Reducing the concentration of fine particulates in ambient air

Three factors are discussed that should be considered for a robust air quality policy.

From many health and epidemiological studies, it is generally accepted that exposure to fine particulate matter (PM) is harmful to human health and that actions should be taken to reduce the concentration of PM in air.

Atmospheric PM is a complicated mixture of particles from different origins, arising both from natural sources and from human activity (anthropogenic PM). The PM mixture changes over time as sources change and as mitigation measures are implemented in response to new regulations. Epidemiological studies that provided the statistical evidence related to health effects did not and, in fact, could not account for changes in PM composition. Thus, the PM concentration in air is the controlling parameter and reducing the overall PM concentration is the air quality policy target. Putting controls on PM sources, both for directly emitted or primary PM and for materials that react in the atmosphere to form secondary PM, will change the composition of PM. If different components of PM have different degrees of harmful effect, then it is important to assess how different reduction strategies that simply reduce the PM mass concentration will perform.

Health impacts of PM

The Clean Air for Europe (CAFE) programme (2001–2005) was the first European policy study to conclude that reducing concentrations of PM would improve human health. The resulting 2005 Thematic Strategy on Air Pollution (TSAP) set as one of its objectives the aim of reducing the calculated 'Years of Life Lost' (YOLL) in the European population due to exposure to $PM_{2.5}$[1] by 47% in 2020 compared to 2000.

The CAFE programme assumed that a life-long exposure to an annual $PM_{2.5}$ concentration of 10 $\mu g/m^3$ would increase the mortality risk by 6%. This mortality risk is based on a 2007 UK study[2] but is similar to values reported by the US Science Advisory Board, the World Health Organization and others. To put this figure in context, starting with the risk profile of a 2005 UK population, a 10 $\mu g/m^3$ reduction in PM would be expected to increase life expectancy by 7.5 months.

The monetary value that is placed on the estimated reduction in mortality risk is called the 'Value of a Life Year' (VOLY) and is used in EU air quality policy to compare health benefits with the costs of mitigating airborne PM using different mitigation measures. Because the VOLY is a large number and is multiplied by the YOLL for the whole EU population, the perceived value of mitigating $PM_{2.5}$ is very high when examining different options for improving ambient air quality.

Of course, many factors other than PM levels affect mortality risk, such as access to health care. In fact, dramatic reductions in mortality risk have occurred over the past 40+ years from improved socio-economic conditions and other measures implemented to improve human

1 $PM_{2.5}$ is particulate matter with diameter of 2.5 micrometres or less.

2 UK Committee on Medical Effects of Air Pollution (COMEAP, 2007).

What are 'Years of Life Lost' and how are they estimated?

An actuarial life-table is typically used to describe the evolution of 100,000 people, called a 'cohort', from birth to death. For example, if the life expectancy is 70 years at birth, then this cohort of 100,000 people contains 70 x 100,000 or 7,000,000 life-years. If the life expectancy at birth is instead 70 years plus one month, then a cohort of 100,000 people would contain 7,008,333 life-years.

There are, of course, many health and environmental factors that can be expected to increase life expectancy, not just a reduction in PM emissions. However, if we assume that a reduction in PM exposure reduces the mortality risk and increases the life expectancy at birth by just one month, then we can attribute these additional life-years solely to the benefits of PM reduction.

Therefore, in theory, a hypothetical cohort of 100,000 people could 'lose' 8,333 years of additional life if they were born into a world where mitigation measures had not been put in place to reduce airborne PM levels. This hypothetical 'loss' is called the 'Years of Life Lost' or YOLL.

Of course, the robustness of these estimates cannot be tested because it is not possible (or ethical) to expose two actual populations of people to different ambient PM concentrations without changing any other factors that affect life expectancy. For this reason, YOLL, when used in an air quality context, is a hypothetical estimate based on assumptions regarding human exposure to air pollutants, such as PM.

health. Since 1960, life expectancy has increased at a rate of approximately 2.5 months per year, a total increase over this period of more than 10 years.

PM reductions that can be realistically achieved by the most cost-effective measures are found to have a value of about 2–3 months over a 10-year time frame, that is, a life expectancy improvement of only about 10% compared to the normal variation that has been observed since 1960. For this reason, it is very difficult to quantify the actual improvement to a population's life expectancy from air quality improvements alone.

Robustness of policy assumptions

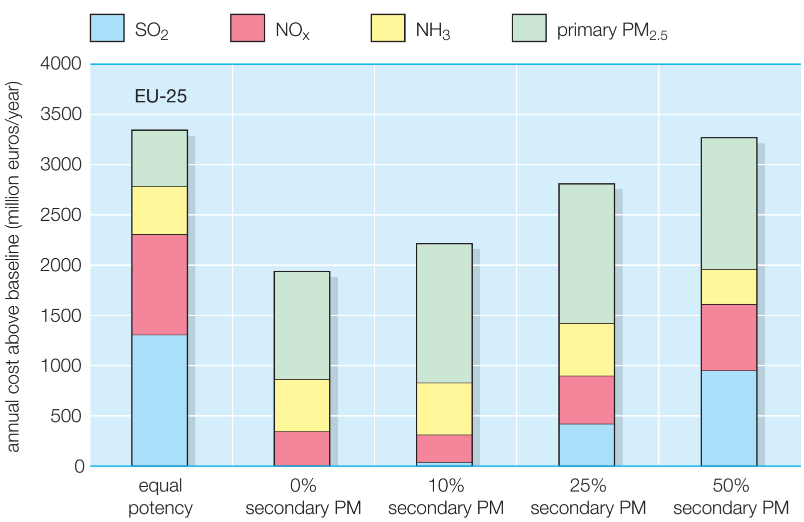
The effect of particle composition

In developing the TSAP, it was assumed that all particles are equally harmful to human health so that all measures to reduce PM concentrations are equally effective. We can examine what the effect might be if particles from some sources are more 'potent' and others less 'potent' in their effect on human health.

As an example, Figure 1 shows what would happen to a control strategy if the inorganic secondary particles, ammonium sulphate and nitrate formed in the atmosphere, were less potent than the primary particles emitted from combustion processes. Controls on SO_2, NO_x and NH_3 are needed to reduce secondary particles, while controls on smoke emissions are needed to reduce primary particles. In this example, the cost of meeting all of the TSAP environmental objectives are calculated, not just the cost of reducing the year 2000 YOLL by 47% in 2020. The bars show the additional cost for the EU-25 (in millions of euros) above the baseline cost of currently agreed legislation, based on assumptions about the relative potency of primary and secondary PM.

Figure 1: The bars show the additional cost above baseline for the EU-25 based on assumptions about the relative potency of primary and secondary PM.

Figure 1 Additional cost above baseline to meet the same TSAP objectives for different assumptions about the relative potency of primary and secondary PM



The left-most bar shows the current TSAP approach where all PM, both primary and secondary, are assumed to be equally potent in their effect on human health. The costs of mitigation measures for controlling SO_2, NO_x, NH_3 and primary $PM_{2.5}$ are shown in terms of their annual cost to implement.

The second bar represents an extreme case where all harmful particle effects are assigned to primary $PM_{2.5}$. The overall cost of mitigation measures is markedly lower. All costs for controlling SO_2 are essentially avoided, NO_x control costs are lower and NH_3 costs are similar. The total cost for controlling primary $PM_{2.5}$ has more than doubled however.

This calculation meets the EU targets for reducing acidification and eutrophication which require reductions in NO_x and NH_3. The contribution of SO_2 to acidification in Europe is now very small as a result of the historical reductions in these emissions.

The remaining three bars in Figure 1 show the effect of re-introducing harmful effects from secondary particles at different levels, from 10% to 25% to 50%. The risk factor for the whole PM mixture is kept constant so there is a compensating reduction in the potency of primary particles compared to the second case. Controls on SO_2 emerge once again as an important mitigating factor as the assumed potency of secondary particles increases. The overall cost of measures also increases.

The effect of sectoral contributions

For emissions reporting purposes, activities are commonly grouped into sectors according to the SNAP (Selected Nomenclature for sources of Air Pollution) convention. Sector 1 is large combustion sources, Sector 2 is non-industrial combustion including domestic sources, Sector 3 is industrial combustion (including oil refineries), Sector 4 is process industries, Sector 7 is on-road vehicles, and Sector 8 is non-road machinery and transport. Sectors 5 and 6 are not shown.

We can consider how best to describe emissions from different sectors when assessing air quality policy. The Integrated Assessment Modelling (IAM) methodology has been used in Europe to evaluate the costs associated with reducing emissions from different sectors. The IAM assesses the effect on emissions concentrations by modelling these reductions in national emissions. This methodology is a good approach for those sectors that are more or less evenly distributed across the country. A fully sectoral modelling approach, however, should represent the geographic distribution of emissions from different sectors and not just their total emissions.

In a previous study[3], a sectoral approach was tested to see if it would give similar or better results when compared to an approach based only on national emissions limits. The relationship between changes in emissions from different sectors on pollutant concentrations was calculated, accounting for the geographic distribution of sector emissions. An important aspect, from a health

[3] *EURODELTA: Evaluation of a sectoral Approach to Integrated Assessment Modelling—Second report*

Figure 2 Results of a sectoral study on $PM_{2.5}$ control scenarios for some European countries

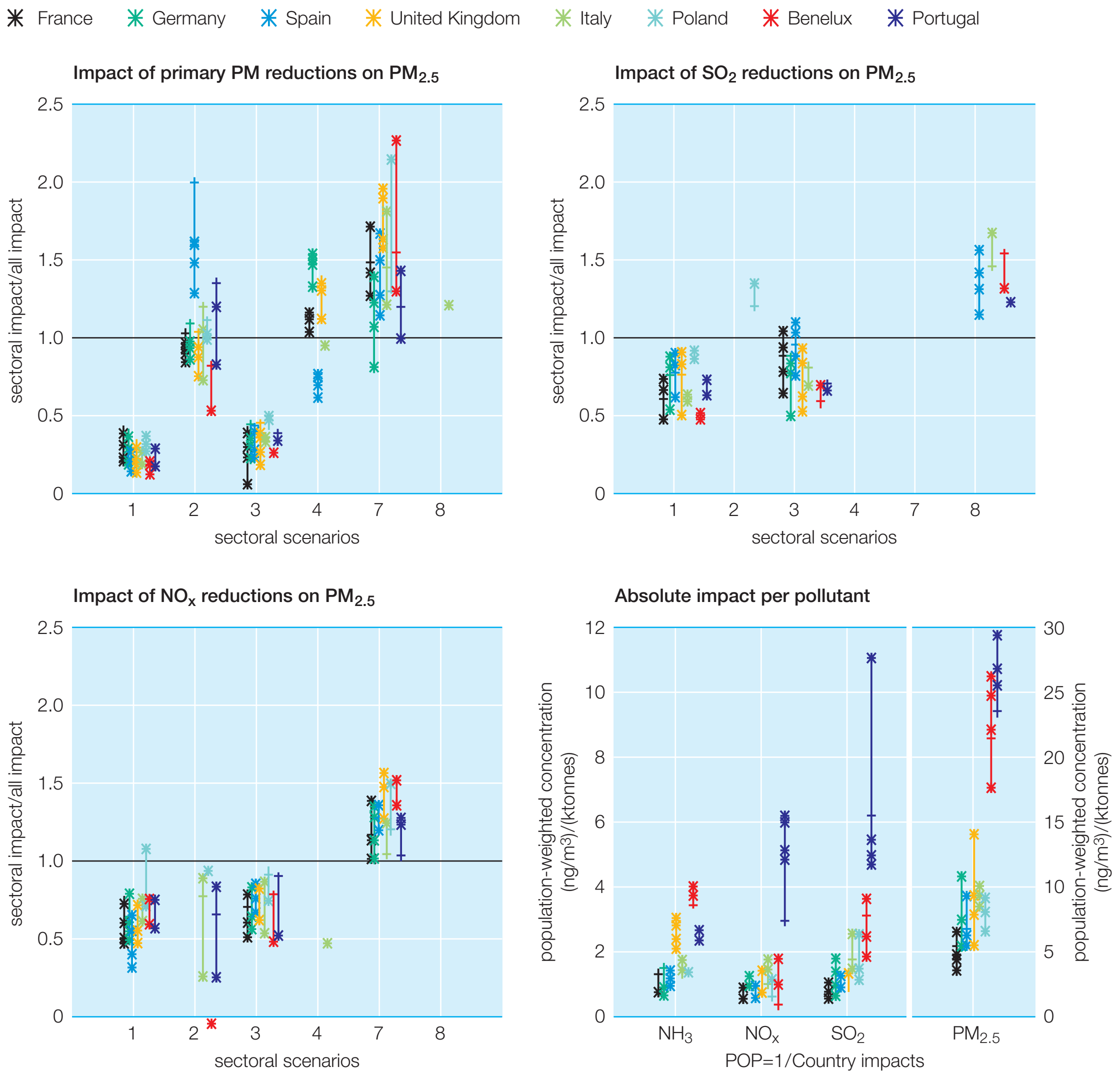


Figure 2: Results from the EURODELTA[3] study show that reducing industrial emissions (Sectors 1 and 3) is generally much less effective than would be expected using the national emissions limit approach. This is an important finding because a relatively high weighting is placed on controlling industrial emissions as part of the Integrated Assessment Methodology process.

assessment viewpoint, is that emissions from sectors that are close to population centres may have a greater effect on human health than those that are farther away.

Results from this study are shown in Figure 2 for different European countries. Looking first at the upper left-hand box, the vertical axis shows the effectiveness of a targeted reduction in $PM_{2.5}$ from a particular sector compared to a reduction in the national emissions limit from all sectors. Points above the 1.0 line mean that an emissions reduction from a particular sector is expected to produce a greater reduction in $PM_{2.5}$ exposure compared to a reduction in the national limit for $PM_{2.5}$ emissions. Consequently, greater reductions in airborne PM could be expected by targeting mitigation measures on specific sectoral emissions. The other boxes show the impact of reductions in SO_2 and NO_x on $PM_{2.5}$ as well as the absolute impact for reductions in each pollutant.

The results show that reducing primary $PM_{2.5}$ from industrial emissions (Sectors 1 and 3) is generally much less effective than would be expected by using the national emissions approach. This is an important finding because it means that the IAM over-emphasises the importance of industrial sources for PM. It also means that targets for $PM_{2.5}$ reductions may not be met if they rely on industrial control measures. It is clearly important that mitigation measures are applied to the most appropriate emissions sources if policy measures are to be successful in achieving the air quality and human health objectives.

The effect of air quality policy on climate cooling

Emissions of carbonaceous particles (black carbon) are known to have a climate warming effect, both as an aerosol and through the effect of PM deposits on snow surfaces. Emissions of SO_2, however, have a strong cooling effect through the formation of sulphate particles. It is now clearly understood that the significant reduction in sulphur emissions achieved over the past 30 years to help reduce acidification in Northern Europe has also removed a substantial climate cooling effect.

While continued reductions in SO_2 emissions based on the current IAM methodology are mainly driven by the effect of PM on human health, the corresponding beneficial effects of sulphate emissions on climate cooling have not been adequately evaluated. If the two effects mentioned in the previous sections are combined—differences in potency of PM components and the lack of effective controls on sectoral emissions—then current mitigation options required by the IAM could be ineffective, costly, and counterproductive from a climate warming and cooling perspective.

Conclusion

There is no question that improving air quality and reducing the impact of air pollutants, such as PM, on human health is an important objective. Mitigation measures, however, should be evaluated fairly, based on their cost-effectiveness for achieving the desired air quality improvements. Otherwise, there is considerable potential that investments will be made to reduce emissions, but that these will not in fact achieve the desired improvements in air quality or human health.

As demonstrated here, there are two serious sensitivities that should be accounted for when designing air quality policy. Individually, they challenge the effectiveness of mitigation measures that can be expected based on the current modelling approach. If these two factors act together, then current policy measures are very likely to under-perform against their expected targets and there may well be consequential and adverse impacts for climate. These aspects should be explored in greater detail to evaluate the robustness of proposed policy measures. More importantly, the IAM used for modelling European air quality policy should be updated to take these effects into account.