the situation considerably. The average exposure of inhabitants of large agglomerations in the EU to concentrations above the recommended level will go down from 180 million in 1990 to 100 million in the BL scenario for 2010, 80 million in the AP scenario for 2010 and 70 million in the TD scenario for 2010. The most significant exceedances to be expected in 2010 are particulates and ozone. For lead, SO_2 and benzene a substantial improvement can be expected. NO_2 exceedance will go down considerable but still substantial exceedances will still exists in 2010.
Additional information concerning the expected emission, concentration, exposure and exceedance can be found in the appendices E-H of this report.

Table 2.5 Expected exposure in the EU-15 to air pollutants in the period 1990-2010 under various scenario's

		EU-15				
		1990	***2010-BL***	***2010-AP-FT***	***2010-AP-NT***	***2010-TD***
Population, total		364	387	387	387	387
Population regional		166	182	182	182	182
Population cities		94	96	96	96	96
Population agglomeration		104	109	109	109	109
PM_{10}	population (millions) above target	351	306	302	296	294
ozone	population (millions) above target	330	265	180	170	110
NO_2	population (millions) above target	125	35	20		15
SO_2	population (millions) above target	170	5	5		0
Benzene population (millions) above target		85	8			2
Pb	population (millions) above target	45	0	0	0	0

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3. Benefit assessment

3.1 Chemicals and particulate matter

3.1.1 Public opinion

Chemical and heavy metal risks are considered to be a middle-ranking issue in Europe today according to average public opinion. In two Eurobarometer surveys in 1992 and 1995, they were ranked 7th and 6th in importance, respectively. This is supported by the survey in Ireland and that by the UK Department of the Environment at the time (both in 1993) which put chemical risks at 6th and 5th places in terms of importance, respectively. In 1995, the Danish rank chemical risk the highest at 4th. ISSP (1993) does not report any ranking on chemical and heavy metal risks.

3.1.2 Expert opinion

Chemical and heavy metal risks are not found in the lexicometric analysis presented in GEP et al. (1997), which is the only report on expert opinion.

3.1.3 Benefit estimation

Table 3.1.1. provides a summary of the benefit estimates for lead, cadmium, dioxins and pesticides. An indication of the level of confidence associated with the estimates is also provided.

Table 3.1.1 Summary of TD benefit estimate in 2010: € million

	Confidence
Lead: € 500 to 2700 million *Cadmium*: € 2.98 million *Dioxins:* € 58.7 million *Pesticides:* 1: € 13 to 28 million 2: € 51.2 million	Underestimate; benefit calculations consider avoided damage due to airborne chemicals only. *Lead*: issue of thresholds remains unresolved: *Pesticides*: overestimate: TD emissions assumed to fall to zero on the basis that alternatives are used. However, substitutes may be more toxic than the existing pesticides

A suitable indicator for future research in this area would be the change in total toxicity loads of chemicals.

In principle, chemical risks cover a vast array of chemical and heavy metal risks. In practice, little is known about emissions for many of these pollutants and even less about dose-response relationships. Accordingly, we focus on those pollutants where some reasonably reliable information is available. However, caution needs to be exercised when interpreting the figures since, even where information is available and thought to be reliable, considerable uncertainties remain (RIVM, 1998).

For purposes of monetary valuation the focus is on the impacts to human health from the following chemicals and heavy metals dispersed through air:

- Heavy metals: lead and cadmium
- Dioxins
- Pesticides

Methodology
Lead

Air concentrations and emissions of lead in the Baseline and TD scenarios estimated by RIVM are given in *Table 3.1.2* Overall, there is an estimated 56% reduction in lead emissions between 1990 and 2010. This is primarily due to the phasing out of leaded petrol (85/210/EEC).

Table 3.1.2 Lead concentrations in 1990 and 2010, EU-15 and wider Europe

	1990	2010BL	2010BL - 1990	2010TD	2010TD-2010 BL
Lead concentrations: $\mu g/m^3$					
EU15	2.2×10^{-2}	9.6×10^{-3}	-1.24×10^{-2}	-	-
Europe	2.1×10^{-2}	6.5×10^{-3}	-1.45×10^{-2}	-	-
Lead emissions: tonnes / year					
EU15	16,373	6,626	-9,747	4,920	-1,706
Europe	37,530	12,421	-25,109	-	-

Valuation studies on lead damage are scarce. EFTEC (1996) estimates marginal damage costs for UN ECE countries. Damages cover reductions in childhood IQ, cardiovascular effects via high blood pressure and hypertension, and neo-natal deaths. Neo-natal deaths dominate the damages, and it is worth noting that the links between airborne lead and premature birth are extremely uncertain, hence considerable uncertainty attaches to the estimates. Damage is presented in M€ per 10^{-9} grams/m^3 air lead. The range of values for EU-15 is shown in Table 3.1.3 along with the RIVM estimates of changes in airborne lead concentrations for the baseline scenario. Table 3.1.3 suggests that there are substantial benefits from reducing airborne lead, some € 14.5 billion in the terminal 2010. Since we also have data on the emission reductions giving rise to the change in concentrations (see Table 3.1.2) we are also able to estimate the economic value of a tonne of lead emitted. In EU-15, this is estimated to be € million 1.5 / tonne lead (i.e. the 'price' of 1 tonne lead emitted = 14.5×10^9 € / 9,747 tonnes = € 1.5 million per tonne lead).

Maddison et al., (1996) estimate damage costs of £280 million in 1993 in the UK from traffic-related lead. Emissions from petrol vehicles in that year were 1500 tonnes lead, suggesting an average figure of some £190,000 / tonne lead, € 280,000 per tonne (1997 prices). Using this value suggests the benefits of reducing airborne lead are some € 2.7 billion in the terminal 2010. However, this figure excludes children's IQ change and neonatal mortality, which dominate the EFTEC estimates given above.

Based on the lead concentrations given in Table 3.1.2., we present the estimates of airborne lead damage in 2010 in Table 3.1.3. Here, we compare the Baseline in 2010 with the 1990 situation. We assume no threshold values for lead health effects.

Table 3.1.3 Estimates of airborne lead damage

Country	Marginal Damage in M€ per Nanogram/m^3	Reduction in Lead Concentrations Baseline/1990. μg/m^3	Value of Benefit of Baseline Scenario Relative to 1990 M€
Austria	16	3.0×10^{-2}	-480
Belgium	23	3.7×10^{-2}	-851
Denmark	11	6.0×10^{-3}	-66
Finland	11	3.7×10^{-3}	-41
France	115	3.3×10^{-2}	-3795
Germany	163	2.0×10^{-2}	-3260
Greece	20	1.6×10^{-2}	-320
Ireland	7	4.7×10^{-3}	-33
Italy	118	2.1×10^{-2}	-2478
Luxembourg	1	3.2×10^{-2}	-32
Netherlands	31	1.7×10^{-2}	-527
Portugal	20	1.1×10^{-2}	-220
Spain	77	1.0×10^{-2}	-770
Sweden	18	5.5×10^{-3}	-99
UK	120	1.3×10^{-2}	-1560
EU15			**-14532**

Note: -ve sign indicates avoided damage to human health due to reduction in lead emissions. This is otherwise interpreted as a benefit of the Baseline scenario relative to 1990.

The use of lead dose response functions that exclude any consideration of thresholds is debatable. Data on urban exposure to airborne lead in the EU (see Dobris 1, p225) show that most major cities were below the WHO Guideline by 1990 and all were below by 1995. If the WHO Guideline value is interpreted as a 'zero risk' threshold, then damages currently should be zero. Inspection of the US EPA's survey of the dose-response functions for airborne lead (US EPA, 1999) does not show the adoption of any thresholds. In light of this, we report two sets of estimates. The first is for a risk threshold set by the WHO Guideline in which case damages, are zero. The second is for a no threshold case, in which case the damage estimates are based on lead emissions (i.e. tonnes) multiplied by low / high unit damage values for a tonne of lead (i.e. € million 1.5 - 0.28 per tonne lead). The results of which are shown in Table 3.1.4.

For comparison, US EPA (1999) estimates suggest that annual avoided effects in 1990 because of the control of lead under the US Clean Air Act amounted to some $166 billion. Comparing policy benefits for lead reductions between 1970 and 1990 (the US case) and 1990 and 2010 (this study) is hazardous.

Table 3.1.4 Benefit estimates of avoided airborne lead damage: € billion

	1990	2010 BL	2010 TD	Benefit of TD over baseline in 2010
EU 15	24.60 - 4.60	10.60 - 1.90	7.87 - 1.47	2.73 - 0.51

Note: the results assume no threshold values for lead health effects

Cadmium

RIVM (1998) estimate emissions of cadmium (Cd) in the baseline and TD scenarios, these are given in Table 3.1.5.

Table 3.1.5 Cadmium emissions in EU-15: Tonnes / year

	1990	2010 Baseline	2010 TD	Reduction in cadmium emissions by moving from Baseline to TD scenario
EU15	204	240	120	120

Marginal damages for cadmium from incinerators are estimated by Rabl et al. (1998) at € 18,300 per tonne Cd. Actual costs depend on the location of the incinerator and the height of the stack. Thus rural sites involve multiplying

the value given by 0.3 and urban sites by 3, and other multipliers are provided for stack conditions. Finally, the reference value is for an area with a population density of 105 people/km^2. Since we are seeking Europe-wide estimates, we adopt the reference value. A final complication, however, is that Rabl et al. adopt a 'value of a cancer' rather than the value of statistical life and they put this at € 1.5 million (1990 price) i.e. € 1.9 million (1997 price). In '*Stratospheric ozone depletion*' we discuss the problem of valuing cancers but the cancers there were skin cancers and these are generally not as serious as other cancers. As it happens, Viscusi (1995) has conducted a valuation experiment in which individuals trade off cancers against risk of death. This suggests that a terminal cancer would have a value equivalent to the VOSL but that a curable cancer would have a value of 0.63 x VOSL. In our case this would give the economic value of a cancer as being € 2.0 million, remarkably close to the € 1.9 million used by Rabl et al., Hence we retain their figure here. Applying the €18,300 per tonne Cd figure to the emission estimates, the results are given in Table 3.1.6.

Table 3.1.6 Benefit estimates of avoided airborne cadmium damage. € million

	1990	2010 Baseline	2010 TD	Benefit of TD scenario over Baseline in 2010
EU 15	4.75	5.96	2.98	2.98

Dioxins

Table 3.1.7 reports the estimates for dioxin emissions in the Baseline and TD scenarios.

Table 3.1.7 Dioxins TEQ emissions in EU15. Grammes / year

	1990	2010 Baseline	2010 TD	Reduction in dioxin emissions moving from Baseline to TD scenario
EU 15	6, 020	4, 181	1, 537	-2, 644

Rabl et al.,(1998) provide a reference value for damages from dioxins at €16.3 billion per tonne dioxins, reflecting the toxicity of dioxin, and again using the € 1.5 million (1990 price) value of a cancer. Applying this figure to the data above, the estimated damage values are given in Table 3.1.8.

Table 3.1.8 Benefit estimates of avoided airborne dioxin damage. € million

	1990	2010 Baseline	2010 TD	Benefit of TD scenario over Baseline
EU 15	125.0	92.8	34.1	58.7

Pesticides

RIVM Baseline scenario data for pesticides are specific to particular pesticides: Penta-chlorine-phenyl, the insecticides Lindane and Endosulfan, and the herbicide Atrazine. There are no economic studies that elicit the willingness to pay to avoid *specific* pesticides, and there are only a few studies that seek directly to value the damages from pesticides generally.

(a) Costs of control approach: Atrazine

There is a study of the costs of the ban on Atrazine in Italy (S∧derqvist, 1994). Estimates of the costs of the ban for Italy as a whole range from 50 to 100 billion lire (1990), or 60-120 billion lire in 1997 prices. The estimates are derived from micro studies of the costs of switching to substitute herbicides and from an analysis of a system of demand equations. Average Italian consumption in the mid 1980s was some 3000 tonnes in Italy, suggesting that the ban cost Italian farmers some 20-40,000 lire per kg of product, €10-21 per kg. This is not a WTP estimate except in the sense that Italian 'society', by banning Atrazine, implies this cost as a minimum willingness to pay to avoid Atrazine's side effects. In cost-benefit terms, using control costs to measure damage costs is somewhat misleading since (a) it implies that the benefit cost ratio is always 1 or greater, and (b) even if interpreted as a WTP concept it is minimum WTP and not maximum WTP as required by economic theory. Hence the estimate should be used with considerable caution.

S∧derqvist (1994) compares his estimates to WTP estimates for removing trihalomethane from drinking water in the USA (Mitchell and Carson, 1986). His per capita Italian cost comes to some 2683 lire in 1990, and Mitchell and

Carson's study estimated the equivalent dollar figure to be 2500 lire when converted at the prevailing exchange rate and expressed in 1990 prices.

Table 3.1.9 gives the emissions of atrazine in the Baseline scenario in 1990 and 2010. No additional targets are defined for AP scenarios because they are achieved in the Baseline already.

Table 3.1.9 Emissions of Atrazine in EU15. Tonnes / year

	1990	2010 BL	2010 TD	2010 BL - 1990	Reduction in atrazine emissions: moving from BL to TD
EU 15	1212	1250	0	38	1250

This study uses the estimate derived from S∧derqvist, the results are given in 3.1.10.

Table 3.1.10 Benefit estimates of airborne Atrazine damage. € million

	1990	2010 BL	2010 BL - 1990	Benefit of TD over baseline in 2010
EU 15	12.10-25.50	13.4 - 28.0	1.25 - 2.53	13.4- 28.0

Emissions are assumed to fall to zero in 2010 in the TD scenario on the basis that agricultural practices are changed or substitutes are used. It is possible that the alternatives employed are more harmful to the environment than the existing pesticides. Thus estimates of avoided damage associated with moving to the TD scenario are an overestimate.

(b) Willingness to pay approach: pesticides

The RIVM Baseline scenario can be generalised to herbicides and pesticides in general. Analysis of the baseline scenario suggests that the four chemicals in question show only slight changes in 2010 'emissions' compared to 1990 (i.e. 4.0kt in 1990 to 3.2kt in 2010 Baseline). This suggests that the Baseline scenario can be characterised by a generally constant use of chemicals as herbicides and pesticides. According to Eurostat (1995), consumption of *all* pesticides (fungicides, herbicides and insecticides) in 1990 was some 500,000 tonnes. Clearly, the range of individual pesticides is enormous and they vary substantially in degrees of toxicity. Hence, seeking a WTP to avoid 'one tonne' of pesticide involves heroic assumptions about what it is that is being valued.

Studies of WTP to avoid pesticide use are surveyed in Pearce and Tinch (1998). 'Macroeconomic' studies include those by Pimentel et al. (1992) and Steiner et al. (1995) for the USA. These are broad brush studies and are open to criticism (see Pearce and Tinch, 1998). Taking the Steiner study, annual externality costs from pesticides are put at some $1.3 to $2.6 billion. US consumption was about 430 million kg in 1990, so that a 'benchmark' figure for externality costs per kg of pesticide would be $3-6 kg, (1990 prices) or some € 3.3-6.6 per kg (1997 prices). This is slightly less than the Atrazine control cost estimate, (i.e. € 10-21 per kg) given above.

Smith (1992) estimates WTP for clean groundwater in the USA (nitrate and pesticide free) at $33 per annum per household exposed to risk. This figure does not translate easily to a national aggregate since the stock of households at risk is not known. All other US studies of WTP relate to specific products (such as grapefruits or apples) and do not lend themselves to aggregation. The only available European study estimating WTP for pesticide reductions is by Foster et al., (1998). Using a contingent ranking exercise they estimate that consumers are WTP the equivalent of some £12 per kg of pesticide to avoid pesticide residues in food. The study was conducted so that separate reasons for WTP could be elicited: concern over risks to bird life proved to be more important than concerns about health risks. Foster et al., (1998) suggest a valuation of € 15 per kg, or €15,000 per tonne of pesticide. A value similar to the mid estimate derived from S∧derqvist. Table 3.1.11 reports the emissions of four pesticides, atrazine, endosulfan, lindane and penta-chlorine-phenyl.

Table 3.1.11 Emissions of four pesticides in EU15. Tonnes / year

	Four Pesticides: tonnes / year				
	1990	2010 BL	2010 TD	2010 BL - 1990	Reduction of pesticide use due to move from Baseline to TD in 2010
EU 15	4,000	3,200	0	800	3,200

Table 3.1.12 reports the monetary estimates of avoided damage of moving from the Baseline to the TD scenario. The benefits of the TD scenario are considered to be an overestimate. This is because emissions are assumed to fall to zero on the basis that there is a change in agricultural practice and substitution of other chemicals. However, the alternative pesticides may have a similar impact on the environment as the existing chemicals.

Table 3.1.12 Benefit estimates of avoided damage due to four pesticides. € million

	1990	2010 BL	Benefit of Baseline 2010 over Baseline 1990	Benefit of TD over Baseline in 2010
EU 15	60	51.2	8.8	51.2

The analysis of the Baseline scenario suggests that the four chemicals in question show only slight changes in 2010 'emissions' compared to 1990. If we assume the Baseline scenario is characterised by a generally constant use of chemicals, herbicides and pesticides at 500,000 tonnes. This means that, applying the €15 per kg pesticide figure to EU consumption of pesticides suggests annual damage costs of € 9 billion.

Summary of results

Table 3.1.13 gives a summary of the avoided monetised impacts to human health resulting from exposure to some chemicals and pesticides, moving from the Baseline scenario to the TD scenario in 2010.

Table 3.1.13 Summary benefits of TD scenario for lead, cadmium, dioxins and pesticides. € million

Benefit in 2010 only		*assuming WHO guideline lead threshold*
Lead	500 - 2700	zero
Cadmium	2.98	
Dioxins	58.7	
Pesticides	13 - 51.2	
Total	575 - 2813	75 - 113

Note: benefits relate to reduced impacts to human health only.

3.2 Human health and air pollution

3.2.1 Public opinion

Urban stress scores a very low ranking in the national and international public opinion surveys used for this analysis. Its position varies between last position in the ISSP survey (1993) and penultimate in all others considered.

3.2.2 Expert opinion

GEP et al. (1997) report 'urban life and transport' as a significant problem. This category includes both noise and air pollution, mainly from motor vehicles and therefore seems directly applicable to this environmental issue. It ranks as the sixth most important environmental issue at the global level. Europeans rank urban stress as the eighth most important issue, with Northern Europe showing more concern than Southern Europe. Interestingly, the respondents show a decreasing concern for this environmental issue as the time horizon extends to the year 2050.

3.2.3 Benefit assessment

Table 3.2.1 gives a summary of the primary benefit estimates for the control of PM_{10} and the secondary benefits from acidification and climate change related measures. Low / high values assume premature mortality is valued with VOLY and VOSL respectively. Table 3.2.1 also gives an estimate of the current level of damage in EU15 due to noise nuisance.

Table 3.2.1 Primary benefit estimates and secondary benefits to acidification and climate change

	Benefit € billion
PM_{10}	
Accelerated Policies scenario (AP) **Primary benefit** from the direct control of primary PM_{10} **Secondary benefit** from acidification due to control of primary PM_{10} from climate change due to control of primary PM_{10} **Secondary benefit** *from acidification due to control of secondary aerosols* *from climate change due to control of secondary aerosols* **Total**	 3.1 - 5.3 0.9 - 1.5 2.6 - 4.6 3.1 - 5.3 5.7 - 9.8 15.3 - 26.4
Technology Driven scenario (TD) *total primary and secondary benefits*	 14.0 - 24.2
Noise Nuisance	
Current damage	13.2

Note that the confidence for the PM_{10} benefits estimate is moderate due to the VOSL figure, although this is modified due to age distribution of mortality incidence. Confidence for noise nuisance is low.

The two issues, PM_{10} and noise, are dealt with separately in the valuation section below. Monetary damage estimates for other pollutants such as lead are dealt with in chemical and heavy metal risks.

Suitable indicators and valuation estimates for future research into air quality and noise nuisance, are as follows:

- PM_{10}: life years at risk
- Noise: percent population annoyed

Methodology valuation of health impacts caused by PM_{10}

The benefit estimates are the avoided health damage secured by moving from the Baseline to the AP or TD scenario. The main health effects of PM_{10} are premature mortality and morbidity, measured in terms of respiratory hospital admissions (RHA), emergency room visits (ERV), restricted activity days (RAD), respiratory symptoms (RS), asthma attacks (AA) and chronic bronchitis (CB). PM_{10} also has effects on visibility, this is not included in the benefits assessment due to the lack of European valuation work in this area.

The benefit calculations are based on PM_{10} concentrations (including $PM_{2.5}$) in urban and rural areas and the average exposures of the European population, to the concentrations. The benefit estimates for PM_{10} are made up of i) primary benefits and ii) secondary benefits from acidification and climate change related measures. The primary benefits estimates are due to end of pipe measures, such as filters, which reduce emissions of primary PM_{10}. There are two sets of secondary benefits. The first set arise due to climate and acidification related measures that reduce primary PM_{10} emissions, these secondary benefit estimates are added to the overall secondary benefit estimates for climate change and acidification. The second set of secondary benefits arises due to climate and acidification related measures that reduce emissions of SO_x and NO_x and therefore reduce the concentration of aerosols, otherwise known as secondary particulate matter. The secondary benefit estimates of reduced secondary particulate matter due to acidification related measures are subsumed in the primary benefit estimates of acidification, whilst the reductions due to climate change are subsumed in the secondary benefit estimate for acidification due to climate change related measures. To avoid double counting, the secondary benefit estimates of reduced secondary aerosols due to acidification and climate related measures are reported in this Technical report as an indication of their size only, they are not added to the overall secondary benefit estimates for climate change and acidification.

It is important to note that not all particulate matter is "anthropogenic", some arises as a natural background level (e.g. dust) and is not generally subject to policy measures. This study considers the reduction of PM_{10}

concentrations in the AP and TD scenarios according to a set of 'anthropogenic' measures only. The non-anthropogenic part of PM_{10} is not affected by these measures and simply remains at the same level.

Mortality and morbidity

The absolute change in mortality from PM_{10} is obtained as follows:

$$dH_{mt} = b_{mt} \cdot dPM10 \cdot CMR \cdot POP \cdot 1/100$$

where, *d* denotes 'change in', H_{mt} is the mortality health end state, i.e. damage, b_{mt} is the exposure-response coefficient and *POP* is population. *CMR* stands for crude mortality rate where, in 1990 the number of EU deaths per 1000 population is estimated as 10.2 (Eurostat 1997) in 1990. In 1995 the crude mortality rates as 10 (Eurostat 1997). Since forecasts for this figure are lacking, we adopt the conservative estimate of 10 for 2010. The factor 100 converts *b* from a percentage to an absolute number.

Similarly, the estimated morbidity effects of PM_{10} are obtained using the following formula:

$$dH_{mb} = b_{mb} \cdot dPM10 \cdot POP$$

where H_{mb} is the measure of morbidity, b_{mb} is the exposure-response coefficient and *POP* is the population at risk of specific morbidity effects.

Population at risk

The fraction of European population in the various groups considered at special risk, such as the elderly, children and asthmatics, are taken from AEA Technology (1999). These are given as:

fraction of children in European population	0.2
fraction of adults in European population	0.8
fraction of people > 65 years in European population	0.14
child asthmatics as fraction of UK population[29]	0.02
adult asthmatics as fraction of UK population	0.04

The exposure-response coefficients required to estimate the number of cases are taken from Maddison et al. (1997) and AEA Technology (1999). The former is based on a meta-analysis of several epidemiological studies from America, South America and Europe, and reports a number of exposure response functions for PM_{10}. Since Maddison's meta-analysis included some European studies (at least for mortality), it is fairly reasonable to adopt such exposure response functions in the EU context. AEA Technology (1999) adopt exposure response relationships from the ExternE Project (European Commission, 1998). Thus two benefit estimates are calculated using the two different sets of exposure response functions. The main results are those based on Maddison et al., (1997), whilst the results derived from AEA Technology (1999) are presented in the sensitivity analysis.

The relevant exposure response coefficients are given in Table 3.2.2. The coefficient, *b*, should be interpreted here as the increase in annual incidence of each symptom. For example, 1) morbidity: the coefficient is the number of cases / year. person.$\mu g/m^3$ and 2) mortality: the coefficient is the % change in mortality rate / year .$\mu g/m^3$.

[29] The figures for asthma in the UK are assumed to apply throughout Europe, AEA Technology (1999).

Table 3.2.2 Exposure response coefficients for health end points caused by PM_{10}

		b central estimate
Maddison et al., (1997)		
Mortality:(slope at 100 µg/m^3)		0.032%
RHA	Respiratory hospital admissions: (general population)	2.94 x 10^{-6}
AA	asthma attacks: (asthmatics)	6.49 x 10^{-2}
ERV	Emergency room visits: (general population)	2.35 x 10^{-4}
RAD	Restricted activity days: (general population)	5.75 x 10^{-2}
RS	Respiratory symptoms: (adults)	0.18
CB	chronic bronchitis: (adults)	6.12 x 10^{-5}
AEA Technology (1999)		
Mortality		0.074%
BU	Bronchodilator usage: (adult asthmatics)	0.103
BU	Bronchodilator usage: (child asthmatics)	0.078
C	Cough (adult asthmatics)	0.168
C	Cough (child asthmatics)	0.133
LRS	Lower respiratory symptom (adult asthmatic)	0.061
LRS	Lower respiratory symptom (child asthmatic)	0.103
CHF	Congestive heart failure (+ 65 years)	1.85 x 10^{-5}
CB	chronic bronchitis (child)	1.61 x 10^{-3}
CB	chronic bronchitis (adults)	4.9 x 10^{-5}
CC	chronic cough (child)	2.07 x 10^{-3}
RAD	Restricted activity day (adults)	0.025
RHA	Respiratory hospital admission (general population)	2.07 x 10^{-6}
CVA	Cerebrovascular hospital admission (general population)	5.04 x 10^{-6}

Note that: Exposure response functions differ between the two sources. Importantly: AEA Technology cite exposure response functions for asthma attacks for asthmatics for tropospheric ozone only, whilst asthma attacks for the entire population due to exposure of PM_{10} are very uncertain, and are therefore not included in the 'core' set of reliable exposure response functions. The exposure response functions for emergency room visits are also considered unreliable and are also omitted from the 'core' set.

Exposure response functions for health impacts due to PM_{10} reported by AEA Technology but omitted from this study due to unreliable valuation data include chronic mortality to the general population. This requires a 'value of reduction of life expectancy, of which there is currently only one estimate Johannesson M and Johansson P O (1995a, 1996).

The benefit of the AP / TD scenario is estimated by using the above equations. Table 3.2.3 presents the average reduction in concentrations for PM_{10} in all areas due to AP and TD scenarios are. PM_{10} concentrations are split into, i) reduction of primary PM_{10} due to end-of-pipe measures, ii) reduction of primary PM_{10} due to acidification and climate change related measures (see column 3) and iii) reduction of secondary aerosols due to acidification and climate change related measures which reduce emissions of NO_x and SO_x and therefore reduce the concentration of secondary aerosols (see column 4).

Table 3.2.3 PM10 concentration reductions due to AP and TD scenarios

	Primary PM_{10} µg/m^3	Secondary Primary PM_{10} µg/m^3	Secondary secondary aerosols µg/m^3	Total PM_{10} reduction µg/m^3
AP	0.75	0.86	2.13	3.73
TD	-	-	-	3.42

Note that the PM_{10} reduction in the TD scenario is less than the AP scenario because there are no spill-over effects from climate change related measures. Due to data limitations on PM_{10} emissions and concentrations for the TD scenario it has not been possible to distribute the concentration reduction across the different sources.

Table 3.2.4 presents the relevant population data and groups of people considered at risk.

Table 3.2.4 Population at risk

	2010 million
Population	386.7
Fraction of children	77.3
Fraction of adults	309.4
Child asthmatics	7.7
Adult asthmatics	15.5
Elderly (+ 65 years)	54.1
Annual death rate per 1000 population	10

The recommended value of statistical life is € 2.6 million (1990 prices), this is € 3.31 million in 1997 prices. However, as discussed in Maddison et al., (1997) pollution-related mortality affects largely the elderly (over 85% of premature deaths are in the over 65 group). There is some evidence that values of risk aversion are lower for this age group at around 70% of the prevailing risk values (see Pearce 1998). This reduces the VOSL to € 2.32 million (1997 prices).

Two sets of values for morbidity are reported in *Table 3.2.5*. The first are based on the values for morbidity in the US reported in Pearce et al., (1996). These are converted to € 1997 values. Values are then adjusted for the European Union by reducing them by 40% as suggested by Navrud (1997)[30]. The second set are taken from AEA Technology (1999), they use values based on work by Markandya (European Commission, 1998). These values are converted to € 1997 prices.

Table 3.2.5 Unit values for morbidity effects

Morbidity effects	Estimation method	Unit value € 1997[31]	Adjusted for EU unit value € 1997
Pearce et al., (1996)			
RHAs ERV RADs AA RS CB	adjusted COI adjusted COI WTP + adjusted COI WTP WTP WTP	13,206 497 68.5 31 9 200,000	7,924 298 41 19 5 120,000
Values based on European Commission (1998)			
BU	as LRS	8	-
C	as LRS	8	-
LRS	WTP	8	-
CHF	Assume as RAD	562	
CB	Rowe et al., (1995)	112112	-
CC	-	240	-
RAD	WTP	80	-
RHA	Assume as RAD	803	-
CVA	Assume as RAD	3612	-

Note that for values based on European Commission (1998), all days in hospital for RHA, CHF and CVA are also restricted activity days (RAD). Assume that the average stay for each is 10, 7 and 45 days respectively.

[30] Navrud estimates some WTP values for respiratory diseases, that are in general not comparable with the exposure response functions in Table 2.2.2. However, Navrud reports a WTP value to avoid asthma, the figure obtained by the Norwegian contingent valuation study is the same as the adjusted value reported *in Table 2.2.2.*

[31] The values are standardised as follows: all converted to 1997 values using 3% inflation. US values multiplied by the ratio of income per capita USA 1997 to EU average income. UK values multiplied by the ratio of UK income per capita to average EU incomes. Norwegian values multiplied by ratio of Norwegian incomes to EU average. Per capita GNP from Europe in Figures, ch 16, purchasing power parity estimates. Norwegian per capita GNP from World Bank, World Development Report 1997, purchasing power parity estimates.

Two benefit estimates are calculated. The first set are based on the exposure response functions from Maddison et al., (1997) combined with the adjusted unit damage values based on Pearce et al., (1996). The results are reported in Table 3.2.6. The second set are based on the exposure response functions from AEA Technology (1999) combined with the second set of morbidity values from European Commission (1998). The benefit estimates for the AP and TD scenario are reported in the sensitivity analysis.

Table 3.2.6. Benefit of AP and TD scenarios

	Primary benefit € billion	Secondary benefit due to primary PM_{10} € billion	Secondary benefit due to secondary aerosols € billion	Total benefit € billion
AP				
Mortality	0.1 - 2.4	*0.2 - 2.7*	*0.4 - 6.7*	0.7 - 11.8
Morbidity	3.0	*3.4*	*8.3*	14.6
Total	3.1 - 5.3	*3.6 - 6.1*	*8.7 - 15.0*	15.3 - 26.4
TD				
Mortality	-	-	-	0.7 - 10.8
Morbidity	-	-	-	13.3
Total	-	-	-	14.0 - 24.1

Low / high estimates assume premature mortality is valued with VOLY, VOSL respectively.

Note that figures in italic represent secondary benefits from climate change and acidification related measures. These values are not included in the B/C analysis of PM_{10} control. Instead, the secondary benefits due to primary PM_{10} are allocated to climate change and acidification based on the distribution in terms of emission reduction, i.e. 75% is due to climate change and 25% due to acidification. They are added to the secondary benefit estimates of climate change and acidification respectively. The secondary benefits due to the control of secondary aerosols from acidification and climate related measures are already subsumed in the primary benefits estimate for acidification and the secondary benefits of climate to acidification respectively. Values are reported here as an indication of their size only.

In the AP scenario, the primary benefits due to end-of-pipe measures to control primary PM_{10} amount to roughly € 3 to 5 billion. The benefit estimates for the TD scenario (€ 14 to 24 billion) are a combination of i) primary benefits due to end-of-pipe measures to reduce primary PM_{10}, ii) secondary benefits from acidification and climate change related measures that reduce primary PM_{10} and iii) secondary benefits from acidification and climate change related measures that reduce secondary aerosol.

3.2.4 Benefit assessment noise nuisance

Unit values for noise nuisance

The environmental assessments in this study do not estimate noise exposure by scenario but, given the policy relevance of noise nuisance, we explore some of the possible damage costs.

Maddison et al., (1996) set out a methodology for evaluating the monetary value cost of noise nuisance. The essential calculation is:

Noise cost per annum = (POP/HHOLDSIZE).HPRICE.NDSI (dB - 55).A

Where: POP is the population exposed to noise in a given noise band, HHOLDSIZE is the household size in persons / household. HPRICE is the average house price, NDSI is the noise depreciation sensitivity index, i.e. percentage depreciation in price for each decibel (dB) of noise above the baseline level. dB is the actual decibel exposure level, 55 is 55dB, the assumed baseline noise exposure level below which NDSI = 0 and A is the annuitisation factor.

As an example, Maddison et al., (1996) estimate that 17.2 million people in the UK are exposed to noise levels in the 55-60 dB range. At 2.4 persons per household (Eurostat, 1995) this corresponds to 7.1 million households. Average house prices are put at £61,000 and the noise depreciation sensitivity index (NDSI) is taken from a meta-analysis of hedonic house price depreciation studies and is NDSI = 0.0067. Average exceedance above the baseline noise level is then (60-55)/2 = 2.5. This gives a total capital loss of £7.3 billion which, at 8% annuity, is £0.584 billion for this noise band. Note that the baseline level of 55 dBA is probably high. Tinch (1995) adopts a baseline of 50 dBA

which adds around £400 million to the total costs in the UK. Tinch's results are otherwise entirely consistent with those of Maddison et al.,.

Maddison et al., (1996) use an NDSI of 0.67% which is based on a review by CSERGE/EFTEC (1994). With some exceptions, other studies not in this meta-analysis lend support to this value of the NDSI - see Table 3.2.7.

Table 3.2.7 Recent Noise Valuation Studies in Europe

Study	Country	NDSI: hedonic	NDSI:CVM
Vainio (1995)	Finland	0.35%	0.7-1.05
Weinberger et al., (1991)	Germany	0.5-1.3%	20 DM/pp/pa/pdB
Collins and Evans (1994)	UK	0.65-1.28%	-
Soguel (1994)	Switzerland	0.91%	23 SWFR pp/pa/pdB
Pommerehne (1987)	Switzerland	1.26%	-
Saelensminde and Hammer (1994), Saelensminde(1997)	Norway	-	225-400NOKpp/pa/dB
Grue et al.,. (1997)	Norway France	0.24-0.54	-
Lambert (1992)		1.0%	-

Bertrand (1997) conducts a meta-analysis of noise valuation studies and finds a fitted equation of:

$$MWTP = e^{2.348+0.00000509Y + 0.0497N}$$

where MWTP is marginal willingness to pay, Y is income and N is noise in dB. Given the mean values for Y and N (50,348 US$, 1993 values and 71.8 dB) this gives an income elasticity of MWTP of 0.26 and an elasticity of MWTP with respect to noise of 3.57.

Population exposure to noise nuisance by dB bans

Data on population exposure to traffic, airport and railway noise are provided in OECD's *Environmental Data Compendium 1993*. The coverage is incomplete and data comes from various years. Mauch and Rothengatter (1994) update the OECD data from other sources and via interpolation of values. We take the Mauch / Rothengatter data and convert them to numbers of *households* exposed to noise, as opposed to people, by dividing by estimated household size in 1990 (Eurostat, 1995, Table 26.1). We also take all forms of noise - road, rail and aircraft. Traffic noise tends to dominate but aircraft noise exposure in the Netherlands and the UK is important. Table 3.2.8 provides the adjusted data, along with the household size data and figures for average values of the housing stock.

Table 3.2.8 Households exposed to combined road, rail and aircraft noise, c1990. Millions.

Country	55-60dBA	60-65dBA	65-70dBA	70-75dBA	75dBA +	H'hold size	€ value per house
Austria	0.29	0.22	0.35	0.13	0.05	2.4	36685
Belgium	1.34	1.16	0.47	0.06	0.02	2.6	42288
Denmark	0.32	0.22	0.20	0.04	0.02	2.2	38775
Finland	0.25	0.16	0.12	0.03	0.01	2.4	45645
France	5.16	4.01	2.60	1.16	0.45	2.5	41964
Germany	7.99	6.21	3.37	1.62	0.39	2.3	34901
Greece	0.59	0.38	0.24	0.05	0.02	2.7	16318
Ireland	0.23	0.15	0.08	0.03	neg	3.8	34735
Italy	5.17	3.48	1.94	0.60	0.18	3.0	46535
Luxemb'g	0.03	0.02	0.01	neg	0	2.7	46744
Netherlands	3.88	1.89	0.33	0.12	0.05	2.3	33401
Portugal	0.84	0.55	0.30	1.00	0.03	2.9	13933
Spain	3.69	2.41	1.33	0.42	0.14	2.5	26276
Sweden	0.52	0.26	0.15	0.03	neg	2.2	39872
UK	7.73	4.20	1.90	0.22	0.24	2.6	37024
EU-15	**38.03**	**25.32**	**13.39**	**5.51**	**1.58**	2.6	**36314**
Norway	0.26	0.20	0.11	0.04	0.01	2.4	40732

Source: noise exposure data from Mauch and Rothengatter (1994); household size and average house values from Eurostat (1995). Values relate to 1990 prices.

To calculate the *capital value* of noise damage in 1990 we need to multiply average noise exceedance above 55 dBA in each noise band, by the relevant house price, and by the depreciation factor 0.0067.

Noise damage: an example calculation

As an example, we consider Austria. Table 3.2.9. gives an illustration of noise cost estimates

Table 3.2.9. Noise cost estimates for Austria: an illustration

Noise band dBA	Average X Exceed dBA	No H'holds X 10^6	0.0067 X	Av price €	= € million
55-60	2.5	0.29	0.0067	36,685	178.2
60-65	7.5	0.22	0.0067	36,685	404.6
65-70	12.5	0.35	0.0067	36,685	1075.3
70-75	17.5	0.13	0.0067	36,685	559.2
75+	22.5	0.05	0.0067	36,685	276.5
Total noise cost (capital values)					2493.8

The total cost is a capital cost which, needs to be annuitised to secure annual damage costs. We choose a 6% real annuity rate to reflect actual interest rates in the housing market and a lifetime of 30 years. The annuitisation factor is then 13.76. Annual noise costs for Austria are then 2493.8/13.76 = € 181 million.

The results for the EU as a whole are shown in Table 3.2.10. The results suggest that, currently, noise damage amounts to some € 13 billion per annum in the EU. This is probably an understatement because of the house price data.

Comparison with other studies for single countries suggests that the actual house prices should be higher than those shown in Table 3.2.10. For example, Maddison et al. (1996) suggest that road traffic noise costs in the UK are some £2.6 billion p.a, or some € 3.1 billion. The estimates in Table 3.2.10 here include aircraft and rail noise costs, so that the Table 3.2.10 estimates should be lowered by about 10% to correct for the difference in coverage, this brings the UK estimate to some € 1.35 billion. The damages here are therefore well below those reported in Maddison et al., However, the average house price used in Maddison et al., is some £61,000 = € 73,000, more than twice the Eurostat figure used in this analysis. If the Eurostat figure is used in place of the £61,000, the effect is to reduce the Maddison road traffic noise damage costs to € 1.55 billion, which is close to the estimate produced here.

Johansson (1996) reports estimates for Sweden of some SEK billion 2.6 per year for road traffic only, or some € 310 million per annum, around three times the total mobile source noise figure for Sweden, shown here. However, Johansson notes that the Swedish estimates are based on an early hedonic price study and may be unreliable.

Verhoef (1996) reports Dutch estimates of some Dfl million 660 for road and rail noise, or some € 300 million, which is above the figure estimated here.

Table 3.2.10 Annual noise damage costs in the EU

Country	Household-weighted exceedances = Σ m.hholds$_i$ x exceedance$_i$	Depreciation factor x av.house price €	Total damage cost annuitised at 6% = capital value of damage cost/13.76 € million p.a
Austria	10.15	255.9	189
Belgium	19.42	302.7	427
Denmark	5.47	259.8	103
Finland	4.07	305.8	90
France	105.91	281.2	2164
Germany	145.80	233.8	2478
Greece	8.65	109.3	69
Ireland	3.23	232.7	55
Italy	77.83	311.8	1764
Luxemb'g	0.35	313.2	8
Netherlands	31.23	223.8	508
Portugal	28.2	93.4	191
Spain	54.44	176.0	696
Sweden	5.66	267.1	110
UK	83.83	248.1	1511
EU-15			10363 (1990 prices) **13202 (1997 prices)**
Norway	4.46	272.9	88

The estimates of noise damage given here are markedly less than those in Mauch and Rothengatter (1994) this is because the calculations are based on household exposure rather than numbers of people exposed to noise.

Sensitivity analysis

Assumptions have been made throughout this analysis. Some may have a significant effect on the results, while others will only make a minor difference. This section examines what happens to the benefit estimates if the assumptions are changed.

Column 1 in Table 3.2.11 presents the current assumptions used in the analysis of urban stress and the results achieved with these assumptions are given in column 2. Changes in these assumptions are given in column 3 and the quantitative effects are given in column 4.

Table 3.2.11 Key assumptions and estimated results of changing these assumptions

Current Assumption	Current value € billion	Revised assumption	Revised value € billion
based on Maddison (1997): VOSL		Based on Maddison (1997)VOLY	
AP		AP	
mortality only	11.8	*mortality only*	0.7
mortality and morbidity	26.4	*mortality and morbidity*	15.3
TD		TD	
mortality only	10.8	*mortality only*	0.6
mortality and morbidity	24.2	*mortality and morbidity*	14.0
VOSL / VOLY adjusted for rising relative price linked to income (i.e. 0.5% p.a)		VOSL / VOLY unadjusted	
AP		AP	
mortality only	0.7- 11.8	*mortality only*	0.6 - 10.7
mortality and morbidity	15.3 - 26.4	*mortality and morbidity*	13.8 - 23.9
TD		TD	
mortality only	0.6 - 10.8	*Mortality only*	0.6 - 9.8
mortality and morbidity	14.0 - 24.2	*mortality and morbidity*	12.7 - 21.9
Maddison (1997) ERFs and Pearce et al. (1996) unit damage values, VOLY / VOSL		AEA Technology, (1999), VOLY / VOSL	
AP		AP	
Mortality	0.7- 11.8	*Mortality*	1.6 - 27.3
mortality and morbidity	15.3 - 26.4	*mortality and morbidity*	69.5 - 95.1
TD		TD	
Mortality	0.6 - 10.8	*Mortality*	1.5 - 25.1
mortality and morbidity	14.0 - 24.2	*mortality and morbidity*	63.7 - 87.2
Maddison (1997) ERFs and Pearce et al. (1996) unit damage values, VOSL - VOLY: unadjusted		AEA Technology, (1999), VOSL Unadjusted	
AP		AP	
Mortality	0.6 - 10.7	*Mortality*	1.5 - 24.7
mortality and morbidity	13.8 - 23.9	*mortality and morbidity*	62.9 - 86.1
TD		TD	
Mortality	0.6 - 9.8	*Mortality*	1.4 - 22.7
mortality and morbidity	12.7 - 21.9	*mortality and morbidity*	57.6 - 79.0

4. Policy package

4.1 Chemicals and particulate matter

4.1.1 Key issues

Lead: the issue of thresholds is fundamental to the benefit analysis of lead emissions reduction. This issue is not resolved in the literature. Assuming WHO thresholds reduces the benefits of further lead control to zero. However, assuming no thresholds suggests the benefits of further control of lead would be substantial at € 0.5 to 2.7 billion.

Cadmium: the benefits of further cadmium control appear to be modest at € 3 million.

Dioxins: despite notoriety as a serious pollutant. Available dose response functions suggest the benefits of further control of dioxins is moderate at about € 58 million.

Pesticides: the key issue is the continued use of highly toxic, persistent pesticides in agriculture.

4.1.2 Recommended key actions

Pesticides tax

If the demand for pesticides is inelastic, a tax on pesticides will tend to be revenue raising rather than 'externality' reducing. While such a tax may therefore appear self-defeating from an environmental standpoint, damage from pesticides is cumulative so that current damage is partly a function of past pesticides releases. This will be especially true of water contamination by pesticides. If revenues can be hypothecated, they can be used for groundwater clean-up programmes, so that revenue-raising taxes nonetheless have an externality reducing function.

But other damage is related to current pesticides releases, notably any biological diversity loss due to current 'doses' of pesticides to ecosystems. On the face of it, pesticide taxes do not address such externalities. However, it is widely suggested that if the tax can be differentiated by the toxicity of the pesticides in question, then, while the overall demand for pesticides is not reduced significantly, substitution between pesticides will occur in such a way that the overall toxic impact of pesticides will be reduced. In short, pesticide use and toxicity could be 'de-coupled' by a pesticide tax. The problem with pesticide tax studies is that few of them simulate the 'cross price effects' of such a policy, i.e. they do not look closely at substitution between types of pesticides (or between pesticides and other inputs such as fertilisers and land).

Bailey and Rapsomanikis (1999), simulate such a toxicity-weighted tax for the UK in the period 1992-1998. Overall price elasticity of demand for pesticides was consistently low and never greater than –0.39. However, cross price elasticities between the 'banded' pesticides (banded according to toxicity) were greater than the 'own' price elasticities, suggesting that farmers might switch between types of pesticide. But faced with a large tax on highly toxic pesticides, taxes on medium to low toxicity pesticides have a greater effect, suggesting that the high toxicity pesticides are more 'necessary' for agricultural production. A toxicity weighted tax may not therefore achieve much by way of reduction in high toxicity pesticides. Moreover, Bailey and Rapsomanikis suggest that significant pesticides taxes could be welfare reducing unless the size of the externality associated with pesticides is above some € 40 per kilogramme.

Using pesticide taxes to control pesticide damage is not therefore an unequivocally sound policy measure. Low aggregate elasticity of demand means that overall pesticide use will not be reduced much. High revenues could, however, be used for clean up of cumulative pesticide contamination in groundwater, if hypothecation of taxes is accepted. The problem of damage related to current pesticide use would remain, and a toxicity-weight tax should,

in principle, achieve the result of de-coupling toxicity from pesticide use. Available evidence, however, is not supportive of this view, although very few studies have been conducted which enlightens the association. Finally, damage from pesticides would have to be high, at around € 40,000 per tonne pesticide to justify tax measures generally. The only available estimate, also for the UK, is of damage equal to €20,000 per tonne (Foster et al., 1998), i.e. only half that required to justify a toxicity-weighted tax in economic terms.

4.1.3 Multiple benefits

A pesticide tax that leads to reduced toxicity levels for pesticides can also benefit the issue of water quality management. Whilst, a number of policy options recommended for other environmental issues will reduce the issue of chemical risks. In particular those policies that aim to reduce particulate matter emissions, such as: i) transport policies, ii) carbon / energy tax, iii) acidifying pollutant emissions tax, iv) agricultural policy reform, v) owner liability against oil spills.

4.2 Human health and air pollution

4.2.1 Key issues

The issue of urban stress is controlled mainly through policy initiatives recommended for the transport sector. All policies designed to reduce vehicle use will be beneficial in terms of PM_{10} emission reductions and noise reductions, an example of synergistic policies. These include traffic demand management schemes such as; congestion tolls at peak times (a form of road user pricing), speed restrictions, parking charges, fuel taxation and vehicle taxation.

Policies implemented to reduce other environmental issues, such as the carbon / energy tax, minimum excise duty and the aviation tax (see *Technical report on climate change*) and the nitrogen and sulphur taxes *(Technical report on acidification and eutrophication*) will also be beneficial in terms of PM_{10} reduction.

4.2.2 Recommended policy initiatives

Noise specific measures

Noise damage is estimated to be very high at € 13 billion p.a. for EU15, in 1990. This suggests noise abatement should be a high priority. Noise-specific measures include:

- general incentives to switch from private to public transport
- vehicle noise regulations as part of annual fleet test

In terms of numbers of people exposed to a given noise level, most noise nuisance in the EU is generated by road traffic. A noise-specific measure would be[32] taxes on vehicle type according to noise levels. The practicalities of a differentiated noise tax are uncertain, however.

Since noise emission limits for vehicles are already part of EC Directives on noise, the issue arises as to whether they can be strengthened in combination with some form of market based instrument. Past experience in the Netherlands suggests that noise charges on road vehicles may be difficult to sustain for institutional and legal reasons (OECD, 1991). It is unclear from the documentation what the nature of the legal difficulties was.

Policy therefore can take two forms:

(a) an explicit noise tax on road vehicles according to their noise levels. This will be easier where road vehicle noise levels are tested as part of the annual safety and emission tests. In the absence of annual noise measurements, proxies for noise levels might be age and type of vehicle (refer to Transport Policy Package);

(b) a noise tax that is built into existing fuel taxes. ECMT (1998) suggest that external noise costs be part of a general fuel tax to reflect overall externalities. They estimate such taxes for passenger cars would be around

[32] The use of low-noise road surfaces and physical barriers can also reduce road traffic noise

€ 0.6 per litre of gasoline, € 0.8 per litre for diesel, € 0.5 per litre for LPG and € 0.6 per litre of fuel for trucks and vans. Of these sums, noise taxation would account for about 5% of the tax. Taxing noise via a fuel tax is practical, although there have to be doubts about the extent to which such taxes do approximate marginal noise damages in monetary terms. However, note that such a differentiated tax can help the switch from more polluting fuels like diesel to less polluting ones like gasoline (See Transport Policy Package).

Nonetheless, various attempts to devise noise taxes have not been particularly successful. In some cases legal objections have been raised (in the context of aircraft taxes in France). In others, tax formulae have been complex or the tax has been extremely difficult to administer - see the experience of the Netherlands (OECD, 1991). In other cases, considerable success has been achieved with financial subsidies for the purchase of quieter vehicles. So long as such subsidies appear as differentiated vehicle user taxes, this incentive is acceptable. Otherwise it offends the polluter pays principle which is at the heart of EU environmental policy.

Noise tax on aircraft[33]

An effective instrument to reduce aircraft noise would be an aviation noise tax differentiated by aircraft type and location (Pearce and Pearce 1999).

The relevant tax for noise nuisance is equal to the marginal environmental damage from aircraft. A noise tax can be taxed directly through the landing fee system and such charges exist at a number of EU airports, but their relationship to marginal damages is tenuous. Existing noise charges tend to be related to revenue targets, which in turn are based on expenditures needed to insulate surrounding properties. To obtain a tax more closely related to actual damage done, an estimate of total damages per aircraft per airport needs to be estimated.

The aviation tax suggested in *Technical Report on Climate Change*, based on Pearce (1999), includes a noise tax set at marginal environmental damage from aircraft at Heathrow airport. The economic value of noise nuisance is derived from a meta-analysis of hedonic house price studies, producing an index which links house price depreciation to a unit of noise. The resulting economic values therefore vary with the level of house prices, and housing density in the surrounding noise 'footprint'.

Pearce (1999) gives the noise externality due to one aircraft movement as:

T = HPD / 365. MQN/100

Where HPD is the annual price depreciation on houses due to noise and MQN is the marginal change in noise arising from the addition or removal of one aircraft of a particular type. HPD reflects the marginal willingness to pay for aircraft noise reduction as reflected in the differential house prices around airports. This is an application of the hedonic property price methodology for placing an economical value of environmental detriment (Freeman, 1993) and a review is given in Pearce and Pearce (1999). Pearce (1999) estimates HPD for Heathrow to be £37.415 million per annum (€57,044 million per annum). MQN is estimated for a number of different aircraft. A noise tax for different aircraft at Heathrow is calculated using the above equation. A selection of aircraft noise taxes is reported in Table 4.*2*.1

Table 4.2.1 Noise tax for selected aircraft

Aircraft type	€
A310	88
B737-100	310
B747-400	230
B767-300	75
MD82	67

Note the substantial difference between taxes for the aircraft involved in most of the movements at Heathrow (B737, MD82). In general the noise component is roughly 18% of the total aviation tax recommended in Action 2.3, for details refer to *Technical report on climate change.* Noise 'taxes' need not be paid but could be used to help allocate airlines' routes, to identify the least environmental cost flight paths.

[33] See Technical background report 11 on climate change, policy package section: aviation tax.

4.2.3 Multiple benefits

As mentioned earlier urban stress will benefit from a number of policy options recommended for other environmental issues. Policies that reduce energy demand derived from fossil fuels will have the greatest benefit to urban stress. I.e. i) transport policy package, ii) carbon / energy tax, minimum excise duty, aviation tax (i.e. noise reduction), iii) acidifying pollutant emissions taxes.

4.3 Policy assessment chemical risks and particulate matter

The main area of concern within the broader problem of chemical risks is the use of highly toxic and persistent pesticides in agriculture, we therefore suggest that the main policy initiative is a pesticide tax based on toxicity ratings.

4.3.1 Causal criterion

Table 4.2.2 lays out the driving forces behind the chemical risk problem, the underlying causes of these driving forces are also identified.

Table 4.2.2. Driving forces and underlying causes of the chemical risk problem

	Driving force	Underlying cause		
		MF	IntF	ImpF
D1	Growth of chemical industry			
D2	Growth of chlor-alkali, metallurgical, non-ferrous and metal processing industry			
D3	Use of persistent pesticides in agriculture	X	X	
D4	Growth of waste incineration capacity to replace landfill		X	

X = main underlying cause, MF = market failure, IntF = intervention failure, ImpF = implementation failure. Note that for driving forces D1 and D2 the main causes are growth in real income.

A pesticide tax will only address the underlying causes if it is possible to differentiate the tax according to the toxicity of chemicals. Otherwise the impact on pesticide use will be very low due to the very inelastic elasticities of demand. New research shows that although the own price elasticity of demand for pesticides in the aggregate is low, the cross price elasticities are greater than the 'own' price elasticities, suggesting that farmers might switch between types of pesticide. This means that while a tax would not have a significant overall effect on pesticide use it could give significant switches between pesticides. Such that the overall toxicity of pesticide use will be lowered.

4.3.2 Efficiency criterion

Limited data availability permits the estimation of a B/C ratio for the control of dioxins only. The B/C ratio for the Technology Driven scenario for dioxin control is estimated at 0.3 (from € million 58.7 / 210).

Benefit assessment of TD scenario

The benefits of TD scenario over the baseline are given in *Table 4.3.1*.

Table 4.3.1 Benefit of TD scenario for lead, cadmium, dioxins and pesticides. € million

Lead	Zero[34] or 500 - 2700
Cadmium	2.98
Dioxins	58.7
Pesticides	13 - 28, 51

The benefit values are underestimates because they consider avoided damage due to airborne chemicals only.

[34] where all major cities are below the WHO lead threshold guideline value and this threshold is interpreted as a 'zero risk' threshold.

Costs of TD scenario

Table 4.3.2 presents the welfare costs associated with the TD scenario for dioxins and PAH only.

Table 4.3.2 Welfare costs for dioxin and PAH control

Additional costs (costs TD 2010 - Baseline costs 2010)	€ million
Dioxins and PAH	210

Public opinion

Eurobarometer (1995) indicates that most Europeans are prepared to change their consumption behaviour as a step to slow down or perhaps even stop the deterioration in the environment as a whole. Although public opinion regarding the issue of 'chemical risks' and more specifically 'pesticide use', is not known with certainty, the findings of Eurobaromter (1995) suggest that the European population may be in favour of measures to control pesticide use. This is supported by the continuing rise in demand for organic produced.

4.3.3 Administrative complexity

Administrative complexity for the pesticide tax is low because it can be introduced as a product tax, however it must be varied according to toxicity.

4.3.4 Equity criterion

The distributional incidence of the pesticide tax is not known, however, the occupational group will be affected (i.e. agricultural sector). In the long run the pesticide tax will affect the price of food. As a proportion of individual's income it is expected to be negligible.

4.3.5 Jurisdictional criterion

The environmental damage due to pesticide use is both a local and transboundary issue. The existing POPS Protocol suggests it is a centralised issue even though a major part of the damages are likely to be locally caused and suffered. As with many taxes, there will be concern that a tax introduced in any one state could result in cross-border purchases, impairing the Single Market. The two circumstances where this is not likely to occur are (a) when the tax is lower than the transactions and transport costs of moving across borders to effect purchases, and (b) when the tax is set uniformly in the EU. From (a) it follows that the higher the tax relative to the price of the product, the more incentive there will be for cross-border purchases. As shown above, it would appear that any tax needs to be large if it is to have significant environmental effects. The issue of uniform rates is therefore an important one in the context of pesticide taxes.

Pesticides are internationally traded, however, a low affect on competitiveness is expected.

Macroeconomic effect

Details of the macroeconomic effects are given in *Technical Report on Socio-Economic Trends, Macro-Economic Impacts and Cost Interface.*

5. Policy assessment Human health and air pollution

The issue of urban stress is controlled mainly through policy initiatives recommended for the transport sector. All policies designed to reduce vehicle use will be beneficial in terms of PM_{10} emission reductions and noise reductions.

Policies implemented to reduce other environmental issues, such as the carbon / energy tax, minimum excise duty and the aviation tax (see climate change) and the nitrogen and sulphur taxes (see acidification) will also be beneficial in terms of PM_{10} reduction.

Noise-specific measures, however, would include a noise tax on cars and a noise tax on aircraft (see *Technical report on climate change*).

5.1 Causal criterion

The underlying causes of urban stress are addressed through the transport policy packages and the noise-specific taxes.

Table 4.3.3 presents the driving forces behind the urban stress problem, the underlying causes are also identified.

Table 4.3.3 Underlying causes and driving forces of the urban stress problem

		Underlying cause		
		MF	InF	ImpF
D1	Industrial and traffic growth	X		
D2	Space heating and electricity generation	X		
D3	Migration to urban areas, partly due to income difference rural and urban areas		X	
D4	Lack of infrastructure in urban areas (e.g. sewage treatment, roads)			X
D5	Decentralisation of economic activities			

X = main underlying cause, MF = market failure, InfF = intervention failure, ImpF = implementation failure. Note that for D1 population growth plays a significant role, and real income growth contributes to D1, D2, D3 and D5.

The suggested policies targeted at traffic, energy production and aviation all target the underlying causes of 'urban stress'.

5.2 Efficiency criterion

Benefit cost ratio for TD / AP scenario

This section compares the primary benefits with the welfare costs and separately the direct costs for the control of PM_{10}. *Table 4.4.1* presents the B/C ratios for the AP scenario.

Table 4.4.1 B/C ratios for the control of PM_{10}: AP scenario only

	Welfare		Direct	
AP: PM_{10}	low	High	low	high
VOSL	5.5	19.8	3.3	11.9
VOLY	3.2	14.5	1.9	8.7

B/C ratios are presented based on VOSL and VOLY. Low B/C ratios assume, PM_{10} / health relationships from Maddison et al. (1997) and unit damage values from Pearce et al. (1996), whilst high B/C ratios assume PM_{10} /

health relationships and values from AEA Technology (1999). The B/C ratios demonstrate that the control of primary PM_{10} passes the cost benefit test for the AP scenario for both cases.

B/C ratios for the TD scenario are 18.6 - 67.1, where premature mortality is valued with VOSL and 12.8 - 58.2, where premature mortality is valued with VOLY. The benefits for the TD scenario are a combination of primary benefits of PM_{10} control and secondary benefits from acidification and climate change related measures. This suggests the B/C ratios may be overestimated.

Scenarios for noise control were not established in this study. However, an estimate of total damage due to noise pollution in urban areas is given below.

Benefit assessment of AP / TD scenario

The benefits of AP and TD scenarios over the baseline are given in *Table 4.4.2*. Benefits are measured as avoided premature mortality and morbidity incidences. Primary benefits for the AP scenario are reported as well as the combined primary and secondary benefits from acidification and climate change related measures. Only the combined primary and secondary benefit estimate is known for the TD scenario. Benefit estimates are reported, where premature mortality is valued with VOLY or VOSL. Low benefit estimates are based on Maddison et al. (1997) exposure response functions and Pearce et al. (1996) unit damage values, whilst upper values assume AEA Technology (1999) PM_{10} health relationships and values.

Table 4.4.2 Primary benefit of AP, primary and secondary benefit of the TD scenarios in 2010

	€ billion	
	VOLY	VOSL
PM_{10}		
AP		
Primary benefit only	3.1 - 13.9	5.3 - 19.0
Primary and secondary benefit	15.3 - 69.5	26.4 - 95.1
TD		
Primary and secondary benefit	14.0 - 63.7	24.2 - 87.2
Noise		
damage due to noise pollution from traffic, airport and railway noise	13.2	

Costs of TD / AP scenario

Table 4.4.3 presents the direct costs and welfare costs for the control of PM_{10}.

Table 4.4.3 Welfare costs and direct costs for the control of PM_{10}

	€ billion	
PM_{10}	Welfare costs	Direct costs
AP	1.0	1.6
TD	1.3	1.8

Public opinion

Public opinion on the introduction of measures to reduce PM_{10} and noise are absent at the European level. Despite this shortfall, Eurobarometer 43.1 (1995) indicates that proposals to increase / introduce taxes on sectors that pollute the environment are generally supported by Europeans.

5.3 Administrative complexity

A vehicle noise tax is administratively simple to implement as it could be attached to existing charges, such as the vehicle license scheme, (in the absence of annual noise measurements, proxies for noise levels might be age, type or size of vehicle), alternatively the noise tax could be built into existing fuel taxes.

5.4 Equity criterion

The equity effects of vehicle noise taxes are ambiguous.

5.5 Jurisdictional criterion

Urban transport issues are a localised and thus not a central issue. However there may need to be co-ordination between countries with respect to differentiated fuel taxes as vehicle owners may engage in arbitration.

Macroeconomic effect

Details of the macroeconomic effect are reported in *Technical Report on Socio-Economic Trends, Macro-Economic Impacts and Cost Interface.*

Appendix

Appendix A-1. Emissions EU (SNAP90 level 1)[35]

PM10 Ktonne	**1990**	**BL**	**TD**	**SO-kp**	**SO**	**AP**
Public power, cogeneration and district heating	461	366	53	250	169	34
Residential, Commercial and Institutional combustion	481	301	265	198	198	180
Industrial combustion*	414	252	55	123	106	33
Production processes **	467	415	228	415	415	228
Road transport	653	196	144	176	157	131
Other mobile sources and machinery	25	23	17	14	14	8
Waste treatment and disposal	100	6	6	6	6	6
Agriculture	23	25	25	25	25	25
TOTAL	2624	1583	793	1205	1088	646

Cadmium tonne	**1990**	**BL**	**TD**	**SO-kp**	**SO**	**AP**
Public power, cogeneration and district heating	19	17	3	10	9	2
Residential, Commercial and Institutional combustion	10	15	15	8	8	8
Industrial combustion*	29	23	6	10	10	3
Production processes **	78	86	32	86	86	32
Road transport	38	56	56	53	53	53
Other mobile sources and machinery	1	1	1	1	1	1
Waste treatment and disposal	17	3	4	3	3	4
Agriculture	5	5	5	5	5	5
TOTAL	197	206	122	176	175	108

Copper tonne	**1990**	**BL**	**TD**	**SO-kp**	**SO**	**AP**
Public power, cogeneration and district heating	106	80	16	56	46	10
Residential, Commercial and Institutional combustion	28	26	26	16	16	16
Industrial combustion*	259	171	26	55	54	13
Production processes **	320	282	145	282	282	145
Road transport	378	576	576	564	564	564
Other mobile sources and machinery	323	318	318	318	318	318
Waste treatment and disposal	37	7	7	7	7	7
TOTAL	1451	1459	1115	1298	1287	1074

Lead tonne	**1990**	**BL**	**TD**	**SO-kp**	**SO**	**AP**
Public power, cogeneration and district heating	196	156	48	110	98	43
Residential, Commercial and Institutional combustion	116	91	91	62	62	62
Industrial combustion*	595	581	145	254	252	80
Production processes **	1952	2051	886	2051	2051	886
Road transport	13111	3603	3603	3290	3290	3290
Other mobile sources and machinery	75	75	75	75	75	75
Waste treatment and disposal	328	70	71	70	70	71
Agriculture	1	1	1	1	1	1
TOTAL	16374	6626	4919	5913	5898	4508

[35] The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in the trends and the differences in scenarios. See section 2.1.2. of this appendix for a discussion on uncertainty in emission estimates for the base year 1990.

Mercury tonne	**1990**	**BL**	**TD**	**SO-kp**	**SO**	**AP**
Public power, cogeneration and district heating	44	37	22	30	30	22
Residential, Commercial and Institutional combustion	13	14	14	7	7	7
Industrial combustion*	34	26	13	11	11	5
Production processes **	115	105	68	105	105	68
Waste treatment and disposal	41	43	14	43	43	14
TOTAL	247	226	130	195	195	115

PAHs (6 of Borneff) tonne	**1990**	**BL**	**TD**	**SO-kp**	**SO**	**AP**
Public power, cogeneration and district heating	18	15	10	13	12	9
Residential, Commercial and Institutional combustion	1420	1827	909	863	863	443
Industrial combustion*	85	100	22	22	22	11
Production processes **	695	784	553	784	784	553
Solvent use	2211	2211	0	2211	2211	0
Road transport	1080	901	901	871	871	1495
Other mobile sources and machinery	47	26	21	25	25	21
Waste treatment and disposal	6	0	0	0	0	0
TOTAL	5561	5863	2418	4788	4788	2532

Dioxins/Furans g I-Teq	**1990**	**BL**	**TD**	**SO-kp**	**SO**	**AP**
Public power, cogeneration and district heating	450	436	399	416	412	389
Residential, Commercial and Institutional combustion	602	785	99	436	436	68
Industrial combustion*	1140	1354	369	586	580	245
Production processes **	1286	1397	449	1397	1397	449
Road transport	73	104	104	104	104	104
Other mobile sources and machinery	13	12	12	12	12	12
Waste treatment and disposal	2456	92	103	92	92	103
TOTAL	6020	4181	1537	3043	3033	1372

Benzene Ktonne	**1990**	**BL**	**TD**
Public power, cogeneration and district heating	1	1	1
Residential, Commercial and Institutional combustion	25	25	12
Industrial combustion*	6	6	6
Production processes **	9	9	9
Extraction and distribution of fossil fuels	6	6	3
Road transport	204	25	14
Other mobile sources and machinery	1	1	1
Waste treatment and disposal	20	20	20
Agriculture	77	76	76
TOTAL	349	169	141

Endosulfan (pesticide) tonne	**1990**	**BL**
Agriculture	1417	1442
TOTAL	1417	1442

Atrazine (pesticide) tonne	**1990**	**BL**
Agriculture	1213	1250
TOTAL	1213	1250

Pentachlorophenol (pesticide)*** tonne	**1990**	**BL**
Solvent use	660	652
Agriculture	233	176
TOTAL	893	828

Lindane (pesticide)**** tonne	**1990**	**BL**
Production processes **	36	40
Solvent use	72	72
Agriculture	364	358
TOTAL	472	470

** Exclusive of combustion in petroleum industries*
*** Inclusive of combustion in petroleum industries*
**** EU-restrictions on the use of pentachlorophenol have not been taken into account in the Baseline although this should have been done given the date of entry into force of EU-regulations (before August 1997). It is expected that due to these EU-regulations emissions will be almost negligible in 2010.*
***** Restrictions on the use of lindane imposed by the UN/ECE-POP-protocol have not been taken into account in the Baseline. This was done so because this protocol was settled in 1998 and therefore did not fit the definition of existing EU policies used throughout this study i.e. policies agreed upon as of August 1997. Nevertheless, it is expected that due to protocol requirements emissions will be almost negligible in 2010.*

Appendix A-2. Emissions 10 accession countries

(SNAP90 level 1)[36]

PM10 Ktonne		1990	**BL**	**TD**
	Public power, cogeneration and district heating	870	225	35
	Residential, Commercial and Institutional combustion	230	290	255
	Industrial combustion*	233	195	19
	Production processes **	470	373	57
	Road transport	130	47	19
	Other mobile sources and machinery	12	11	10
	Waste treatment and disposal	0	0	0
	Agriculture	8	9	9
	TOTAL	1952	1150	403

Cadmium tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	22	7	0
	Residential, Commercial and Institutional combustion	59	101	101
	Industrial combustion*	34	16	3
	Production processes **	34	30	13
	Road transport	8	10	10
	Other mobile sources and machinery	0	0	0
	Waste treatment and disposal	1	1	1
	TOTAL	159	166	129

Copper tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	113	38	3
	Residential, Commercial and Institutional combustion	219	229	229
	Industrial combustion*	313	107	9
	Production processes **	242	187	41
	Road transport	88	109	109
	Other mobile sources and machinery	27	27	27
	Waste treatment and disposal	3	3	0
	TOTAL	1006	700	418

Lead tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	173	63	3
	Residential, Commercial and Institutional combustion	361	425	425
	Industrial combustion*	395	114	9
	Production processes **	965	816	446
	Road transport	2104	165	165
	Other mobile sources and machinery	164	164	164
	Waste treatment and disposal	24	25	2
	TOTAL	4187	1772	1214

[36] The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in the trends and the differences in scenarios. See section 2.1.2. of this appendix for a discussion on uncertainty in emission estimates for the base year 1990.

Mercury tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	28	27	11
	Residential, Commercial and Institutional combustion	8	7	7
	Industrial combustion*	15	10	5
	Production processes **	21	19	15
	Waste treatment and disposal	5	5	2
	TOTAL	77	68	39

PAHs (6 of Borneff) tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	11	1	0
	Residential, Commercial and Institutional combustion	773	582	290
	Industrial combustion*	20	12	9
	Production processes **	306	303	187
	Solvent use	353	353	353
	Road transport	59	36	76
	Other mobile sources and machinery	15	12	11
	TOTAL	1539	1299	925

Dioxins/Furans g I-Teq		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	35	25	13
	Residential, Commercial and Institutional combustion	487	380	16
	Industrial combustion*	34	22	9
	Production processes **	1699	1592	1378
	Road transport	2	3	3
	Other mobile sources and machinery	2	2	2
	Waste treatment and disposal	277	287	1
	TOTAL	2539	2314	1423

Benzene Ktonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	0.3	0.3	0.3
	Residential, Commercial and Institutional combustion	7	7	4
	Industrial combustion*	1	1	1
	Production processes **	4	4	4
	Extraction and distribution of fossil fuels	1	1	0
	Road transport	36	3	2
	Other mobile sources and machinery	0	0	0
	Waste treatment and disposal	6	6	6
	Agriculture	18	18	18
	TOTAL	74	41	36

Endosulfan (pesticide) tonne		**1990**	**BL**
	Agriculture	136	166
	TOTAL	136	166

Atrazine (pesticide) tonne		**1990**	**BL**
	Agriculture	297	440
	TOTAL	297	440

Pentachlorophenol (pesticide) tonne	**1990**	**BL**
Solvent use	80	80
Agriculture	379	627
TOTAL	459	707

Lindane (pesticide)* ** tonne	**1990**	**BL**
Agriculture	99	115
TOTAL	99	115

** Exclusive of combustion in petroleum industries*
*** Inclusive of combustion in petroleum industries*
**** Restrictions on the use of lindane imposed by the UN/ECE-POP-protocol have not been taken into account in the Baseline. This was done so because this protocol was settled in 1998 and therefore did not fit the definition of existing EU policies used throughout this study i.e. policies agreed upon as of August 1997. Nevertheless, it is expected that due to protocol requirements emissions will be almost negligible in 2010.*

Appendix A-3. Emissions other countries[37] (not including the Russian Federation, Georgia, Ukraine, Azerbaijan) (SNAP90 level 1)[38]

PM10 Ktonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	315	125	12
	Residential, Commercial and Institutional combustion	189	122	108
	Industrial combustion*	466	157	20
	Production processes **	436	234	33
	Road transport	105	63	17
	Other mobile sources and machinery	30	29	22
	Waste treatment and disposal	0	0	0
	Agriculture	5	5	5
	TOTAL	1546	734	217

Cadmium tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	16	5	0
	Residential, Commercial and Institutional combustion	14	7	7
	Industrial combustion*	9	5	1
	Production processes **	34	23	7
	Road transport	11	10	10
	Other mobile sources and machinery	6	6	1
	Waste treatment and disposal	2	2	1
	TOTAL	91	58	27

Copper tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	71	18	2
	Residential, Commercial and Institutional combustion	106	63	63
	Industrial combustion*	48	18	2
	Production processes **	168	107	26
	Road transport	69	76	76
	Other mobile sources and machinery	87	87	83
	Waste treatment and disposal	3	3	0
	TOTAL	552	371	251

Lead tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	274	47	3
	Residential, Commercial and Institutional combustion	107	67	67
	Industrial combustion*	53	28	8
	Production processes **	1069	707	251
	Road transport	5010	170	170
	Other mobile sources and machinery	166	167	147
	Waste treatment and disposal	144	147	114
	TOTAL	6824	1333	760

[37] Albania, Bosnia Herzegovina, Belarus, Switzerland, Republic of Croatia, Republic of Moldova, FYR Macedonia, Norway, Ukraine, Federal Republic of Yugoslavia

[38] The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in the trends and the differences in scenarios. See section 2.1.2. of this appendix for a discussion on uncertainty in emission estimates for the base year 1990.

Mercury tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	13	9	3
	Residential, Commercial and Institutional combustion	9	6	6
	Industrial combustion*	9	6	3
	Production processes **	14	12	5
	Waste treatment and disposal	8	8	4
	TOTAL	54	41	21

PAHs (6 of Borneff) tonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	3	1	0
	Residential, Commercial and Institutional combustion	671	443	222
	Industrial combustion*	22	12	7
	Production processes **	225	198	112
	Solvent use	916	916	915
	Road transport	47	32	52
	Other mobile sources and machinery	56	39	21
	TOTAL	1941	1640	1329

Dioxins/Furans g I-Teq		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	35	14	6
	Residential, Commercial and Institutional combustion	636	418	17
	Industrial combustion*	61	34	14
	Production processes **	372	302	56
	Road transport	2	2	2
	Other mobile sources and machinery	20	20	20
	Waste treatment and disposal	307	320	25
	TOTAL	1434	1112	141

Benzene Ktonne		**1990**	**BL**	**TD**
	Public power, cogeneration and district heating	1	1	1
	Residential, Commercial and Institutional combustion	3	3	1
	Industrial combustion*	2	2	2
	Production processes **	12	12	12
	Extraction and distribution of fossil fuels	1	1	1
	Road transport	27	4	2
	Other mobile sources and machinery	0	0	0
	Waste treatment and disposal	5	5	5
	Agriculture	23	23	23
	TOTAL	73	50	46

Endosulfan (pesticide) tonne		**1990**	**BL**
	Agriculture	131	134
	TOTAL	131	134

Atrazine (pesticide)	**1990**	**BL**
tonne		
Agriculture	165	151
TOTAL	165	151

Pentachlorophenol (pesticide)	**1990**	**BL**
tonne		
Solvent use	104	104
Agriculture	106	102
TOTAL	210	206

Lindane (pesticide)* **	**1990**	**BL**
tonne		
Agriculture	375	332
TOTAL	375	332

** Exclusive of combustion in petroleum industries*

*** Inclusive of combustion in petroleum industries*

**** Restrictions on the use of lindane imposed by the UN/ECE-POP-protocol have not been taken into account in the Baseline. This was done so because this protocol was settled in 1998 and therefore did not fit the definition of existing EU policies used throughout this study i.e. policies agreed upon as of August 1997. Nevertheless, it is expected that due to protocol requirements emissions will be almost negligible in 2010.*

Appendix B-1. Emissions PM_{10} in EU countries (SNAP90 sector level 1)[39]

Austria ktonne	PM10	1990	BL	TD	SO-kp	SO	AP
	Public power, cogeneration and district heating	1.3	0.6	0.2	0.6	0.5	0.1
	Residential, Commercial and Institutional combustion	8.4	10.0	8.8	4.6	4.6	4.2
	Industrial combustion*	2.6	4.5	1.5	1.9	1.6	0.8
	Production processes **	8.5	9.3	5.3	9.3	9.3	5.3
	Road transport	8.6	2.6	1.9	2.3	2.1	1.7
	Other mobile sources and machinery	0.5	0.3	0.3	0.2	0.2	0.2
	Waste treatment and disposal	0.0	0.0	0.0	0.0	0.0	0.0
	Agriculture	0.5	0.4	0.4	0.4	0.4	0.4
	TOTAL	30	28	18	19	19	13

Belgium ktonne	PM10	1990	BL	TD	SO-kp	SO	AP
	Public power, cogeneration and district heating	1.8	2.0	1.4	0.2	0.1	0.0
	Residential, Commercial and Institutional combustion	13.4	12.8	11.3	8.9	8.9	8.2
	Industrial combustion*	8.2	11.2	2.9	4.4	4.2	1.2
	Production processes **	25.6	28.1	19.8	28.1	28.1	19.8
	Road transport	19.0	5.7	4.2	5.1	4.6	3.8
	Other mobile sources and machinery	0.4	0.2	0.1	0.2	0.2	0.1
	Waste treatment and disposal	0.7	0.3	0.3	0.3	0.3	0.3
	Agriculture	0.9	1.1	1.1	1.1	1.1	1.1
	TOTAL	70	61	41	48	47	34

Germany ktonne	PM10	1990	BL	TD	SO-kp	SO	AP
	Public power, cogeneration and district heating	346.4	264.1	21.2	173.9	118.7	16.6
	Residential, Commercial and Institutional combustion	267.0	95.0	83.6	53.3	53.3	49.1
	Industrial combustion*	249.7	110.1	17.4	38.4	29.0	8.7
	Production processes **	201.8	121.3	59.7	121.3	121.3	59.7
	Road transport	175.0	52.5	38.5	47.3	42.0	35.0
	Other mobile sources and machinery	3.6	2.5	1.5	2.4	2.4	1.3
	Waste treatment and disposal	0.7	1.4	1.2	1.4	1.4	1.2
	Agriculture	4.3	3.6	3.6	3.6	3.6	3.6
	TOTAL	1248	650	227	442	372	175

[39] The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in the trends and the differences in scenarios. See section 2.1.2. of this appendix for a discussion on uncertainty in emission estimates for the baseyear 1990.

Denmark		1990	BL	TD	SO-kp	SO	AP
Ktonne	PM10						
	Public power, cogeneration and district heating	2.6	2.3	2.0	6.5	5.4	0.3
	Residential, Commercial and Institutional combustion	9.2	9.2	8.1	5.0	5.0	4.6
	Industrial combustion*	2.4	1.3	0.6	1.0	0.9	0.5
	Production processes **	3.0	3.4	1.5	3.4	3.4	1.5
	Road transport	11.0	3.3	2.4	3.0	2.6	2.2
	Other mobile sources and machinery	1.5	1.0	0.5	0.8	0.8	0.3
	Waste treatment and disposal	0.2	0.1	0.1	0.1	0.1	0.1
	Agriculture	0.7	0.8	0.8	0.8	0.8	0.8
	TOTAL	31	21	16	21	19	10

Spain		1990	BL	TD	SO-kp	SO	AP
Ktonne	PM10						
	Public power, cogeneration and district heating	14.9	23.6	9.2	17.8	8.9	3.6
	Residential, Commercial and Institutional combustion	10.5	11.9	10.5	9.4	9.4	8.6
	Industrial combustion*	21.6	8.9	1.6	7.8	6.8	1.4
	Production processes **	29.5	36.7	20.2	36.7	36.7	20.2
	Road transport	58.0	17.4	12.8	15.7	13.9	11.6
	Other mobile sources and machinery	5.7	7.8	6.8	2.4	2.4	1.4
	Waste treatment and disposal	1.5	0.1	0.5	0.1	0.1	0.5
	Agriculture	1.9	2.8	2.8	2.8	2.8	2.8
	TOTAL	144	109	64	93	81	50

Finland		1990	BL	TD	SO-kp	SO	AP
Ktonne	PM10						
	Public power, cogeneration and district heating	3.5	3.8	0.9	3.2	2.7	1.0
	Residential, Commercial and Institutional combustion	12.4	8.8	7.8	10.8	10.8	9.9
	Industrial combustion*	8.6	18.6	5.4	5.3	5.0	2.1
	Production processes **	5.9	6.5	3.8	6.5	6.5	3.8
	Road transport	12.0	3.6	2.6	3.2	2.9	2.4
	Other mobile sources and machinery	0.5	0.5	0.4	0.2	0.2	0.1
	Waste treatment and disposal	0.0	0.0	0.0	0.0	0.0	0.0
	Agriculture	0.2	0.2	0.2	0.2	0.2	0.2
	TOTAL	43	42	21	29	28	19

France		1990	BL	TD	SO-kp	SO	AP
Ktonne	PM10						
	Public power, cogeneration and district heating	3.5	2.0	0.4	1.6	0.9	0.4
	Residential, Commercial and Institutional combustion	56.0	54.3	47.8	44.9	44.9	41.3
	Industrial combustion*	47.0	35.1	9.9	26.8	25.2	7.6
	Production processes **	50.7	52.1	27.8	52.1	52.1	27.8
	Road transport	110.0	33.0	24.2	29.7	26.4	22.0
	Other mobile sources and machinery	2.0	1.7	1.0	1.7	1.7	1.0
	Waste treatment and disposal	78.0	1.6	1.5	1.6	1.6	1.5
	Agriculture	4.6	4.4	4.4	4.4	4.4	4.4
	TOTAL	352	184	117	163	157	106

United Kingdom Ktonne	PM10	1990	BL	TD	SO-kp	SO	AP
	Public power, cogeneration and district heating	31.0	15.3	6.5	15.4	8.1	3.7
	Residential, Commercial and Institutional combustion	38.0	19.9	17.5	13.7	13.7	12.6
	Industrial combustion*	25.3	21.9	5.4	8.6	7.3	2.5
	Production processes **	45.3	50.1	28.0	50.1	50.1	28.0
	Road transport	94.0	28.2	20.7	25.4	22.6	18.8
	Other mobile sources and machinery	5.0	3.8	2.8	3.1	3.1	2.0
	Waste treatment and disposal	2.8	0.1	0.6	0.1	0.1	0.6
	Agriculture	2.7	3.0	3.0	3.0	3.0	3.0
	TOTAL	244	142	85	119	108	71

Greece Ktonne	PM10	1990	BL	TD	SO-kp	SO	AP
	Public power, cogeneration and district heating	10.9	9.5	3.5	21.1	16.8	6.9
	Residential, Commercial and Institutional combustion	6.5	12.0	10.6	8.4	8.4	7.7
	Industrial combustion*	5.1	6.1	1.7	4.9	4.3	1.4
	Production processes **	7.1	9.3	4.7	9.3	9.3	4.7
	Road transport	18.0	5.4	4.0	4.9	4.3	3.6
	Other mobile sources and machinery	2.4	2.4	2.1	0.8	0.8	0.5
	Waste treatment and disposal	0.0	0.0	0.1	0.0	0.0	0.1
	Agriculture	0.4	0.5	0.5	0.5	0.5	0.5
	TOTAL	50	45	27	50	44	25

Ireland ktonne	PM10	1990	BL	TD	SO-kp	SO	AP
	Public power, cogeneration and district heating	3.7	4.9	1.5	3.1	2.9	1.2
	Residential, Commercial and Institutional combustion	16.4	14.8	13.0	13.8	13.8	12.7
	Industrial combustion*	2.4	0.9	0.4	0.9	0.7	0.5
	Production processes **	1.1	1.2	0.6	1.2	1.2	0.6
	Road transport	4.9	1.5	1.1	1.3	1.2	1.0
	Other mobile sources and machinery	0.2	0.2	0.2	0.1	0.1	0.0
	Waste treatment and disposal	0.0	0.0	0.0	0.0	0.0	0.0
	Agriculture	0.6	0.6	0.6	0.6	0.6	0.6
	TOTAL	29	24	17	21	20	17

Italy ktonne	PM10	1990	BL	TD	SO-kp	SO	AP
	Public power, cogeneration and district heating	33.9	33.8	4.0	2.4	1.3	0.2
	Residential, Commercial and Institutional combustion	26.8	32.3	28.4	12.7	12.7	11.7
	Industrial combustion*	28.0	23.9	5.3	14.4	13.3	3.1
	Production processes **	50.9	58.8	34.9	58.8	58.8	34.9
	Road transport	100.0	30.0	22.0	27.0	24.0	20.0
	Other mobile sources and machinery	0.7	0.5	0.2	0.5	0.5	0.2
	Waste treatment and disposal	16.0	1.6	1.5	1.6	1.6	1.5
	Agriculture	2.8	3.3	3.3	3.3	3.3	3.3
	TOTAL	259	184	100	121	116	75

Luxembourg ktonne		1990	BL	TD	SO-kp	SO	AP
	PM10						
	Public power, cogeneration and district heating	0.0	0.0	0.0	0.0	0.0	0.0
	Residential, Commercial and Institutional combustion	0.7	0.5	0.4	0.5	0.5	0.4
	Industrial combustion*	0.7	0.3	0.0	0.3	0.2	0.0
	Production processes **	3.7	3.5	2.5	3.5	3.5	2.5
	Road transport	0.8	0.2	0.2	0.2	0.2	0.2
	Other mobile sources and machinery	0.0	0.0	0.0	0.0	0.0	0.0
	Waste treatment and disposal	0.0	0.0	0.0	0.0	0.0	0.0
	Agriculture	0.0	0.0	0.0	0.0	0.0	0.0
	TOTAL	6	5	3	4	4	3

Netherlands ktonne		1990	BL	TD	SO-kp	SO	AP
	PM10						
	Public power, cogeneration and district heating	2.5	2.0	0.3	1.4	1.3	0.2
	Residential, Commercial and Institutional combustion	3.7	2.3	2.0	1.5	1.5	1.4
	Industrial combustion*	3.2	2.0	0.4	1.0	0.8	0.3
	Production processes **	20.7	18.4	10.1	18.4	18.4	10.1
	Road transport	22.8	6.9	5.0	6.2	5.5	4.6
	Other mobile sources and machinery	1.2	1.1	0.9	0.7	0.7	0.4
	Waste treatment and disposal	0.2	0.3	0.2	0.3	0.3	0.2
	Agriculture	2.7	2.9	2.9	2.9	2.9	2.9
	TOTAL	57	36	22	32	31	20

Portugal ktonne		1990	BL	TD	SO-kp	SO	AP
	PM10						
	Public power, cogeneration and district heating	4.1	2.0	1.3	1.8	0.8	0.1
	Residential, Commercial and Institutional combustion	2.1	8.5	7.5	3.5	3.5	3.2
	Industrial combustion*	3.2	2.9	0.7	2.2	1.8	0.7
	Production processes **	4.1	5.6	3.1	5.6	5.6	3.1
	Road transport	10.0	3.0	2.2	2.7	2.4	2.0
	Other mobile sources and machinery	0.2	0.2	0.1	0.2	0.2	0.1
	Waste treatment and disposal	0.0	0.0	0.1	0.0	0.0	0.1
	Agriculture	0.5	0.7	0.7	0.7	0.7	0.7
	TOTAL	24	23	16	17	15	10

Sweden ktonne		1990	BL	TD	SO-kp	SO	AP
	PM10						
	Public power, cogeneration and district heating	0.5	0.7	0.4	0.8	0.5	0.1
	Residential, Commercial and Institutional combustion	10.4	8.8	7.8	6.7	6.7	6.2
	Industrial combustion*	5.9	4.9	2.2	5.5	4.9	2.5
	Production processes **	9.0	10.3	5.9	10.3	10.3	5.9
	Road transport	8.8	2.6	1.9	2.4	2.1	1.8
	Other mobile sources and machinery	0.9	0.5	0.4	0.4	0.4	0.3
	Waste treatment and disposal	0.2	0.3	0.3	0.3	0.3	0.3
	Agriculture	0.3	0.4	0.4	0.4	0.4	0.4
	TOTAL	36	29	19	27	26	17

** Exclusive combustion in petroleum industries*

** Inclusive combustion in petroleum industries

Appendix B-2. Emissions PM_{10} in accession countries (SNAP90 sector level 1)[40]

Bulgaria ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	76.6	16.1	2.7
	Residential, Commercial and Institutional combustion	7.3	8.0	7.0
	Industrial combustion*	38.6	21.1	4.2
	Production processes **	34.7	38.7	3.6
	Road transport	19.0	12.4	3.0
	Other mobile sources and machinery	0.5	0.5	0.3
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.7	0.7	0.7
	TOTAL	177	97	22

Czech Republic ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	84.6	16.2	2.7
	Residential, Commercial and Institutional combustion	35.5	9.7	8.5
	Industrial combustion*	11.7	3.6	0.7
	Production processes **	80.3	53.3	9.9
	Road transport	16.0	3.2	2.2
	Other mobile sources and machinery	0.8	0.8	0.8
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.8	1.0	1.0
	TOTAL	230	88	26

Estonia ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	3.7	3.1	0.2
	Residential, Commercial and Institutional combustion	4.0	2.9	2.5
	Industrial combustion*	11.2	22.1	1.1
	Production processes **	9.3	6.1	0.8
	Road transport	1.6	1.0	0.3
	Other mobile sources and machinery	0.7	0.7	0.5
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.1	0.1	0.1
	TOTAL	31	36	5

[40] The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in the trends and the differences in scenarios. See section 2.1.2. of this appendix for a discussion on uncertainty in emission estimates for the baseyear 1990.

Hungary ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	30.9	7.7	1.3
	Residential, Commercial and Institutional combustion	25.0	23.1	20.3
	Industrial combustion*	2.4	1.2	0.4
	Production processes **	35.0	31.7	3.2
	Road transport	20.0	4.0	2.8
	Other mobile sources and machinery	0.3	0.3	0.1
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	1.1	1.1	1.1
	TOTAL	115	69	29

Lithuania ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	8.8	1.7	0.3
	Residential, Commercial and Institutional combustion	9.5	9.4	8.3
	Industrial combustion*	26.4	88.3	4.5
	Production processes **	21.2	10.9	1.6
	Road transport	3.7	2.4	0.6
	Other mobile sources and machinery	1.6	1.6	1.3
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.1	0.1	0.1
	TOTAL	71	114	17

Latvia ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	6.3	3.7	0.4
	Residential, Commercial and Institutional combustion	6.8	5.7	5.0
	Industrial combustion*	19.3	25.1	1.3
	Production processes **	15.8	10.5	1.3
	Road transport	2.7	1.8	0.4
	Other mobile sources and machinery	1.2	1.1	0.9
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.1	0.1	0.1
	TOTAL	52	48	9

Poland ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	510.4	120.2	16.0
	Residential, Commercial and Institutional combustion	104.1	198.1	174.3
	Industrial combustion*	108.9	18.4	4.1
	Production processes **	113.2	90.2	18.7
	Road transport	38.0	7.6	5.3
	Other mobile sources and machinery	4.4	4.2	3.8
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	2.4	3.0	3.0
	TOTAL	881	442	225

Romania ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	86.2	50.1	9.9
	Residential, Commercial and Institutional combustion	16.1	16.0	14.1
	Industrial combustion*	6.4	9.1	1.3
	Production processes **	109.6	104.6	11.5
	Road transport	17.0	11.1	2.7
	Other mobile sources and machinery	1.7	1.7	1.7
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	2.3	2.3	2.3
	TOTAL	239	195	43

Slovakia ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	48.3	3.5	0.6
	Residential, Commercial and Institutional combustion	20.5	16.7	14.7
	Industrial combustion*	6.4	5.1	0.8
	Production processes **	45.7	22.5	5.6
	Road transport	9.1	1.8	1.3
	Other mobile sources and machinery	0.5	0.5	0.5
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.5	0.5	0.5
	TOTAL	131	51	24

Slovenia ktonne	PM10	1990	BL	TD
	Public power, cogeneration and district heating	14.3	3.0	0.5
	Residential, Commercial and Institutional combustion	1.2	0.3	0.3
	Industrial combustion*	1.4	0.7	0.1
	Production processes **	5.2	4.3	0.5
	Road transport	2.7	1.8	0.4
	Other mobile sources and machinery	0.0	0.0	0.0
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.1	0.1	0.1
	TOTAL	25	10	2

** Exclusive combustion in petroleum industries*

*** Inclusive combustion in petroleum industries*

Appendix B-3. Emissions PM_{10} other countries (SNAP90 sector level 1)[41]

Albania		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	1.0	0.6	0.0
	Residential, Commercial and Institutional combustion	1.7	1.6	1.4
	Industrial combustion*	6.4	5.0	0.5
	Production processes **	0.0	0.0	0.0
	Road transport	1.6	1.0	0.3
	Other mobile sources and machinery	0.0	0.0	0.0
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.1	0.2	0.2
	TOTAL	11	8	2

Bosnia Herzegovina		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	31.7	28.2	1.3
	Residential, Commercial and Institutional combustion	2.8	1.4	1.2
	Industrial combustion*	3.1	1.3	0.1
	Production processes **	11.8	10.0	1.1
	Road transport	6.2	4.0	1.0
	Other mobile sources and machinery	0.0	0.0	0.0
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.2	0.2	0.2
	TOTAL	56	45	5

Belarus		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	23.7	13.3	1.6
	Residential, Commercial and Institutional combustion	25.7	6.9	6.1
	Industrial combustion*	72.1	71.6	3.8
	Production processes **	60.7	30.4	4.4
	Road transport	10.0	6.5	1.6
	Other mobile sources and machinery	4.5	4.4	3.4
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.4	0.3	0.3
	TOTAL	197	133	21

Switzerland		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	0.1	0.1	0.0
	Residential, Commercial and Institutional combustion	11.1	11.1	9.7
	Industrial combustion*	2.0	2.9	0.7
	Production processes **	2.3	2.3	1.3
	Road transport	5.2	1.4	1.0
	Other mobile sources and machinery	0.1	0.1	0.1
	Waste treatment and disposal	0.3	0.3	0.3
	Agriculture	0.3	0.2	0.2
	TOTAL	21	18	13

[41] The number of digits in the estimates do not represent the actual accuracy of each estimate but are shown to provide better insight in the trends and the differences in scenarios. See section 2.1.2. of this appendix for a discussion on uncertainty in emission estimates for the baseyear 1990.

Republic of Croatia		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	33.8	3.4	0.6
	Residential, Commercial and Institutional combustion	2.9	3.0	2.6
	Industrial combustion*	3.3	1.5	0.3
	Production processes **	12.5	10.4	1.1
	Road transport	6.5	2.0	1.0
	Other mobile sources and machinery	0.0	0.0	0.0
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.3	0.2	0.2
	TOTAL	59	20	6

Republic of Moldova		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	15.4	12.1	0.5
	Residential, Commercial and Institutional combustion	1.4	0.4	0.3
	Industrial combustion*	1.5	1.2	0.1
	Production processes **	5.7	4.7	0.5
	Road transport	3.0	2.0	0.5
	Other mobile sources and machinery	0.0	0.0	0.0
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.1	0.1	0.1
	TOTAL	27	21	2

FYR Macedonia		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	15.4	12.1	0.5
	Residential, Commercial and Institutional combustion	1.4	0.4	0.3
	Industrial combustion*	1.5	1.2	0.1
	Production processes **	5.7	4.7	0.5
	Road transport	3.0	2.0	0.5
	Other mobile sources and machinery	0.0	0.0	0.0
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	0.1	0.1	0.1
	TOTAL	27	21	2

Norway		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	0.0	0.0	0.0
	Residential, Commercial and Institutional combustion	1.7	1.8	1.6
	Industrial combustion*	3.7	3.2	1.0
	Production processes **	6.3	6.6	2.1
	Road transport	3.8	1.0	0.8
	Other mobile sources and machinery	2.4	2.4	1.6
	Waste treatment and disposal	0.1	0.1	0.1
	Agriculture	0.1	0.2	0.2
	TOTAL	18	15	7

Ukraine		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	122.0	22.1	4.9
	Residential, Commercial and Institutional combustion	133.9	93.7	82.5
	Industrial combustion*	365.7	64.3	12.8
	Production processes **	303.5	142.9	19.2
	Road transport	52.0	33.8	8.3
	Other mobile sources and machinery	23.0	22.0	17.0
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	2.0	2.2	2.2
	TOTAL	1002	381	147

Federal Republic of Yugoslavia		1990	BL	TD
kton	PM10			
	Public power, cogeneration and district heating	71.7	32.5	2.6
	Residential, Commercial and Institutional combustion	6.3	2.3	2.0
	Industrial combustion*	7.0	4.4	0.5
	Production processes **	27.1	21.9	2.5
	Road transport	14.0	9.1	2.2
	Other mobile sources and machinery	0.1	0.1	0.1
	Waste treatment and disposal	0.0	0.0	0.0
	Agriculture	1.5	1.3	1.3
	TOTAL	128	72	11

** Exclusive combustion in petroleum industries*
*** Inclusive combustion in petroleum industries*

Appendix C: Maps for deposition and exceedances of critical loads

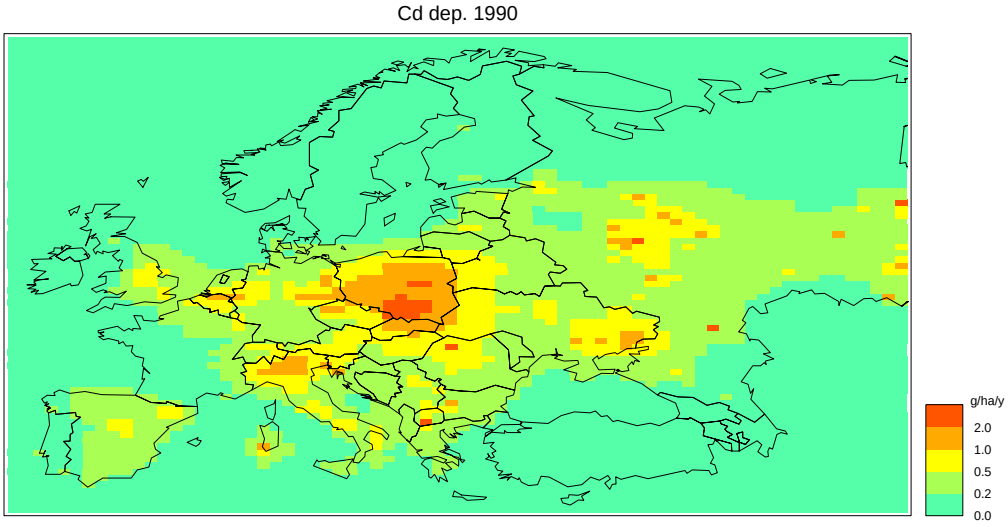

Figure C.1. a. Calculated cadmium deposition over Europe in 1990 (g ha^{-1} y^{-1}).

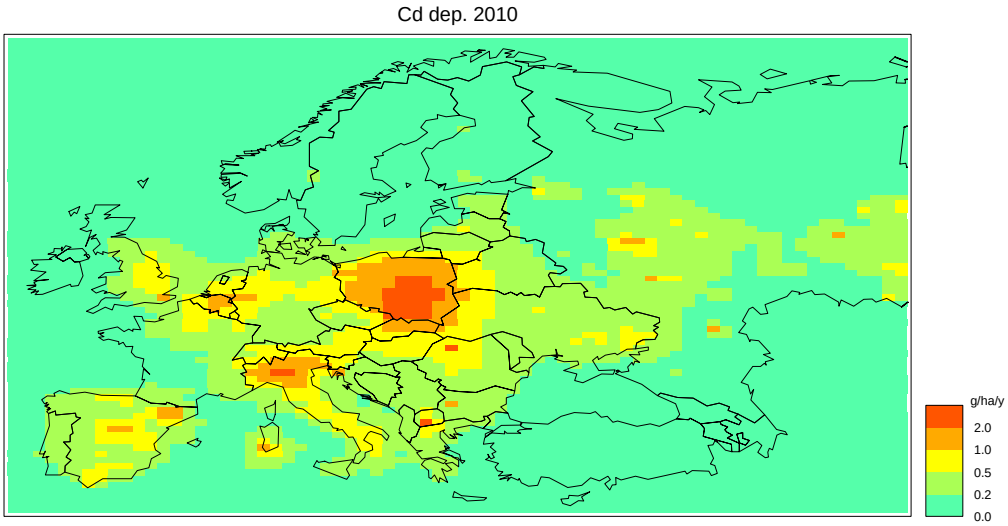

Figure C.1. b. Calculated cadmium deposition over Europe in 2010 under BL conditions (g ha^{-1} y^{-1}).

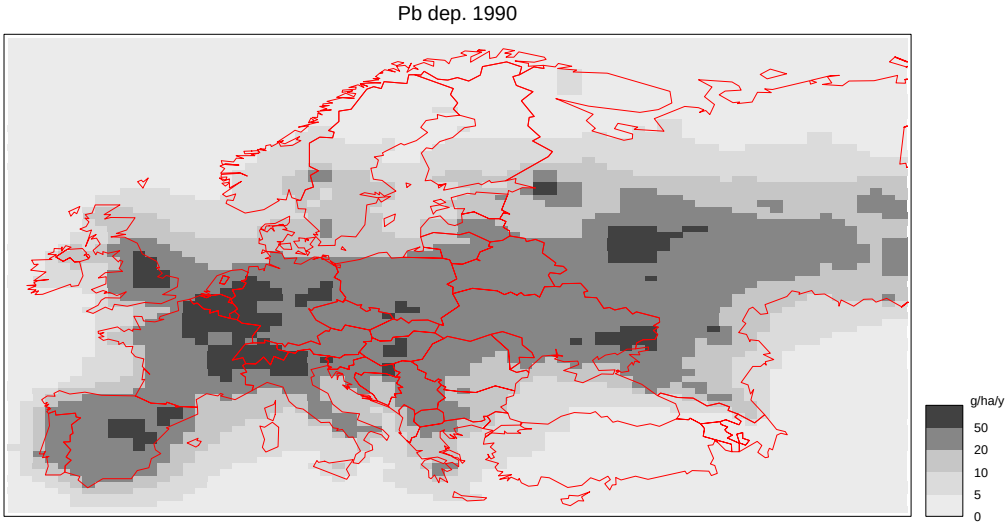

Figure C.1.c. Calculated lead deposition over Europe in 1990 (g ha^{-1} y^{-1}). High depositions for Spain are unlikely and can be explained by the fact that modelling results are based on preliminary TNO emission data characterised by extremely high emissions for the sector 'other transport' in Spain (see Appendix D).

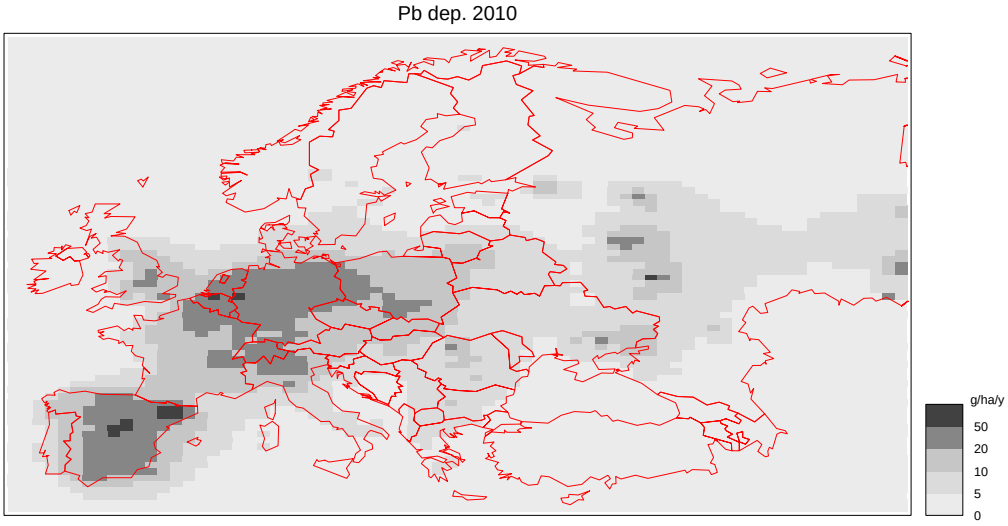

Figure C.1.d. Calculated lead deposition over Europe in 2010 under BL conditions (g ha^{-1} y^{-1}). High depositions for Spain are unlikely and can be explained by the fact that modelling results are based on preliminary TNO emission data characterised by extremely high emissions for the sector 'other transport' in Spain (see Appendix D).

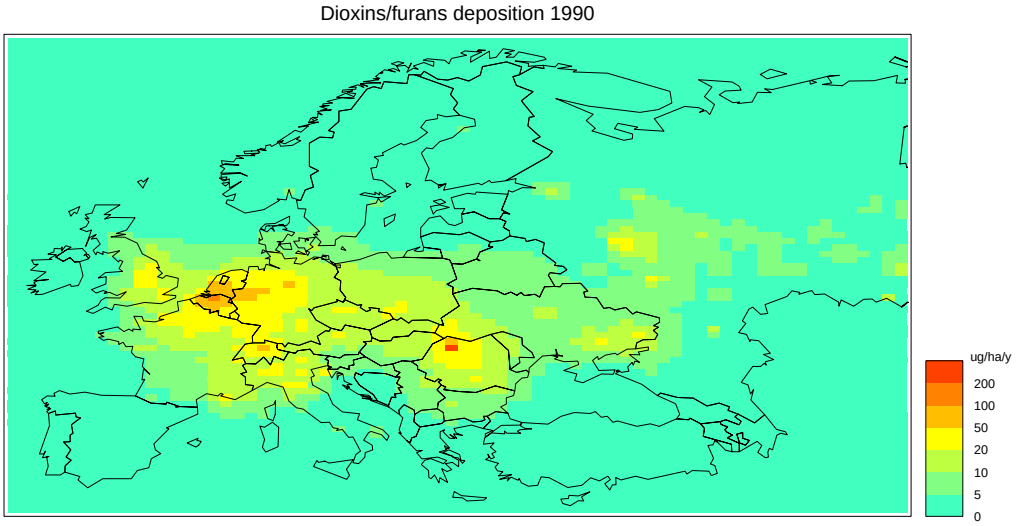

Figure C.1.e. Calculated dioxins/furans deposition over Europe in 1990 (µg TEQ ha^{-1} y^{-1}).

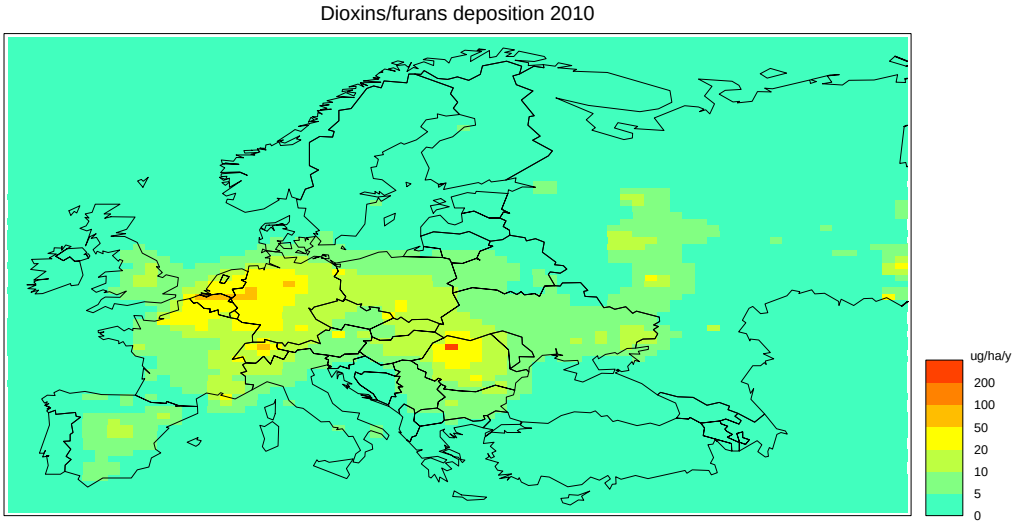

Figure C.1.f.. Calculated dioxins/furans deposition over Europe in 2010 under BL conditions (µg TEQ ha^{-1} y^{-1}).

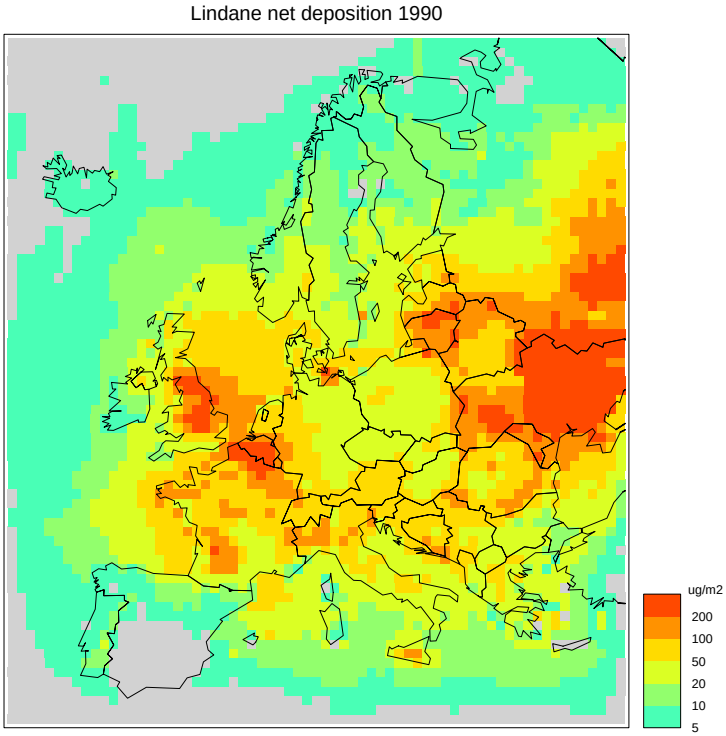

Figure C.2.a.. Calculated lindane deposition over Europe in 1990 (µg m^{-2} y^{-1}).

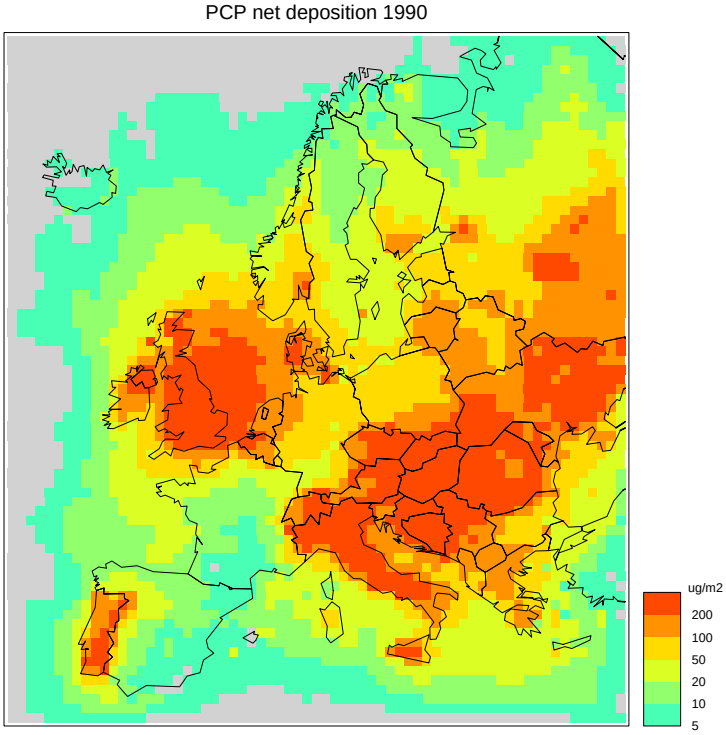

Figure C.2.b. Calculated PCP deposition over Europe in 1990 (µg m^{-2} y^{-1}).

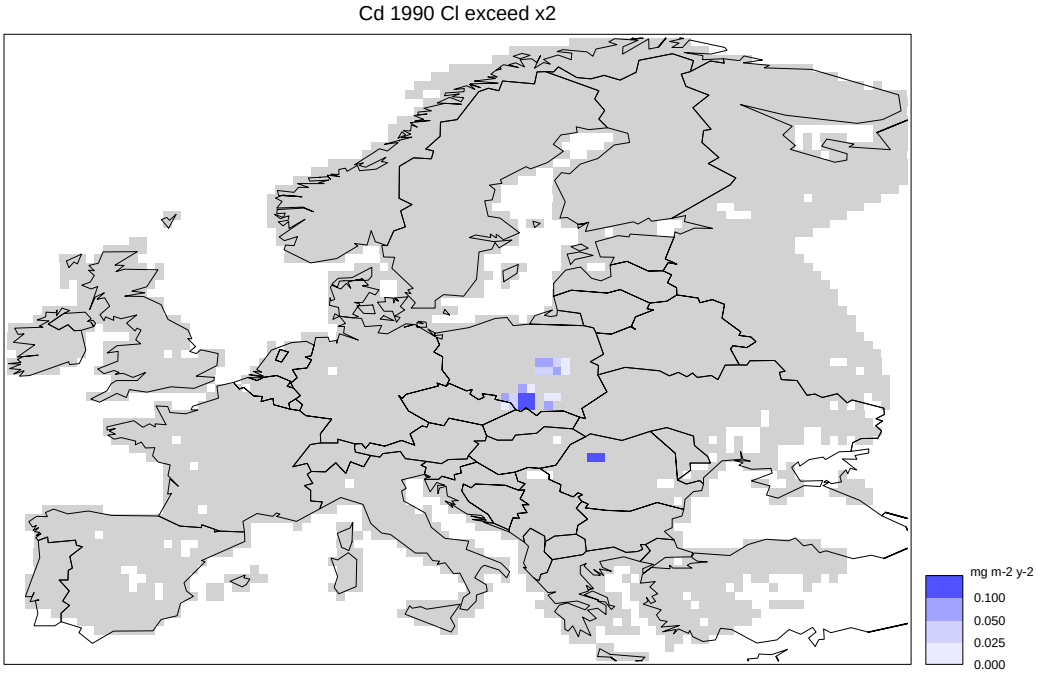

Figure C.3.. a. Exceedances of critical loads for forest soils for cadmium over Europe in 1990 (mg m^{-2} y^{-1}).

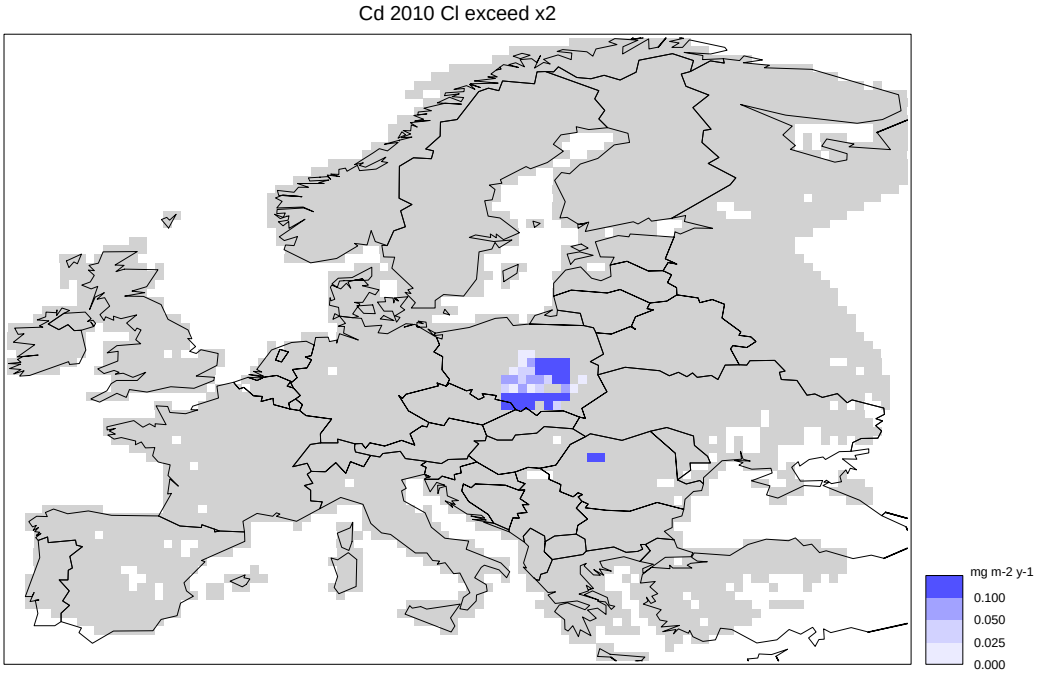

Figure C.3. b. Exceedances of critical loads for forest soils for cadmium over Europe in 2010 under BL conditions (mg m^{-2} y^{-1}).

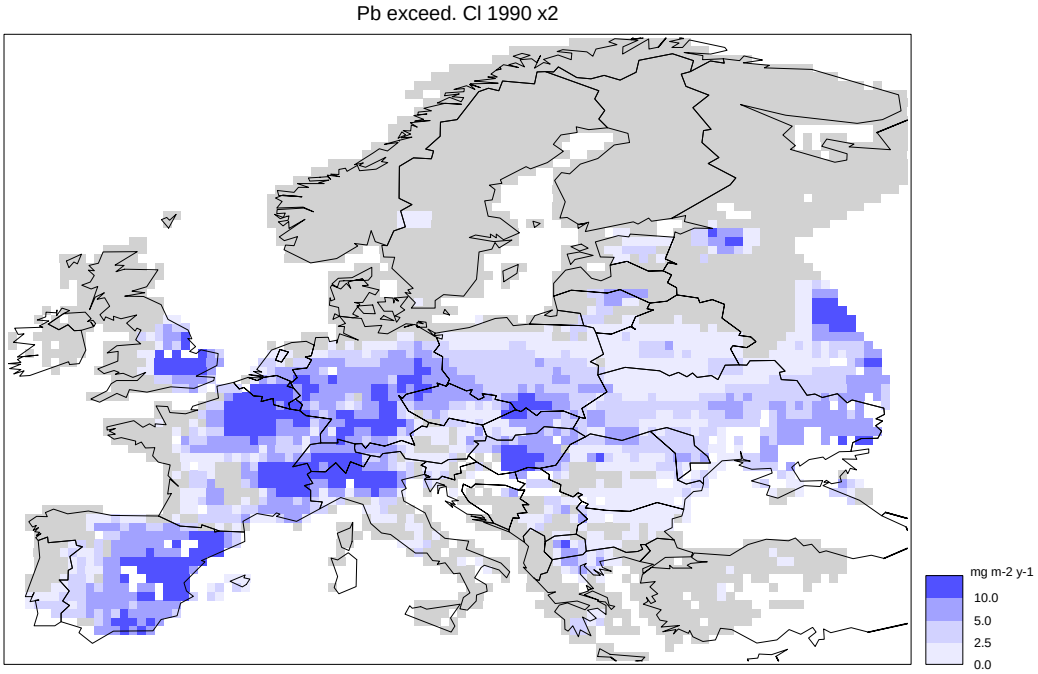

Figure C.3.c. Exceedances of critical loads for forest soils for lead over Europe in 1990 (mg m^{-2} y^{-1}).

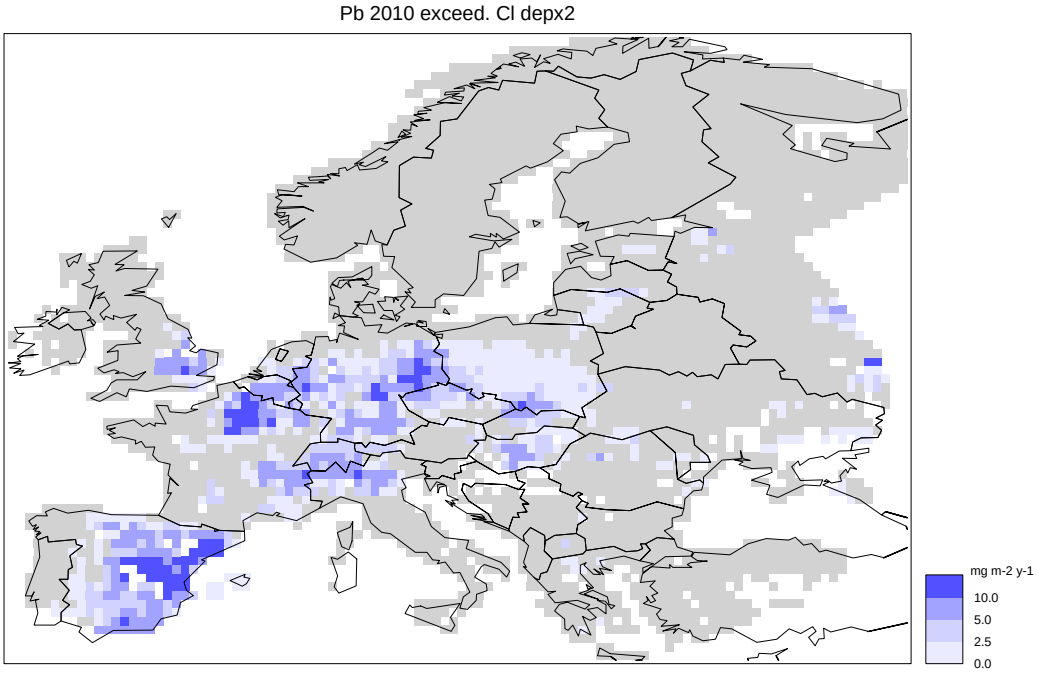

Figure C.4.d. Exceedances of critical loads for forest soils for lead over Europe in 2010 under BL conditions (mg m^{-2} y^{-1}).

Appendix D: Revisions of preliminary TNO emission data set

The Baseline (BL), Technology Driven (TD) and Accelerated Policy scenarios (AP) presented throughout this technical document are based on a study performed by TNO. The TNO data set and methodological aspects have been reported in a separate background document (Visschedijk et al., 1998). Parts of this dataset have been changed by RIVM for use within this study. An explanation for these revisions is given in this appendix.[42]

- For Cu, Cd, Pb and PAHs, 1990 TNO emission estimates (and thus future trends) were corrected for some extreme outliers. The sector 'other transport' in Spain appeared to dominate the EU emissions for Cu, Cd, Pb; the PAH emissions appeared to be dominated by residential combustion in France. Such high emissions seemed unlikely and were revised using aggregated per capita emissions derived from other EU countries. In addition, TNO projections for these pollutants did not reflect anticipated future emissions reductions due to the continued implementation of current EU waste incineration directives. Thus, future emissions for Cu, Cd, Pb and Hg have been recalculated per country from the expected amount of waste generated and applicable emission standards for particulates and mercury[43]. Finally, the emissions of PAHs under the TD scenario have been updated by recognition of an extra control option i.e. the replacement of PAH-based wood preservation techniques with alternative techniques.
- For PM_{10}, new insights were considered on the magnitude of emissions from lifestock agricultural stable emissions. Furthermore, emission trends for road transport emissions were recalculated. For waste incineration, BL and TD scenarios were corrected for the effects of current EU emission standards, which were not reflected in the TNO projections. Finally, TD emissions trends for residential combustion were revised by taken into consideration advanced technologies for coal and biomass burning in households.

Data revisions for PM_{10} emissions are presented in more detail below. As mentioned, PM_{10} emission figures have been updated for the agricultural, road transport and waste incineration sectors.

PM_{10} agriculture
Emission factors used by TNO for livestock emissions from stables are extremely high compared to emission factors derived from PM_{10} measurements on stables in various European countries (Groot Koerkamp et al., 1996). Using these revised factors, EU emissions for livestock emissions are estimated at 23 ktonne instead of the 306 ktonne reported by TNO. It should be noted that the figures for agriculture used throughout this study do not account for blown-up dust from bare agriculture land areas, land preparation, fertiliser application and harvesting.

PM_{10} road transport
According to the reported TNO data set, 57% of total PM_{10} emissions from road traffic in the reference year 1990 are due to other sources than combustion i.e. tyre and brake wear, wear of the road surface and resuspension. Only 43% comes from tailpipe emissions. TNO estimates are based on a study by Israel et al., (1994; TNO, 1997a). With such high emission estimates for non-tailpipe emissions, almost no reduction in emissions for road traffic is projected, despite the stricter PM_{10} emission standards for exhaust emissions. This trend is explained by the non-exhaust emissions which will increase with the increase in road traffic.

Compared to TNO estimates, national emission inventories for the Netherlands and the UK report much lower emissions for non-exhaust transport sources, although it should be noted that both inventories recognise that emissions may be underestimated due to the fact that resuspension is excluded from the inventory. Both inventories recognise that this is an area that needs further investigation in order to quantify this source accurately. In the Dutch emissions inventory 5% of the total traffic emission is reported as non-exhaust.

Given the scarce knowledge on the magnitude of emissions due to resuspension, RIVM considered it unwise for this study to include such emissions. For this study it was decided to present emission trends only for those sources for which the magnitude of emissions is relatively well known. These sources are tailpipe emissions and emissions due to tyre, brake and road wear. Due to the strict planning for the related EU-98 report (EEA,1999) it was not possible to make elaborate new calculations for the sector road tranport. Therefore it was decided to revise the TNO-data in a quick and simple way keeping in mind the choice made that a possible overestimation of future traffic emissions due to the incorporation of a rather unknown source such as resuspension should in any way be avoided. The TNO-dataset was used as a starting point. Total emissions for the year 1990 reported by TNO were kept unchanged but the share of non-exhaust emissions in the overall total was lowered according to the results of the Dutch emission inventory (see table D-

[42] Revised emission figures for PM_{10} have been incorporated in the EU98 State of the Environment report (EEA, 1999, chapter 3.12), However, revised final data could not be assimilated in time for Cu, Cd, Pb and PAHs. For these substances, original TNO data have been used in the EU98 report.
[43] Emission reductions for Cu, Cd and Pb have been derived from percentage particulate emission reductions.

1). These new emission figures were used for model calculations for the related EU-98 report. The intention was to update these estimates in a later stage. However, this seemed to be impractical due to time and resource limitations, and also because the results of the related EU98-report should preferably be consistent with this study, and so the revised dataset was incorporated in this study as the final dataset. However, some more detailed emission calculations have been performed by RIVM in a later stage to verify the errors that may have been introduced so.

In total four data sets have been prepared in the course of this study:
1. TNO data set (Visschedijk et al., 1998, TNO)
2. **Revised data set (final dataset used throughout this study and for the EU98 report, RIVM),**
3. Improved data set 1 (emission factors for non-exhaust according to the Dutch emissions inventory and exhaust emissions according to the TNO data set, RIVM)
4. Improved data set 2 (emission factors for non-exhaust and exhaust according to the Dutch emissions inventory, RIVM).

Results for these data sets and details on methodology are presented below.

TNO data set (PM_{10} transport)
The TNO dataset incorporates high emission estimates for resuspension based on a study by Israel (1994). Results and methodology have been reported in separate TNO reports (Berdowski, 1997a, Visschedijk et al., 1998). This dataset has not been used for this study.

Revised final data set (PM_{10} transport)
When air quality calculations had to be finished for the EU98 report, no time was available to revise the reported TNO-dataset in a elaborate way. Thus , TNO figures were revised in a simple and quick way, keeping in mind the choice made that a possible overestimation of future traffic emissions due to the incorporation of a rather unknown source such as resuspension should in any way be avoided. To keep in line with the EU98 report such revised data where also used for the EU priority study.

Calculations were performed *at the overall EU level.* Total road transport emission data for the EU for 1990 were not changed in comparison to TNO data. However, the share of non-exhaust in total road transport emissions reported by TNO (57%, Visschedijk et al., 1998) was changed according to the Dutch emission inventory i.e. 5% non-exhaust (Annema et al., 1997). Recalculated emissions for exhaust and non-exhaust were allocated to separate vehicle categories according to the emission-split reported by TNO.

Exhaust emissions for vehicle categories were projected by multiplying the re-calculated 1990-emissions by two factors: one representing the growth in gasoline and diesel use and the other representing the effect of new EU standards (Visschedijk et al., 1998). Non-exhaust emissions were projected by using the growth in gasoline and diesel use as proxies (Visschedijk et al., 1998). For the purpose of urban stress calculations country estimates were also needed. Estimates by country were derived by simply multiplying country estimates for 1990 by the estimated decline in EU emissions (70%).

As previously mentioned, one should consider that the results of the simple analysis are far from ideal. However, RIVM has used these figures as input for the urban stress calculations for the EU98 report (EEA, 1999), since no other dataset was available at that time. After exceeding the deadline for the EU98 report, RIVM made a more comprehensive analysis *(improved data sets 1 and 2, see below)* to get an idea of the error made by using the simplified approach. These alternative emissions calculations were performed at the country level.

Improved data set 1 (PM_{10} road transport)
Non-exhaust emissions reported by TNO for the reference year 1990 were recalculated using emission factors from the Dutch emission inventory (CBS, 1998). Dutch non-exhaust emission factors are given in *Table D-1.* Non-exhaust emissions for 2010 were projected using the growth in gasoline and diesel use as proxies (Visschedijk et al., 1998). Exhaust TNO-emission estimates for 1990 were not changed. Exhaust emissions by vehicle category were projected by multiplying the exhaust emissions reported by TNO for 1990 by two factors: one factor representing the growth in gasoline and diesel use per country, the other the effect of new emission standards for vehicles (Visschedijk et al., 1998).

Improved data set 2 (PM_{10} road transport)
Non-exhaust as well as exhaust emissions for 1990 reported by TNO were re-calculated using Dutch emission factors (CBS, 1998). Calculations were performed at the country level in the same way as stated above.

Table D-1 shows the TNO and the Dutch emission factors used for the reference year 1990. Calcualtion results are shown in *Table D-2*. Comparing the simple approach (*revised data set*) with the improved more realistic approaches (see *improved data sets 1 and 2*), following conclusions may be drawn:

- the *relative* change in total transport emissions (exhaust, tyre, break and road wear) in the 1990-2010 period is estimated correctly with the simple approach, but *absolute* emission reductions for 1990-2020 are overestimated.
- the magnitude of *non-exhaust emissions* are more or less correct, but *exhaus*t emissions in 1990 and 2010 are overestimated (and with this also the potential for further emission reduction due to full penetration of EURO-4 emission standards).

Conclusions on PM_{10} emission dataset fort road transport used in this study

Summarising the results, it may be concluded that the magnitude of non-exhaust emissions (tyre wear, brake wear, wear of the road surface and resuspension) are highly determinative for future PM_{10} emissions trends from road transport (*table D-2*) . When interpreting results of this study it should be considered that traffic emissions do not incorporate resuspension emissions. The magnitude of estimates for remaining *non-exhaust emissions* (tyre, break and road wear) is more or less correct. *Exhaus*t emissions in 1990 and 2010 have been clearly overestimated (with about 25% for 1990 and 5% for 2010). Furthermore, should it be recognised that real-world emissions by road transport may be much higher due to the resuspension of dust (Harrison et al., 1996). However knowledge about this emissionsource is scarce. If estimates by TNO would be right this would have consequences for estimated emissiontrends from mobile sources. According to the TNO-dataset an increase with 10% is anticipated for the year 2010 compared to 1990. Resuspension emissions clearly need further investigation.

PM_{10} waste incineration

TNO projections did not reflect effects on emissions of current EU waste incineration directives (emissions increased by 200%). Therefore, emissions for particulates for 2010 have been recalculated per country from the expected amount of waste generated and applicable particulates emission standards.

Table D-1. PM_{10} emission factors for road transport (g/km) for 1990 used in the TNO study (Berdowski et al., 1997a) and in the Dutch Emission Inventory (CBS, 1998) with TNO applied average emission factors for gasoline and diesel cars combined

		TNO (Berdowski et al., 1997a) g/km	Dutch Emission Inventory (CBS, 1998) G/km
Exhaust			
	HDV diesel	0,830	0,969
	HDV gasoline	0,830	0,400
	LDV diesel	0,050	0,434
	LDV gasoline	0,050	0,055
	LDV gasoline 2stroke	0,610	0,610
	Motorbikes<50cc	0,017	0,040
	Motorbikes>50cc	0,050	0,120
	Pass. Cars 2-stroke	0,610	0,610
	Pass. Cars 4-stroke	0,050	0,026
	Pass. Cars diesel	0,050	0,242
Non-exhaust			
	HDV diesel	1,170	0,038
	HDV gasoline	1,170	0,038
	LDV diesel	0,070	0,009
	LDV gasoline	0,070	0,009
	LDV gasoline 2stroke	0,070	0,009
	Motorbikes<50cc	0,023	0,002
	Motorbikes>50cc	0,023	0,004
	Pass. Cars 2-stroke	0,070	0,007
	Pass. Cars 4-stroke	0,070	0,007
	Pass. Cars diesel	0,070	0,007

Table D-2. PM_{10} emissions in 1990 and 2010 for road transport (SNAP7) according to different calculations

		1990 Ktonne	2010 BL Ktonne	Change 1990-2010 %	Change 1990-2010 Ktonne
TNO data set	Exhaust	281	168	- 40%	- 113
	Non-exhaust	369	541	+ 47%	+ 172
	Total	**650**	**710**	**+ 9%**	**+ 60**
Revised data set	Exhaust	619	147	- 76%	- 472
(used in 'EC, 2000'	Non-exhaust	32	48	+ 50%	+ 16
And 'EEA, 1999')	**Total**	**651**	**195**	**- 70%**	**- 456**
Improved data set 1	Exhaust	272	65	- 76%	- 207
	Non-exhaust	22	31	+ 42%	+ 9
	Total	**294**	**95**	**- 68%**	**- 198**
Improved data set 2	Exhaust	497 (373)*	143 (107)*	- 71% (- 71%)*	- 354 (- 266)*
	Non-exhaust	22	31	+ 41%	+ 9
	Total	**519 (395)***	**174 (138)***	**- 66% (- 65%)***	**- 345 (- 257)***

** Figures between brackets are based on the assumption that 75% of measured tailpipe emissions are emitted to the air and the remainig to soil and water. These figures are presented here because they have been reported in earlier versions of this reprt. The factor 0.75 is typical for the PM_{10}-inventory of the Netherlands, Within the Netherlands the use of this factor is under discussion and will probably be skippped because there is no sound scientific basis for the introduction of such a factor.*

Appendix E: 1990 and 2010 emission factors for 23 European agglomerations

NOx	Emissions City			Emissions g/km2 City		
City	1990	2010	ECE	1990	2010	ECE
Amsterdam	40	19	12	69450	29634	19099
Athens	93	91	61	265947	258448	173689
Barcelona	88	63	36	192730	150484	87090
Berlin	127	48	35	144558	56502	41079
Birmingham	108	44	23	721943	301719	157475
Brussels	41	22	12	100864	66215	37243
Cologne	110	45	33	315622	115118	83695
Copenhagen	70	34	22	105028	64907	42991
Dublin	30	19	8	261415	93567	40776
Helsinki	52	36	19	216903	110368	58609
Lisbon	36	42	27	361238	386936	250719
London	497	207	108	314850	131012	68379
Luxembourg	4	2	1	198571	91904	50959
Lyon	35	18	12	235787	109257	73071
Madrid	126	90	52	970363	692798	400945
Manchester	107	44	23	383561	180303	94105
Marseilles	34	16	11	458165	140256	93803
Milan	163	87	54	347693	182615	113982
Paris	261	126	84	217293	92102	61597
Rome	132	75	47	1053658	0	0
Stockholm	71	38	27	214905	186504	134175
Stuttgart	96	40	29	480765	41993	30530
Vienna	59	29	21	279909	113309	82390

PB	Emissions City			Emissions/km2 City		
City	1990	2010	ECE	1990	2010	ECE
Amsterdam	27	27	0	45899	41865	0
Athens	154	224	49	439365	635818	139170
Barcelona	349	354	278	764700	851561	669321
Berlin	68	54	25	76844	63423	29694
Birmingham	59	57	0	396201	384853	0
Brussels	36	36	39	89650	106966	115151
Cologne	59	51	22	167777	129219	56402
Copenhagen	18	17	0	27388	33101	0
Dublin	6	6	0	51638	28573	0
Helsinki	33	44	0	136626	134406	0
Lisbon	106	236	17	1060283	2170462	157154
London	273	264	0	172789	167110	0
Luxembourg	13	12	0	568849	583937	0
Lyon	39	39	0	259903	235311	0
Madrid	501	512	399	3850131	3920422	3055219
Manchester	59	56	0	210498	229983	0
Marseilles	38	35	0	505026	302075	0
Milan	131	135	48	278970	283791	101100
Oslo	10	10	0	25871	21265	0
Paris	287	271	0	239518	198363	0
Rome	106	117	39	845397		
Bucharest	52	55	30	284074	330472	183454
Budapest	273	571	39	520613	884983	60560
Prague	39	36	20	186419	166778	94247
Sofia	42	47	18	209419	257810	99840

SO2	Emissions City			Emissions g/km2 City		
City	1990	2010	ECE	1990	2010	ECE
Amsterdam	14	4	3.91	24759	6170	6170
Athens	155	111	15.46	443297	314049	43930
Barcelona	169	71	13.90	370713	171668	33412
Berlin	221	22	19.32	250945	25177	22596
Birmingham	150	37	10.00	1002496	254352	67855
Brussels	37	20	7.19	90832	60732	21477
Cologne	192	20	18.00	547902	51296	46038
Copenhagen	47	22	8.44	70276	42472	16144
Dublin	47	35	7.92	404685	176780	39908
Helsinki	45	27	14.16	187990	82529	43420
Lisbon	48	41	7.69	475440	373990	70621
London	691	174	46.54	437203	110444	29464
Luxembourg	3	1	0.46	120869	35450	21713
Lyon	29	14	4.66	193093	82168	28153
Madrid	243	103	20.08	1866474	790326	153821
Manchester	149	37	9.92	532617	151997	40549
Marseilles	28	12	4.22	375204	105481	36141
Milan	134	37	17.98	285016	77619	37861
Paris	214	95	32.44	177947	69266	23733
Rome	108	32	15.56	863721		

Appendix F: 1990 en 2010 urban emission factors for 15 EU countries

1990		SO2 ton p	SO2 pop> 750.000	Nox kg/inh	Nox pop> 750.000	Benzene	Benzene pop> 750.000	VOC	Pb g/inh	Pb pop> 750.000	PM10	PM10 pop> 750.000	PM10antr. kg/inh	PM10antr. pop> 750.000	PM10sec	PM10sec pop> 750.000
		Total		Total		Total		Total	Total		Total		Total		Total	
eu	Austria	12	12	30	19	0,89	0,94	46	28	12	4,0	55	3,9	2,4	71	52
eu	Belgium	32	32	36	23	0,67	0,54	38	72	4,9	7,1	100	7,0	3,6	116	96
eu	Denmark	36	35	52	46	1,29	1,26	35	35	3,0	6,1	143	5,9	4,3	150	139
eu	Finland	47	37	56	31	1,09	0,91	43	43	32	8,8	123	8,6	3,6	175	119
eu	France	23	22	28	21	1,11	0,99	42	77	14	6,3	77	6,2	2,5	87	74
eu	Germany	67	14	34	23	0,89	0,58	39	29	2,9	16	66	16	4,5	178	62
eu	Greece	52	63	39	49	1,01	1,05	36	50	4,0	5,1	196	5,0	2,5	158	193
eu	Ireland	49	50	29	26	1,09	1,16	31	38	5,5	8,5	139	8,4	5,7	138	133
eu	Italy	30	30	36	27	0,86	0,87	36	30	26	4,7	100	4,6	1,9	112	98
eu	Luxembourg	- 37	-	58	-	1,31	-	50	193	-	16	-	15	-	160	-
eu	Netherlands	14	13	36	26	0,95	0,79	33	18	0,2	3,9	68	3,8	4,3	83	64
eu	Portugal	29	24	21	21	0,68	0,87	22	65	23	2,6	80	2,5	1,9	88	78
eu	Spain	56	58	30	24	1,02	0,95	26	46	32	3,8	147	3,7	2,0	152	145
eu	Sweden	14	12	40	27	1,26	1,02	59	62	6,8	4,3	67	4,2	2,3	89	64
eu	United Kingdom	66	65	49	37	0,96	0,83	46	47	1,7	4,5	183	4,3	2,1	200	180
Total/average EU		**45**	**24**	**36**	**28**	**0,96**	**0,55**	**39**	**45**	**12**	**7,3**	**118,3**	**7,2**	**2,7**	**141**	**116**
2010 BL		ton														
eu	Austria	6,1	6,7	11	8,1	0,43	0,75	25			3,4	27	3,4	2,1	28	25
eu	Belgium	19	18	19	14	0,25	0,44	20			5,9	59	5,8	2,9	65	56
eu	Denmark	17	17	29	30	0,76	0,85	18			4,0	83	4,0	3,2	78	79
eu	Finland	21	14	29	17	0,70	0,52	20			7,9	55	7,8	2,0	86	53
eu	France	8,5	7,5	13	11	0,48	0,69	20			3,0	32	2,9	1,7	37	31
eu	Germany	8,5	4,5	14	11	0,40	0,42	15			7,5	28	7,5	2,2	38	26
eu	Greece	34	44	31	42	0,48	0,70	18			4,2	151	4,1	2,4	111	148
eu	Ireland	25	26	19	21	0,64	0,92	13			6,4	86	6,4	4,6	77	81
eu	Italy	11	10	21	16	0,36	0,64	21			3,3	46	3,2	1,5	53	44
eu	Luxembourg	- 8,8	-	24	-	0,58	-	15			10,0	-	10,0	-	54	-
eu	Netherlands	3,3	1,7	16	12	0,49	0,68	15			2,2	25	2,1	3,4	32	22
eu	Portugal	15	9,8	20	17	0,40	0,52	14			2,3	46	2,2	1,3	59	44
eu	Spain	20	19	22	19	0,56	0,50	17			2,8	66	2,7	1,5	72	65
eu	Sweden	9,3	8,1	24	19	0,53	0,70	23			3,1	46	3,0	1,7	56	45
eu	United Kingdom	17	16	20	16	0,35	0,69	21			2,5	55	2,4	1,3	62	54
Total/average EU		**13**	**8**	**18**	**16**	**0,44**	**0,38**	**19**			**4,1**	**51**	**4,1**	**1,8**	**52**	**49**

2010-TD		SO2 kg/inh Total	SO2 pop> 750.000	Nox kg/inh Total	Nox pop> 750.000	Benzene Total	Benzene pop> 750.000	VOC Total	PM10 Total	PM10 pop> 750.000	PM10antr. kg/inh Total	PM10antr. pop> 750.000	PM10sec Total	PM10sec pop> 750.000
eu	Austria	5,0	5,6	10,1	7,3	0,38	0,73	0,0	2,3	24	2,2	1,6	25	22
eu	Belgium	6,3	6,2	10,4	6,8	0,25	0,44	0,0	3,9	24	3,9	2,2	28	22
eu	Denmark	6,7	6,5	20	19	0,70	0,80	0,0	3,0	45	3,0	2,4	45	43
eu	Finland	11,3	7,4	17	8,5	0,66	0,48	0,0	4,0	29	3,9	1,6	47	27
eu	France	3,0	2,7	8,2	6,6	0,44	0,67	0,0	1,9	17	1,9	1,3	19	16
eu	Germany	5,5	2,6	9,5	7,8	0,37	0,41	0,0	2,6	18	2,6	1,2	25	17
eu	Greece	6,0	8,3	18	26	0,46	0,68	0,0	2,5	58	2,4	1,8	39	56
eu	Ireland	9,5	9,7	8,7	7,9	0,57	0,87	0,0	4,6	34	4,6	3,9	32	31
eu	Italy	4,9	4,7	13	9,2	0,34	0,63	0,0	1,8	24	1,8	1,1	29	23
eu	Luxembourg	4,4	-	149	-	0,55	-	0,0	7,1	-	6,9	-	244	-
eu	Netherlands	2,7	1,5	10	7,4	0,44	0,67	0,0	1,3	16	1,3	1,7	21	14
eu	Portugal	3,8	2,4	10	8,5	0,39	0,51	0,0	1,6	19	1,6	1,0	23	18
eu	Spain	5,1	4,7	14	10	0,54	0,48	0,0	1,6	26	1,6	1,0	31	25
eu	Sweden	7,6	6,7	17	13	0,45	0,67	0,0	2,1	34	2,1	1,3	41	33
eu	United Kingdom	5,0	4,2	11	7,8	0,28	0,67	0,0	1,5	21	1,4	0,8	26	20
Total/average EU		**5**	**3**	**11**	**9,2**	**0,40**	**0,37**	**0,0**	**2,1**	**23,7**	**2,1**	**1,2**	**27**	**22**
2010-AP-NT														
eu	Austria	4,3	4,5	11	9,1			19	2,3	24	2,3	1,1	25	23
eu	Belgium	5,6	4,1	9,5	5,5			9,3	4,5	18	4,5	1,5	25	16
eu	Denmark	11	11	25	25			16	3,6	63	3,6	4,2	60	59
eu	Finland	22	16	24	14			24	5,3	53	5,2	1,8	78	51
eu	France	3,9	4,0	10	7,2			14	2,5	20	2,5	1,2	24	19
eu	Germany	4,6	1,2	11	7,9			11	4,3	16	4,3	1,1	27	15
eu	Greece	34	53	23	36			16	4,1	156	4,0	1,8	99	154
eu	Ireland	6,7	6,3	14	11			13	5,4	33	5,4	3,9	34	29
eu	Italy	2,9	2,3	15	9,8			19	2,1	20	2,0	0,6	28	20
eu	Luxembourg	8,8	-	12	-			13	9,6	-	9,5	-	36	-
eu	Netherlands	2,3	1,5	11	8,0			9,1	1,9	19	1,9	3,3	22	15
eu	Portugal	8,9	5,3	13	10			10	1,5	27	1,5	0,7	36	26
eu	Spain	19	18	17	13			15	2,1	54	2,0	0,8	62	54
eu	Sweden	6,1	5,3	19	15			23	2,8	35	2,7	1,8	41	34
eu	United Kingdom	6,4	5,6	18	13			17	1,9	32	1,8	0,7	41	31
Total/average EU		**7**	**5**	**14**	**12**			**15**	**2,8**	**35**	**2,8**	**1,1**	**36**	**33**

		SO2		Nox		VOC	PM10		PM10ant		PM10sec	
	2010-AP-FT	kg/inh	kg/inh	kg/inh	kg/inh	kg/inh	kg/inh	kg/inh	kg/inh	kg/inh	kg/inh	kg/inh
		Coun-try	pop >750.000	Coun-try	pop >750.000	Coun-try	Coun-try	pop >750.000	Coun-try	pop >750.000	Coun-try	pop >750.000
eu	Austria	4,3	4,5	10,9	9,1	16,8	1,6	23	1,5	0,6	25	23
eu	Belgium	5,6	4,1	9,5	5,5	9,6	3,3	17	3,3	1,0	25	16
eu	Denmark	10,8	10,6	25,5	24,6	16,0	2,0	60	1,9	1,2	60	59
eu	Finland	21,5	15,6	23,8	13,7	23,7	3,7	52	3,6	0,8	78	51
eu	France	3,9	4,0	10,4	7,2	13,7	1,7	20	1,7	0,9	24	19
eu	Germany	4,6	1,2	11,5	7,9	11,1	2,0	15	2,0	0,6	27	15
eu	Greece	34,1	52,6	22,6	35,7	15,5	2,4	155	2,3	0,8	99	154
eu	Ireland	6,7	6,3	13,6	11,0	13,0	4,4	31	4,4	2,1	34	29
eu	Italy	2,9	2,3	14,6	9,8	17,8	1,3	20	1,3	0,3	28	20
eu	Luxembourg	8,8	-	12,2	-	13,2	6,8	-	6,8	-	36	-
eu	Netherlands	2,3	1,5	11,4	8,0	9,3	1,2	17	1,2	1,9	22	15
eu	Portugal	8,9	5,3	12,5	10,3	10,2	1,0	27	1,0	0,5	36	26
eu	Spain	18,7	17,6	17,3	13,2	15,4	1,3	54	1,2	0,4	62	54
eu	Sweden	6,1	5,3	19,1	14,9	18,4	1,9	35	1,9	1,0	41	34
eu	United Kingdom	6,4	5,6	18,2	13,3	17,1	1,2	32	1,2	0,4	41	31
	Total/average EU	7	5	14	11,8	14,4	1,7	34,0	1,7	0,6	36	33

Appendix G: 1990 en 2010 concentration, exceedances and exposure for various pollutants for 15 EU countries

Lead (1990)	Concentration yearly average		Regional		Agglomerations	
ug/m3	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	0,09	0,28	0,00	0,00	0,00	0,00
Belgium	0,04	0,14	0,00	0,00	0,00	0,00
Denmark	0,02	0,06	0,00	0,00	0,00	0,00
Finland	0,12	0,36	0,00	0,00	0,00	0,00
France	0,20	0,63	0,00	0,00	0,48	70,56
Germany	0,04	0,14	0,00	0,00	0,00	0,00
Greece	0,33	1,03	0,00	0,00	1,62	100,00
Ireland	0,02	0,07	0,00	0,00	0,00	0,00
Italy	0,23	0,73	0,00	0,00	0,75	100,00
Luxembourg	0,03	0,08	0,00	0,00	0,00	0,00
Netherlands	0,01	0,04	0,00	0,00	0,00	0,00
Portugal	0,05	0,16	0,00	0,00	0,00	0,00
Spain	0,32	0,98	0,00	0,00	1,46	100,00
Sweden	0,04	0,13	0,00	0,00	0,00	0,00
United	0,05	0,15	0,00	0,00	0,00	0,00

1990	target value: daily maximum 125 ug/m3 --> annual average >24 ug/m3					
SO2	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	18	45	0,00	0,00	1,36	100,00
Belgium	16	27	0,00	0,00	0,38	50,01
Denmark	6	14	0,00	0,00	0,00	0,00
Finland	6	32	0,00	0,00	0,68	89,98
France	10	43	0,00	0,00	1,27	100,00
Germany	25	41	0,27	35,33	1,14	100,00
Greece	12	56	0,00	0,00	1,98	100,00
Ireland	5	18	0,00	0,00	0,00	0,00
Italy	8	56	0,00	0,00	2,01	100,00
Luxembourg	9	10	0,00	0,00	0,00	0,00
Netherlands	10	13	0,00	0,00	0,00	0,00
Portugal	6	9	0,00	0,00	0,00	0,00
Spain	7	35	0,00	0,00	0,79	100,00
Sweden	3	9	0,00	0,00	0,00	0,00
United Kingdom	25	44	0,27	35,38	1,32	100,00
2010/BL						
Austria	10	25	0,00	0,00	0,29	38,67
Belgium	10	18	0,00	0,00	0,00	0,00
Denmark	3	6	0,00	0,00	0,00	0,00
Finland	2	12	0,00	0,00	0,00	0,00
France	3	13	0,00	0,00	0,00	0,00
Germany	6	12	0,00	0,00	0,00	0,00
Greece	7	32	0,00	0,00	0,66	87,29
Ireland	3	12	0,00	0,00	0,00	0,00
Italy	3	20	0,00	0,00	0,00	0,00
Luxembourg	1	2	0,00	0,00	0,00	0,00
Netherlands	2	2	0,00	0,00	0,00	0,00
Portugal	3	4	0,00	0,00	0,00	0,00
Spain	3	17	0,00	0,00	0,00	0,00
Sweden	1	5	0,00	0,00	0,00	0,00
United Kingdom	6	11	0,00	0,00	0,00	0,00
2010/AP-NT						
Austria	6	13	0,00	0,00	0,00	0,00
Belgium	2	5	0,00	0,00	0,00	0,00
Denmark	2	4	0,00	0,00	0,00	0,00
Finland	3	11	0,00	0,00	0,00	0,00
France	2	7	0,00	0,00	0,00	0,00
Germany	2	4	0,00	0,00	0,00	0,00
Greece	10	40	0,00	0,00	1,11	100,00
Ireland	1	4	0,00	0,00	0,00	0,00
Italy	1	8	0,00	0,00	0,00	0,00
Luxembourg	2	2	0,00	0,00	0,00	0,00
Netherlands	1	2	0,00	0,00	0,00	0,00
Portugal	1	2	0,00	0,00	0,00	0,00
Spain	2	14	0,00	0,00	0,00	0,00
Sweden	1	3	0,00	0,00	0,00	0,00
United Kingdom	2	6	0,00	0,00	0,00	0,00

2010/AP-FT SO2	target value: daily maximum 125 ug/m3 --> annual average >24 ug/m3					
	Regional	Agglomerations	Regional		Agglomerations	
	Concentration yearly average		max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	6	13	0,00	0,00	0,00	0,00
Belgium	2	5	0,00	0,00	0,00	0,00
Denmark	2	4	0,00	0,00	0,00	0,00
Finland	3	11	0,00	0,00	0,00	0,00
France	2	7	0,00	0,00	0,00	0,00
Germany	2	4	0,00	0,00	0,00	0,00
Greece	10	40	0,00	0,00	1,11	100,00
Ireland	1	4	0,00	0,00	0,00	0,00
Italy	1	8	0,00	0,00	0,00	0,00
Luxembourg	2	2	0,00	0,00	0,00	0,00
Netherlands	1	2	0,00	0,00	0,00	0,00
Portugal	1	2	0,00	0,00	0,00	0,00
Spain	2	14	0,00	0,00	0,00	0,00
Sweden	1	3	0,00	0,00	0,00	0,00
United Kingdom	2	6	0,00	0,00	0,00	0,00
2010/TD						
Austria	9	21	0,00	0,00	0,06	8,12
Belgium	4	7	0,00	0,00	0,00	0,00
Denmark	1	3	0,00	0,00	0,00	0,00
Finland	2	10	0,00	0,00	0,00	0,00
France	2	7	0,00	0,00	0,00	0,00
Germany	4	7	0,00	0,00	0,00	0,00
Greece	2	9	0,00	0,00	0,00	0,00
Ireland	2	6	0,00	0,00	0,00	0,00
Italy	2	11	0,00	0,00	0,00	0,00
Luxembourg	1	2	0,00	0,00	0,00	0,00
Netherlands	2	2	0,00	0,00	0,00	0,00
Portugal	1	1	0,00	0,00	0,00	0,00
Spain	1	6	0,00	0,00	0,00	0,00
Sweden	1	4	0,00	0,00	0,00	0,00
United Kingdom	3	5	0,00	0,00	0,00	0,00

1990	Target value: annual average 40 ug/m3					
NO2	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	21	57	0,00	0,00	0,60	100,00
Belgium	26	46	0,00	0,00	0,28	66,54
Denmark	11	34	0,00	0,00	0,00	0,00
Finland	11	60	0,00	0,00	0,67	100,00
France	22	59	0,00	0,00	0,66	100,00
Germany	26	56	0,00	0,00	0,57	100,00
Greece	25	47	0,00	0,00	0,33	77,27
Ireland	4	17	0,00	0,00	0,00	0,00
Italy	33	79	0,00	0,00	1,18	100,00
Luxembourg	17	33	0,00	0,00	0,00	0,00
Netherlands	23	43	0,00	0,00	0,19	45,64
Portugal	5	17	0,00	0,00	0,00	0,00
Spain	11	45	0,00	0,00	0,26	60,06
Sweden	17	46	0,00	0,00	0,30	69,53
United Kingdom	28	54	0,00	0,00	0,50	100,00
2010/BL						
Austria	10	34	0,00	0,00	0,00	0,00
Belgium	16	31	0,00	0,00	0,00	0,00
Denmark	7	24	0,00	0,00	0,00	0,00
Finland	5	43	0,00	0,00	0,22	51,28
France	12	42	0,00	0,00	0,19	44,57
Germany	14	39	0,00	0,00	0,09	21,08
Greece	24	47	0,00	0,00	0,32	75,30
Ireland	3	11	0,00	0,00	0,00	0,00
Italy	20	62	0,00	0,00	0,74	100,00
Luxembourg	8	17	0,00	0,00	0,00	0,00
Netherlands	13	27	0,00	0,00	0,00	0,00
Portugal	4	16	0,00	0,00	0,00	0,00
Spain	8	38	0,00	0,00	0,08	19,28
Sweden	11	34	0,00	0,00	0,00	0,00
United Kingdom	13	32	0,00	0,00	0,00	0,00
2010/AP-NT						
Austria	9	38	0,00	0,00	0,08	17,59
Belgium	7	19	0,00	0,00	0,00	0,00
Denmark	5	20	0,00	0,00	0,00	0,00
Finland	4	37	0,00	0,00	0,05	11,74
France	8	37	0,00	0,00	0,04	8,62
Germany	9	33	0,00	0,00	0,00	0,00
Greece	18	43	0,00	0,00	0,19	45,59
Ireland	2	9	0,00	0,00	0,00	0,00
Italy	15	57	0,00	0,00	0,60	100,00
Luxembourg	4	11	0,00	0,00	0,00	0,00
Netherlands	8	20	0,00	0,00	0,00	0,00
Portugal	2	12	0,00	0,00	0,00	0,00
Spain	5	34	0,00	0,00	0,00	0,00
Sweden	7	27	0,00	0,00	0,00	0,00
United Kingdom	10	29	0,00	0,00	0,00	0,00

2010/AP-FT NO2	Target value: annual average 40 ug/m3					
	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	9	38	0,00	0,00	0,08	17,59
Belgium	7	19	0,00	0,00	0,00	0,00
Denmark	5	20	0,00	0,00	0,00	0,00
Finland	4	37	0,00	0,00	0,05	11,74
France	8	37	0,00	0,00	0,04	8,62
Germany	9	33	0,00	0,00	0,00	0,00
Greece	18	43	0,00	0,00	0,19	45,59
Ireland	2	9	0,00	0,00	0,00	0,00
Italy	15	57	0,00	0,00	0,60	100,00
Luxembourg	4	11	0,00	0,00	0,00	0,00
Netherlands	8	20	0,00	0,00	0,00	0,00
Portugal	2	12	0,00	0,00	0,00	0,00
Spain	5	34	0,00	0,00	0,00	0,00
Sweden	7	27	0,00	0,00	0,00	0,00
United Kingdom	10	29	0,00	0,00	0,00	0,00
2010/TD						
Austria	9	34	0,00	0,00	0,00	0,00
Belgium	10	21	0,00	0,00	0,00	0,00
Denmark	5	18	0,00	0,00	0,00	0,00
Finland	4	36	0,00	0,00	0,03	6,89
France	9	35	0,00	0,00	0,00	0,00
Germany	10	31	0,00	0,00	0,00	0,00
Greece	17	37	0,00	0,00	0,05	11,41
Ireland	2	7	0,00	0,00	0,00	0,00
Italy	16	57	0,00	0,00	0,58	100,00
Luxembourg	5	10	0,00	0,00	0,00	0,00
Netherlands	9	19	0,00	0,00	0,00	0,00
Portugal	3	10	0,00	0,00	0,00	0,00
Spain	6	33	0,00	0,00	0,00	0,00
Sweden	8	28	0,00	0,00	0,00	0,00
United Kingdom	9	22	0,00	0,00	0,00	0,00

1990	Target Values: annual average 5 ug/m3					
benzene	**Concentration yearly average**		**Regional**		**Agglomerations**	
	Regional	**Agglomerations**	**max ex-ceedance**	**cum exposure**	**max ex-ceedance**	**cum exposure**
Austria	2,8	11,0	0,00	0,00	1,83	100,00
Belgium	1,9	2,9	0,00	0,00	0,00	0,00
Denmark	0,8	2,1	0,00	0,00	0,00	0,00
Finland	1,0	8,3	0,00	0,00	1,05	100,00
France	2,9	10,0	0,00	0,00	1,54	100,00
Germany	2,6	6,4	0,00	0,00	0,51	76,12
Greece	2,0	13,3	0,00	0,00	2,48	100,00
Ireland	0,6	2,2	0,00	0,00	0,00	0,00
Italy	4,7	23,5	0,02	3,49	5,35	100,00
Luxembourg	2,1	3,0	0,00	0,00	0,00	0,00
Netherlands	1,6	2,7	0,00	0,00	0,00	0,00
Portugal	0,5	1,3	0,00	0,00	0,00	0,00
Spain	1,5	6,5	0,00	0,00	0,53	79,09
Sweden	1,2	3,9	0,00	0,00	0,00	0,00
United Kingdom	1,9	4,4	0,00	0,00	0,00	0,00
2010/BL						
Austria	1,0	4,2	0,00	0,00	0,00	0,00
Belgium	0,6	1,0	0,00	0,00	0,00	0,00
Denmark	0,4	1,1	0,00	0,00	0,00	0,00
Finland	0,5	4,2	0,00	0,00	0,00	0,00
France	1,1	3,7	0,00	0,00	0,00	0,00
Germany	1,1	2,6	0,00	0,00	0,00	0,00
Greece	0,7	4,6	0,00	0,00	0,00	0,00
Ireland	0,3	1,1	0,00	0,00	0,00	0,00
Italy	1,2	6,0	0,00	0,00	0,39	57,93
Luxembourg	0,9	1,3	0,00	0,00	0,00	0,00
Netherlands	0,8	1,4	0,00	0,00	0,00	0,00
Portugal	0,2	0,5	0,00	0,00	0,00	0,00
Spain	0,6	2,8	0,00	0,00	0,00	0,00
Sweden	0,5	1,5	0,00	0,00	0,00	0,00
United Kingdom	0,6	1,4	0,00	0,00	0,00	0,00
2010/TD						
Austria	0,8	3,2	0,00	0,00	0,00	0,00
Belgium	0,9	0,8	0,00	0,00	0,00	0,00
Denmark	0,8	0,9	0,00	0,00	0,00	0,00
Finland	0,8	3,5	0,00	0,00	0,00	0,00
France	0,8	2,9	0,00	0,00	0,00	0,00
Germany	0,7	1,9	0,00	0,00	0,00	0,00
Greece	0,9	3,9	0,00	0,00	0,00	0,00
Ireland	0,6	0,7	0,00	0,00	0,00	0,00
Italy	0,8	5,0	0,00	0,00	0,10	15,66
Luxembourg	0,9	1,2	0,00	0,00	0,00	0,00
Netherlands	0,6	0,8	0,00	0,00	0,00	0,00
Portugal	0,9	0,4	0,00	0,00	0,00	0,00
Spain	0,9	2,6	0,00	0,00	0,00	0,00
Sweden	0,8	1,2	0,00	0,00	0,00	0,00
United Kingdom	0,7	1,0	0,00	0,00	0,00	0,00

1990	20 days>8h120 ug/3					
Ozone	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	47	33	1,37	100,00	0,66	100,00
Belgium	81	57	3,05	100,00	1,84	100,00
Denmark	28	20	0,42	100,00	0,00	0,00
Finland	5	3	0,00	0,00	0,00	0,00
France	68	48	2,41	100,00	1,38	100,00
Germany	66	47	2,32	100,00	1,33	100,00
Greece	33	23	0,64	100,00	0,15	100,00
Ireland	19	13	0,00	0,00	0,00	0,00
Italy	44	31	1,19	100,00	0,53	100,00
Luxembourg	96	67	3,81	100,00	2,37	100,00
Netherlands	50	35	1,49	100,00	0,74	100,00
Portugal	44	31	1,20	100,00	0,54	100,00
Spain	40	28	1,01	100,00	0,40	100,00
Sweden	8	6	0,00	0,00	0,00	0,00
United Kingdom	25	18	0,26	100,00	0,00	0,00
2010/BL						
Austria	25	17	0,24	100,00	0,00	0,00
Belgium	51	36	1,55	100,00	0,78	100,00
Denmark	13	9	0,00	0,00	0,00	0,00
Finland	3	2	0,00	0,00	0,00	0,00
France	32	23	0,62	100,00	0,13	100,00
Germany	34	24	0,69	100,00	0,18	100,00
Greece	21	15	0,07	100,00	0,00	0,00
Ireland	12	8	0,00	0,00	0,00	0,00
Italy	25	17	0,23	100,00	0,00	0,00
Luxembourg	48	33	1,38	100,00	0,66	100,00
Netherlands	33	23	0,64	100,00	0,15	100,00
Portugal	32	22	0,58	100,00	0,11	100,00
Spain	23	16	0,13	100,00	0,00	0,00
Sweden	4	3	0,00	0,00	0,00	0,00
United Kingdom	20	14	0,01	100,00	0,00	0,00
2010/AP-NT						
Austria	25	18	0,27	100,00	0,00	0,00
Belgium	41	29	1,07	100,00	0,45	100,00
Denmark	12	8	0,00	0,00	0,00	0,00
Finland	3	2	0,00	0,00	0,00	0,00
France	24	17	0,20	100,00	0,00	0,00
Germany	27	19	0,36	100,00	0,00	0,00
Greece	21	15	0,05	100,00	0,00	0,00
Ireland	2	1	0,00	0,00	0,00	0,00
Italy	21	15	0,06	100,00	0,00	0,00
Luxembourg	39	27	0,95	100,00	0,37	100,00
Netherlands	27	19	0,35	100,00	0,00	0,00
Portugal	30	21	0,52	100,00	0,07	100,00
Spain	15	11	0,00	0,00	0,00	0,00
Sweden	4	3	0,00	0,00	0,00	0,00
United Kingdom	16	11	0,00	0,00	0,00	0,00

2010/AP-FT Ozone	Concentration yearly average		20 days>8h120 ug/3 Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	26	18	0,30	40,14	0,00	0,00
Belgium	41	29	1,17	100,00	0,48	63,91
Denmark	12	8	0,00	0,00	0,00	0,00
Finland	3	2	0,00	0,00	0,00	0,00
France	23	16	0,18	23,61	0,00	0,00
Germany	27	19	0,38	49,96	0,00	0,00
Greece	21	15	0,07	9,32	0,00	0,00
Ireland	0	0	0,00	0,00	0,00	0,00
Italy	21	15	0,07	8,99	0,00	0,00
Luxembourg	38	27	0,99	100,00	0,36	46,93
Netherlands	27	19	0,36	47,08	0,00	0,00
Portugal	31	22	0,62	82,10	0,10	12,92
Spain	14	10	0,00	0,00	0,00	0,00
Sweden	4	3	0,00	0,00	0,00	0,00
United Kingdom	16	11	0,00	0,00	0,00	0,00
2010/TD						
Austria	20	14	0,00	0,00	0,00	0,00
Belgium	36	25	0,78	100,00	0,25	100,00
Denmark	8	6	0,00	0,00	0,00	0,00
Finland	2	2	0,00	0,00	0,00	0,00
France	21	14	0,03	100,00	0,00	0,00
Germany	23	16	0,16	100,00	0,00	0,00
Greece	16	11	0,00	0,00	0,00	0,00
Ireland	8	6	0,00	0,00	0,00	0,00
Italy	16	11	0,00	0,00	0,00	0,00
Luxembourg	30	21	0,52	100,00	0,06	100,00
Netherlands	24	17	0,18	100,00	0,00	0,00
Portugal	21	15	0,07	100,00	0,00	0,00
Spain	15	11	0,00	0,00	0,00	0,00
Sweden	3	2	0,00	0,00	0,00	0,00
United Kingdom	14	10	0,00	0,00	0,00	0,00

1990 PM10	PM10, seasalt and soil resuspension included, target value: annual average >40 ug/m3					
	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	43	59	0,22	86,30	0,76	100,00
Belgium	37	43	0,02	8,83	0,23	87,01
Denmark	20	23	0,00	0,00	0,00	0,00
Finland	17	32	0,00	0,00	0,00	0,00
France	36	49	0,00	0,00	0,43	100,00
Germany	45	53	0,30	100,00	0,56	100,00
Greece	54	66	0,60	100,00	1,00	100,00
Ireland	23	27	0,00	0,00	0,00	0,00
Italy	41	60	0,16	61,64	0,80	100,00
Luxembourg	31	33	0,00	0,00	0,00	0,00
Netherlands	37	42	0,01	3,56	0,19	75,05
Portugal	23	24	0,00	0,00	0,00	0,00
Spain	31	37	0,00	0,00	0,00	0,00
Sweden	13	19	0,00	0,00	0,00	0,00
United Kingdom	24	26	0,00	0,00	0,00	0,00
2010/BL						
Austria	29	41	0,00	0,00	0,14	55,34
Belgium	30	34	0,00	0,00	0,00	0,00
Denmark	16	18	0,00	0,00	0,00	0,00
Finland	13	21	0,00	0,00	0,00	0,00
France	27	35	0,00	0,00	0,00	0,00
Germany	29	33	0,00	0,00	0,00	0,00
Greece	49	56	0,43	100,00	0,68	100,00
Ireland	20	23	0,00	0,00	0,00	0,00
Italy	31	40	0,00	0,00	0,10	40,82
Luxembourg	20	21	0,00	0,00	0,00	0,00
Netherlands	25	28	0,00	0,00	0,00	0,00
Portugal	23	23	0,00	0,00	0,00	0,00
Spain	28	31	0,00	0,00	0,00	0,00
Sweden	11	16	0,00	0,00	0,00	0,00
United Kingdom	15	17	0,00	0,00	0,00	0,00
2010/AP-NT						
Austria	27	33	0,00	0,00	0,00	0,00
Belgium	23	25	0,00	0,00	0,00	0,00
Denmark	15	18	0,00	0,00	0,00	0,00
Finland	13	21	0,00	0,00	0,00	0,00
France	24	30	0,00	0,00	0,00	0,00
Germany	26	28	0,00	0,00	0,00	0,00
Greece	49	54	0,42	100,00	0,60	100,00
Ireland	18	21	0,00	0,00	0,00	0,00
Italy	27	32	0,00	0,00	0,00	0,00
Luxembourg	18	18	0,00	0,00	0,00	0,00
Netherlands	24	27	0,00	0,00	0,00	0,00
Portugal	22	22	0,00	0,00	0,00	0,00
Spain	27	29	0,00	0,00	0,00	0,00
Sweden	10	16	0,00	0,00	0,00	0,00
United Kingdom	14	15	0,00	0,00	0,00	0,00

2010/AP-FT PM10	PM10, seasalt and soil resuspension included, target value: annual average >40 ug/m3					
	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	27	30	0,00	0,00	0,00	0,00
Belgium	23	25	0,00	0,00	0,00	0,00
Denmark	14	15	0,00	0,00	0,00	0,00
Finland	13	17	0,00	0,00	0,00	0,00
France	24	29	0,00	0,00	0,00	0,00
Germany	25	27	0,00	0,00	0,00	0,00
Greece	48	52	0,40	100,00	0,52	100,00
Ireland	18	19	0,00	0,00	0,00	0,00
Italy	27	31	0,00	0,00	0,00	0,00
Luxembourg	18	18	0,00	0,00	0,00	0,00
Netherlands	22	24	0,00	0,00	0,00	0,00
Portugal	22	22	0,00	0,00	0,00	0,00
Spain	27	29	0,00	0,00	0,00	0,00
Sweden	9	13	0,00	0,00	0,00	0,00
United Kingdom	14	14	0,00	0,00	0,00	0,00
2010/TD						
Austria	27	37	0,00	0,00	0,00	0,00
Belgium	25	28	0,00	0,00	0,00	0,00
Denmark	13	15	0,00	0,00	0,00	0,00
Finland	12	19	0,00	0,00	0,00	0,00
France	24	31	0,00	0,00	0,00	0,00
Germany	26	29	0,00	0,00	0,00	0,00
Greece	39	44	0,08	29,41	0,26	100,00
Ireland	18	20	0,00	0,00	0,00	0,00
Italy	28	35	0,00	0,00	0,00	0,00
Luxembourg	18	18	0,00	0,00	0,00	0,00
Netherlands	21	22	0,00	0,00	0,00	0,00
Portugal	21	22	0,00	0,00	0,00	0,00
Spain	26	28	0,00	0,00	0,00	0,00
Sweden	10	14	0,00	0,00	0,00	0,00
United Kingdom	13	14	0,00	0,00	0,00	0,00

1990 PM10	PM10, seasalt and soil resuspension included, target value: annual average >20 ug/m3					
	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	43	59	1,91	100,00	3,00	100,00
Belgium	37	43	1,50	100,00	1,91	100,00
Denmark	20	23	0,27	48,63	0,48	86,97
Finland	17	32	0,03	6,27	1,10	100,00
France	36	49	1,42	100,00	2,33	100,00
Germany	45	53	2,06	100,00	2,60	100,00
Greece	54	66	2,68	100,00	3,50	100,00
Ireland	23	27	0,47	84,27	0,75	100,00
Italy	41	60	1,78	100,00	3,09	100,00
Luxembourg	31	33	1,08	100,00	1,21	100,00
Netherlands	37	42	1,47	100,00	1,85	100,00
Portugal	23	24	0,52	93,35	0,58	100,00
Spain	31	37	1,08	100,00	1,44	100,00
Sweden	13	19	0,00	0,00	0,23	41,69
United Kingdom	24	26	0,53	96,18	0,72	100,00
2010/BL						
Austria	29	41	0,93	100,00	1,74	100,00
Belgium	30	34	0,95	100,00	1,24	100,00
Denmark	16	18	0,00	0,00	0,13	22,95
Finland	13	21	0,00	0,00	0,37	66,93
France	27	35	0,76	100,00	1,36	100,00
Germany	29	33	0,92	100,00	1,17	100,00
Greece	49	56	2,33	100,00	2,84	100,00
Ireland	20	23	0,29	52,20	0,50	90,11
Italy	31	40	1,07	100,00	1,67	100,00
Luxembourg	20	21	0,26	47,85	0,31	56,32
Netherlands	25	28	0,62	100,00	0,85	100,00
Portugal	23	23	0,45	81,47	0,48	87,24
Spain	28	31	0,86	100,00	1,04	100,00
Sweden	11	16	0,00	0,00	0,00	0,00
United Kingdom	15	17	0,00	0,00	0,03	5,21
2010/AP-NT						
Austria	27	33	0,77	100,00	1,21	100,00
Belgium	23	25	0,50	90,55	0,65	100,00
Denmark	15	18	0,00	0,00	0,13	23,09
Finland	13	21	0,00	0,00	0,34	62,33
France	24	30	0,56	100,00	0,99	100,00
Germany	26	28	0,68	100,00	0,85	100,00
Greece	49	54	2,30	100,00	2,68	100,00
Ireland	18	21	0,13	23,17	0,31	56,56
Italy	27	32	0,79	100,00	1,12	100,00
Luxembourg	18	18	0,12	22,61	0,14	25,36
Netherlands	24	27	0,55	99,52	0,78	100,00
Portugal	22	22	0,39	69,66	0,40	72,40
Spain	27	29	0,80	100,00	0,91	100,00
Sweden	10	16	0,00	0,00	0,00	0,00
United Kingdom	14	15	0,00	0,00	0,00	0,00

2010/AP-FT PM10	PM10, seasalt and soil resuspension included, target value: annual average >20 ug/m3					
	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	27	30	0,74	100,00	1,01	100,00
Belgium	23	25	0,47	85,02	0,60	100,00
Denmark	14	15	0,00	0,00	0,00	0,00
Finland	13	17	0,00	0,00	0,04	8,21
France	24	29	0,55	98,77	0,93	100,00
Germany	25	27	0,61	100,00	0,74	100,00
Greece	48	52	2,28	100,00	2,52	100,00
Ireland	18	19	0,11	20,16	0,21	38,13
Italy	27	31	0,78	100,00	1,06	100,00
Luxembourg	18	18	0,11	19,76	0,12	22,23
Netherlands	22	24	0,39	70,06	0,54	97,94
Portugal	22	22	0,38	69,06	0,39	71,29
Spain	27	29	0,79	100,00	0,88	100,00
Sweden	9	13	0,00	0,00	0,00	0,00
United Kingdom	14	14	0,00	0,00	0,00	0,00
2010/TD						
Austria	27	37	0,75	100,00	1,45	100,00
Belgium	25	28	0,59	100,00	0,82	100,00
Denmark	13	15	0,00	0,00	0,00	0,00
Finland	12	19	0,00	0,00	0,18	33,44
France	24	31	0,54	96,76	1,03	100,00
Germany	26	29	0,69	100,00	0,88	100,00
Greece	39	44	1,61	100,00	1,98	100,00
Ireland	18	20	0,12	21,69	0,30	53,93
Italy	28	35	0,85	100,00	1,31	100,00
Luxembourg	18	18	0,10	18,16	0,13	24,10
Netherlands	21	22	0,31	56,44	0,42	76,39
Portugal	21	22	0,37	66,02	0,39	70,30
Spain	26	28	0,73	100,00	0,87	100,00
Sweden	10	14	0,00	0,00	0,00	0,00
United Kingdom	13	14	0,00	0,00	0,00	0,00

1990 PM10	PM10 secundair aerosol Concentration yearly average	
	Regional	**Agglomerations**
Austria	25	28
Belgium	15	15
Denmark	9	9
Finland	5	6
France	15	16
Germany	20	22
Greece	20	21
Ireland	6	6
Italy	16	20
Luxembourg	16	17
Netherlands	14	14
Portugal	3	3
Spain	6	7
Sweden	6	7
United Kingdom	11	12

2010/BL		
Austria	11	12
Belgium	8	8
Denmark	5	5
Finland	2	3
France	6	7
Germany	8	8
Greece	15	16
Ireland	4	4
Italy	8	9
Luxembourg	5	6
Netherlands	4	4
Portugal	2	2
Spain	3	4
Sweden	4	5
United Kingdom	4	4

2010/AP-NT		
Austria	10,1	11,1
Belgium	2,8	2,8
Denmark	3,8	3,8
Finland	2,4	2,7
France	4,0	4,4
Germany	4,6	4,8
Greece	14,9	15,6
Ireland	1,7	1,7
Italy	4,2	5,5
Luxembourg	4,0	4,1
Netherlands	3,0	3,0
Portugal	1,1	1,1
Spain	2,6	2,9
Sweden	3,0	3,4
United Kingdom	2,8	2,9

2010/AP-FT PM10	PM10 secundair aerosol Concentration yearly average	
	Regional	**Agglomerations**
Austria	10,1	11,1
Belgium	2,8	2,8
Denmark	3,8	3,8
Finland	2,4	2,7
France	4,0	4,4
Germany	4,6	4,8
Greece	14,9	15,6
Ireland	1,7	1,7
Italy	4,2	5,5
Luxembourg	4,0	4,1
Netherlands	3,0	3,0
Portugal	1,1	1,1
Spain	2,6	2,9
Sweden	3,0	3,4
United Kingdom	2,8	2,9

2010/TD		
Austria	8,8	9,6
Belgium	3,7	3,8
Denmark	2,7	2,7
Finland	1,4	1,6
France	3,3	3,7
Germany	4,9	5,1
Greece	5,2	5,3
Ireland	1,5	1,5
Italy	4,7	6,0
Luxembourg	3,5	3,6
Netherlands	3,0	3,0
Portugal	0,7	0,7
Spain	1,6	1,9
Sweden	3,2	3,5
United Kingdom	1,9	1,9

1990 **PM10 anthropogenic emission, target value>annual average >40 ug/m3**

PM10	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	27,4	42,9	0,00	0,00	0,21	82,31
Belgium	18,8	24,7	0,00	0,00	0,00	0,00
Denmark	9,8	12,8	0,00	0,00	0,00	0,00
Finland	6,6	21,8	0,00	0,00	0,00	0,00
France	17,0	29,9	0,00	0,00	0,00	0,00
Germany	26,3	34,0	0,00	0,00	0,00	0,00
Greece	20,9	32,5	0,00	0,00	0,00	0,00
Ireland	7,1	11,1	0,00	0,00	0,00	0,00
Italy	18,9	37,4	0,00	0,00	0,02	8,92
Luxembourg	18,4	20,2	0,00	0,00	0,00	0,00
Netherlands	22,1	27,5	0,00	0,00	0,00	0,00
Portugal	3,1	4,0	0,00	0,00	0,00	0,00
Spain	6,9	12,1	0,00	0,00	0,00	0,00
Sweden	6,4	13,3	0,00	0,00	0,00	0,00
United Kingdom	13,0	15,6	0,00	0,00	0,00	0,00
2010/BL						
Austria	13,5	25,1	0,00	0,00	0,00	0,00
Belgium	11,1	15,2	0,00	0,00	0,00	0,00
Denmark	5,8	7,8	0,00	0,00	0,00	0,00
Finland	3,2	11,4	0,00	0,00	0,00	0,00
France	7,7	16,1	0,00	0,00	0,00	0,00
Germany	10,1	13,6	0,00	0,00	0,00	0,00
Greece	16,0	23,2	0,00	0,00	0,00	0,00
Ireland	4,6	7,5	0,00	0,00	0,00	0,00
Italy	8,8	17,3	0,00	0,00	0,00	0,00
Luxembourg	6,8	7,5	0,00	0,00	0,00	0,00
Netherlands	10,1	13,4	0,00	0,00	0,00	0,00
Portugal	2,1	2,6	0,00	0,00	0,00	0,00
Spain	3,8	6,3	0,00	0,00	0,00	0,00
Sweden	4,7	9,6	0,00	0,00	0,00	0,00
United Kingdom	4,7	5,8	0,00	0,00	0,00	0,00
2010/AP-NT						
Austria	11,3	17,5	0,00	0,00	0,00	0,00
Belgium	4,7	6,8	0,00	0,00	0,00	0,00
Denmark	4,6	7,8	0,00	0,00	0,00	0,00
Finland	3,2	11,1	0,00	0,00	0,00	0,00
France	4,8	11,0	0,00	0,00	0,00	0,00
Germany	6,7	9,2	0,00	0,00	0,00	0,00
Greece	15,6	20,9	0,00	0,00	0,00	0,00
Ireland	2,3	4,9	0,00	0,00	0,00	0,00
Italy	4,9	9,5	0,00	0,00	0,00	0,00
Luxembourg	4,8	5,1	0,00	0,00	0,00	0,00
Netherlands	9,1	12,4	0,00	0,00	0,00	0,00
Portugal	1,2	1,4	0,00	0,00	0,00	0,00
Spain	2,9	4,5	0,00	0,00	0,00	0,00
Sweden	3,5	9,5	0,00	0,00	0,00	0,00
United Kingdom	3,3	4,0	0,00	0,00	0,00	0,00

2010/AP-FT	PM10 anthropogenic emission, target value>annual average >40 ug/m3					
PM10	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	10,8	14,7	0,00	0,00	0,00	0,00
Belgium	4,3	6,1	0,00	0,00	0,00	0,00
Denmark	4,2	4,9	0,00	0,00	0,00	0,00
Finland	2,8	6,8	0,00	0,00	0,00	0,00
France	4,7	10,1	0,00	0,00	0,00	0,00
Germany	5,8	7,5	0,00	0,00	0,00	0,00
Greece	15,2	18,7	0,00	0,00	0,00	0,00
Ireland	2,0	3,4	0,00	0,00	0,00	0,00
Italy	4,7	8,7	0,00	0,00	0,00	0,00
Luxembourg	4,6	4,8	0,00	0,00	0,00	0,00
Netherlands	6,8	9,0	0,00	0,00	0,00	0,00
Portugal	1,2	1,3	0,00	0,00	0,00	0,00
Spain	2,8	4,1	0,00	0,00	0,00	0,00
Sweden	3,3	6,6	0,00	0,00	0,00	0,00
United Kingdom	3,1	3,7	0,00	0,00	0,00	0,00
2010/TD						
Austria	11,0	20,9	0,00	0,00	0,00	0,00
Belgium	6,0	9,1	0,00	0,00	0,00	0,00
Denmark	3,3	4,9	0,00	0,00	0,00	0,00
Finland	2,0	8,8	0,00	0,00	0,00	0,00
France	4,5	11,5	0,00	0,00	0,00	0,00
Germany	6,9	9,6	0,00	0,00	0,00	0,00
Greece	5,7	11,1	0,00	0,00	0,00	0,00
Ireland	2,1	4,7	0,00	0,00	0,00	0,00
Italy	5,7	12,2	0,00	0,00	0,00	0,00
Luxembourg	4,5	5,0	0,00	0,00	0,00	0,00
Netherlands	5,7	7,3	0,00	0,00	0,00	0,00
Portugal	0,9	1,3	0,00	0,00	0,00	0,00
Spain	2,0	3,9	0,00	0,00	0,00	0,00
Sweden	3,6	7,8	0,00	0,00	0,00	0,00
United Kingdom	2,4	3,2	0,00	0,00	0,00	0,00

1990 **PM10 anthropogenic emission, target value>annual average >20 ug/m3**

PM10	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	27,4	42,9	0,80	100,00	1,89	100,00
Belgium	18,8	24,7	0,19	34,40	0,60	100,00
Denmark	9,8	12,8	0,00	0,00	0,00	0,00
Finland	6,6	21,8	0,00	0,00	0,40	71,94
France	17,0	29,9	0,06	11,30	0,97	100,00
Germany	26,3	34,0	0,72	100,00	1,26	100,00
Greece	20,9	32,5	0,34	61,14	1,15	100,00
Ireland	7,1	11,1	0,00	0,00	0,00	0,00
Italy	18,9	37,4	0,19	34,89	1,50	100,00
Luxembourg	18,4	20,2	0,16	29,75	0,28	51,52
Netherlands	22,1	27,5	0,42	76,42	0,80	100,00
Portugal	3,1	4,0	0,00	0,00	0,00	0,00
Spain	6,9	12,1	0,00	0,00	0,00	0,00
Sweden	6,4	13,3	0,00	0,00	0,00	0,00
United Kingdom	13,0	15,6	0,00	0,00	0,00	0,00
2010/BL						
Austria	13,5	25,1	0,00	0,00	0,63	100,00
Belgium	11,1	15,2	0,00	0,00	0,00	0,00
Denmark	5,8	7,8	0,00	0,00	0,00	0,00
Finland	3,2	11,4	0,00	0,00	0,00	0,00
France	7,7	16,1	0,00	0,00	0,00	0,00
Germany	10,1	13,6	0,00	0,00	0,00	0,00
Greece	16,0	23,2	0,00	0,00	0,50	89,65
Ireland	4,6	7,5	0,00	0,00	0,00	0,00
Italy	8,8	17,3	0,00	0,00	0,08	15,12
Luxembourg	6,8	7,5	0,00	0,00	0,00	0,00
Netherlands	10,1	13,4	0,00	0,00	0,00	0,00
Portugal	2,1	2,6	0,00	0,00	0,00	0,00
Spain	3,8	6,3	0,00	0,00	0,00	0,00
Sweden	4,7	9,6	0,00	0,00	0,00	0,00
United Kingdom	4,7	5,8	0,00	0,00	0,00	0,00
2010/AP-NT						
Austria	11,3	17,5	0,00	0,00	0,09	17,42
Belgium	4,7	6,8	0,00	0,00	0,00	0,00
Denmark	4,6	7,8	0,00	0,00	0,00	0,00
Finland	3,2	11,1	0,00	0,00	0,00	0,00
France	4,8	11,0	0,00	0,00	0,00	0,00
Germany	6,7	9,2	0,00	0,00	0,00	0,00
Greece	15,6	20,9	0,00	0,00	0,33	60,46
Ireland	2,3	4,9	0,00	0,00	0,00	0,00
Italy	4,9	9,5	0,00	0,00	0,00	0,00
Luxembourg	4,8	5,1	0,00	0,00	0,00	0,00
Netherlands	9,1	12,4	0,00	0,00	0,00	0,00
Portugal	1,2	1,4	0,00	0,00	0,00	0,00
Spain	2,9	4,5	0,00	0,00	0,00	0,00
Sweden	3,5	9,5	0,00	0,00	0,00	0,00
United Kingdom	3,3	4,0	0,00	0,00	0,00	0,00

2010/AP-FT	PM10 anthropogenic emission, target value>annual average >20 ug/m3					
PM10	Concentration yearly average		Regional		Agglomerations	
	Regional	Agglomerations	max ex-ceedance	cum exposure	max ex-ceedance	cum exposure
Austria	10,8	14,7	0,00	0,00	0,00	0,00
Belgium	4,3	6,1	0,00	0,00	0,00	0,00
Denmark	4,2	4,9	0,00	0,00	0,00	0,00
Finland	2,8	6,8	0,00	0,00	0,00	0,00
France	4,7	10,1	0,00	0,00	0,00	0,00
Germany	5,8	7,5	0,00	0,00	0,00	0,00
Greece	15,2	18,7	0,00	0,00	0,18	32,56
Ireland	2,0	3,4	0,00	0,00	0,00	0,00
Italy	4,7	8,7	0,00	0,00	0,00	0,00
Luxembourg	4,6	4,8	0,00	0,00	0,00	0,00
Netherlands	6,8	9,0	0,00	0,00	0,00	0,00
Portugal	1,2	1,3	0,00	0,00	0,00	0,00
Spain	2,8	4,1	0,00	0,00	0,00	0,00
Sweden	3,3	6,6	0,00	0,00	0,00	0,00
United Kingdom	3,1	3,7	0,00	0,00	0,00	0,00
2010/TD						
Austria	11,0	20,9	0,00	0,00	0,33	60,40
Belgium	6,0	9,1	0,00	0,00	0,00	0,00
Denmark	3,3	4,9	0,00	0,00	0,00	0,00
Finland	2,0	8,8	0,00	0,00	0,00	0,00
France	4,5	11,5	0,00	0,00	0,00	0,00
Germany	6,9	9,6	0,00	0,00	0,00	0,00
Greece	5,7	11,1	0,00	0,00	0,00	0,00
Ireland	2,1	4,7	0,00	0,00	0,00	0,00
Italy	5,7	12,2	0,00	0,00	0,00	0,00
Luxembourg	4,5	5,0	0,00	0,00	0,00	0,00
Netherlands	5,7	7,3	0,00	0,00	0,00	0,00
Portugal	0,9	1,3	0,00	0,00	0,00	0,00
Spain	2,0	3,9	0,00	0,00	0,00	0,00
Sweden	3,6	7,8	0,00	0,00	0,00	0,00
United Kingdom	2,4	3,2	0,00	0,00	0,00	0,00

Appendix H: Ranking of 15 EU countries relatieve to emission and concentration

Relative position of the EU countries, averaged over: SO2, NO2, PM10 and Benzene

ranking average over all scenario's and 1990

emis	country	conc
1	Luxembourg	3
2	Germany	10
3	Netherlands	5
5	Portugal	1
6	Austria	13
6	France	12
8	Belgium	8
8	Italy	15
8	Sweden	6
10	Finland	11
10	United Kingdom	7
11	Spain	10
13	Ireland	3
13	Denmark	3
15	Greece	14

ranking changes BL-TD compared to 1990

emis	country	conc
1	Luxembourg	4
2	Germany	11
3	Austria	13
4	Netherlands	6
5	Sweden	5
6	Portugal	1
7	France	12
8	Belgium	7
9	Italy	15
10	Finland	10
11	Ireland	2
12	Spain	9
13	Denmark	3
14	United Kingdom	8
15	Greece	14

negative
neutral
positive

The left column in the left table indicates the average position of the emission per inhabitant over all scenario's , the right column for the concentration. The colour indicates the relative change of position when concentrations are compared with emissions. E.g. Denmark and Ireland have relative an unfavourable position when compared to emission (position 13,3 and 12,7 respectively). Their position in the concentration ranking is relative favourable, 3,0 and 3,3 respectively. Main reasons: both countries are located in a climatic zone with relative high windspeed (= quick dilution of the pollutants) and are geographical located at places with a low background from other places (especially true for Ireland).
The right table compares the 2010 scenario's for baseline and best available technology, presented is the average ranking and the relative change in position, green indicates an improve in position from the base to the bat scenario. For the Netherlands as well the relative emission as concentration position improves. For Denmark the ranking of the emission situation decreases from baseline to bat while the ranking for the concentration doesn't change.

Appendix I: Burden of disease associated with environmental exposures; results of provisional calculations

In this appendix results are presented of a tentative effort to assess the disease burden in EU 15 associated with a set of environmental exposures for 1990 and 2010.

To describe and compare the disease burden associated with environmental exposures, and, eventually, to perform cost effectiveness analysis of options for environmental policy obviously some sort of 'public health currency' is required. Considering the fact that annual mortality or even loss of life expectancy do not fully represent the environmental health loss we tentatively applied an approach largely based on the 'burden of disease' measure that was developed by Murray and Lopez on behalf of the World Bank and WHO. To assess the global disease burden, and consequently the health policy priorities in different regions in the world, they employed 'disability adjusted life years' (DALYs). This health impact measure combines years of life lost and years lived with disease or disability that are standardised by means of severity weights[i,ii]. Our adaptation of the DALY-concept was inspired by the notion that the multiform health loss due to environmental exposure is fairly well characterised by three dominant aspects of public health, viz. *quantity* of life (life expectancy), *quality* of life, and *social magnitude* (or number of people affected).

Figure 1 sketches the basic idea behind our approach. At birth potentially each of us may expect around eighty years of healthy life. However, due to our genetic program, our often-unfavourable life-styles, poverty, occupational or environmental conditions or just bad luck, most of us will encounter disease that will reduce the quality of part of our life-years. These diseases may manifest themselves in episodes, chronic or even progressive until death. Some of us will die abruptly, for instance caused by an accident or an infectious disease. Thus, public health loss is defined as time spent with reduced quality of life, aggregated over the population involved. The methodology to estimate burden of disease burden associated with environmental exposures is described elsewhere[iii].

In the framework of the present study adequate data and future projections were only available for particulate matter, ozone and UV. For each EU-country we calculated attributable risks with respect to relevant health outcomes by combining population weighted exposure distributions for air pollutants (see paragraph..) with relative risk estimates derived from the epidemiological literature. Country-specific health-outcome incidence data were derived based on the WHO Health for All database[iv]. Subsequently for each country the number of cases was estimated by combing baseline incidence rates with the country-specific attributive risks (for future disease burden calculations we assumed no major changes in incidence rates). Table 1 presents the set of endpoints we used to arrive at estimates of attributable disease burden and the number of DALYs lost per 1000 cases. Finally a total exposure attributable disease burden was calculated by aggregating the number of DALYs for each health outcome. The disease burden associated with additional UV-exposure due to ozone layer degradation was calculated by aggregating country-specific skin cancer annual morbidity and mortality estimates of Slaper et al., (chapter ..) and Dutch burden of disease data[v].

Figures 2, 3, 4 and 5 show for 1990 and for the Baseline and Accelerated Policy scenarios the disease burden per 1 million inhabitants for each EU-country that can be attributed exposure to particulate air pollution, photochemical air pollution (ozone) and UV due to ozone layer degradation, respectively. The disease burden associated with these exposures appears to be dominated by the long-term effects of particulates. However, it has to be noted that these estimates are based on the results of a limited number of North-American cohort studies; the internal and external validity of which still needs to be verified, preferably in well-designed European epidemiological studies. The results are discussed briefly in the box 'Public health loss due to environmental exposures' at the end of section 3.5.2 of the main report.

Table 1. Disability-Adjusted Life Years (DALYs) per 1000 cases lost to air pollution and UV-radiation.

Environmental factor	**Health outcome**	DALY per 1000 cases
particulate air pollution	*mortality*	
long-term	- cardiopulmonary	5800
	- lung cancer	9000
	morbidity	
	- chronic respiratory symptoms (< 15)	40
	- chronic bronchitis (> 15)	2638
particulate air pollution	*mortality*	
short-term	- respiratory	175
	- coronary heart dis	175
	- pneumonia	175
	- other	175
	hospital admission	
	- respiratory	22
	- cardiovascular	25
	emergency room visits	
	- respiratory	16
	aggravation of asthma	
	- asthmatic attacks	0,6
	- use of bronchodilators	0,6
	aggravation of resp. symptoms	
	- upper respiratory. tract	0,8
	- lower respiratory. tract	6
	affected lung function	
	- decreased FEV1 >10%	0
ozone	*mortality*	
	- respiratory	175
	- cardiovascular	175
	- pneumonia	175
	- other	175
	hospital admission	
	- respiratory disease	29
	emergency room visits	
	- respiratory disease	16
UV-A/UV-B	*mortality*	
O_3-layer degradation	melanoma	23000
	other	20200
	morbidity	
	melanoma	690
	basal cell	11
	squamous cell	41

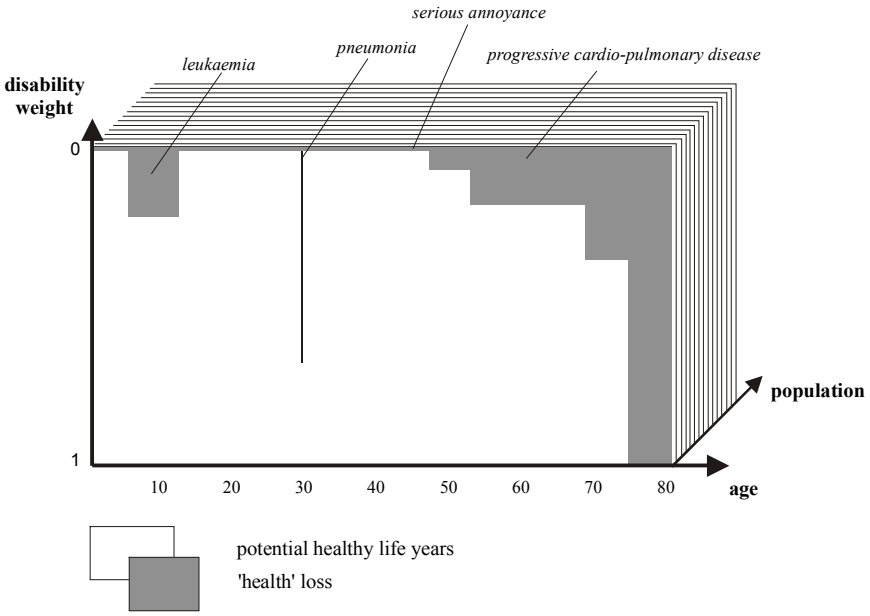

Figure 1. Diagram of the concept of disability adjusted life years.

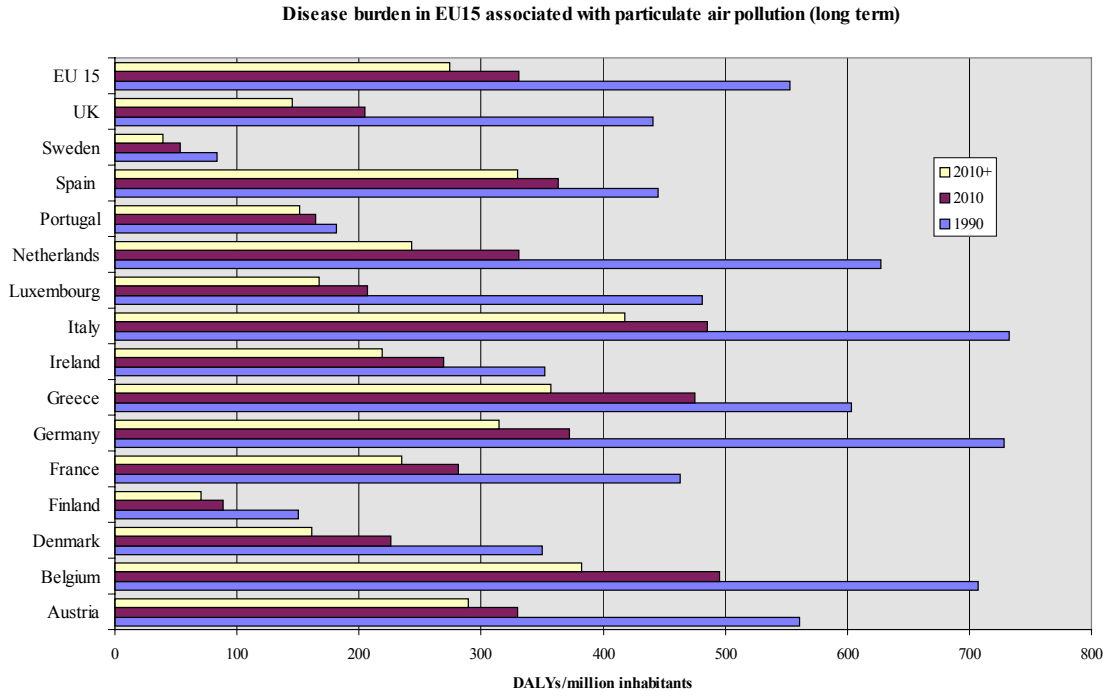

Figure 2. Disability-Adjusted Life Years (DALYs) lost annually to long term-exposure to particulate air pollution.

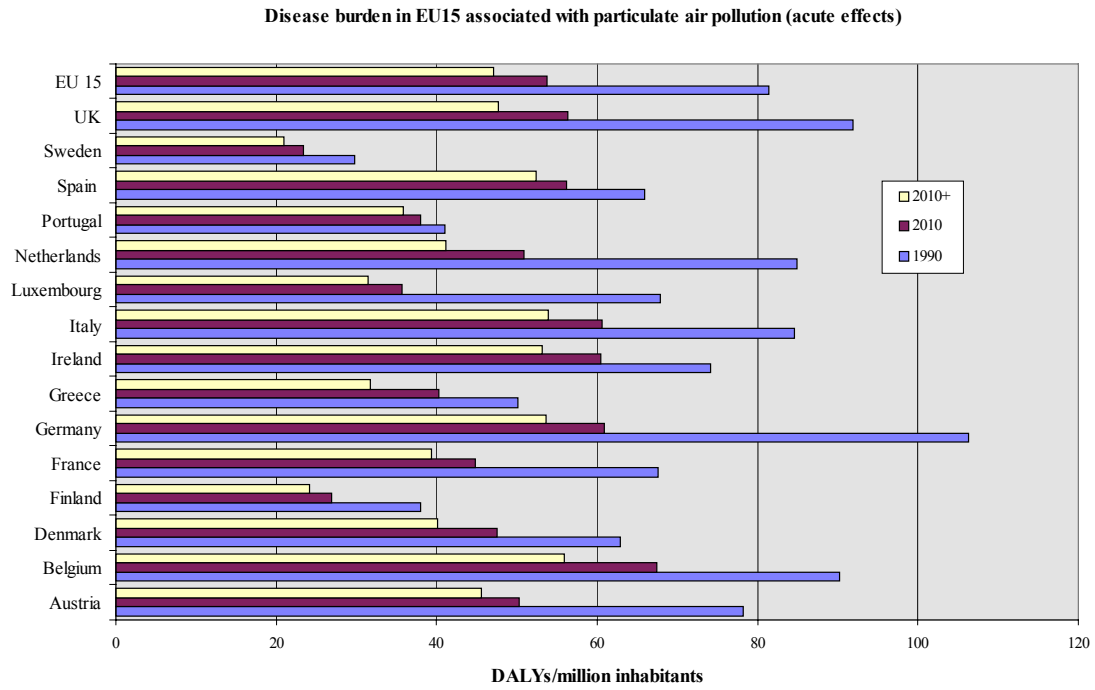

Figure 3. Disability-Adjusted Life Years (DALYs) lost annually to short-term exposure to particulate air pollution.

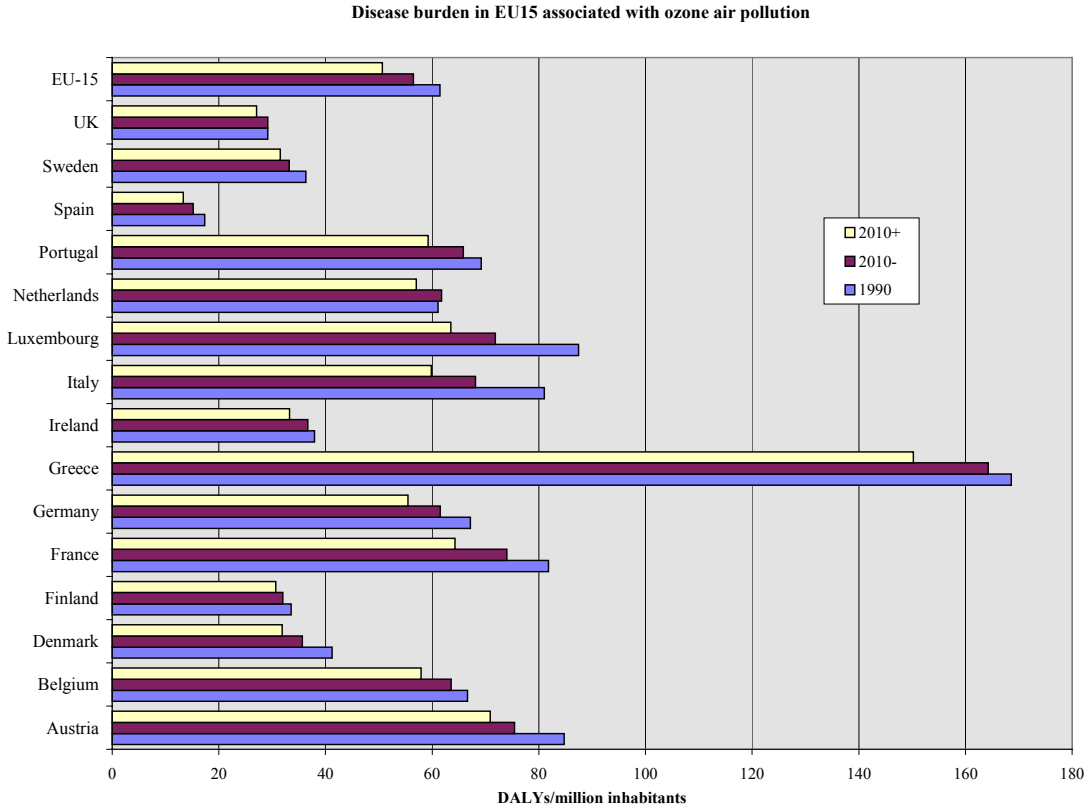

Figure 4. Disability-Adjusted Life Years (DALYs) lost annually to exposure to ozone.

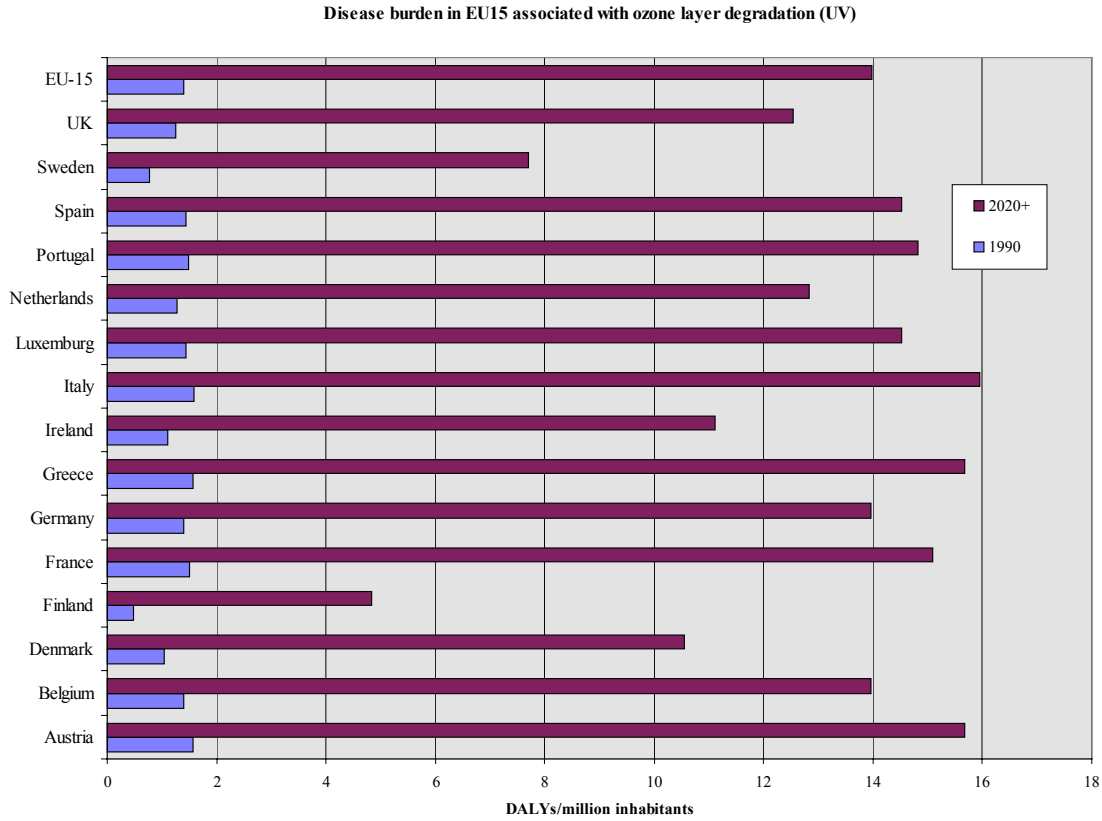

Figure 5. Annual disease burden (DALYs) due to ozone layer degradation related UV-exposure (skin cancer).

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