On health risks of ambient PM in the Netherlands

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Abstract

Particulate Matter (PM) in the ambient air can lead to health effects and even to premature mortality. This result has been found in a score of epidemiological studies, but their cause is not yet clear. It is certain, however, that these effects are so serious and so extensive that further action is warranted. In the scientific literature ambient aerosols are known as PM, short for Particulate Matter. Depending on the diameter or size of the particles, they are termed PM_{10} (for particles with diameters of up to approximately 10 micrometres) or $PM_{2.5}$ (for those less than 2.5 micrometres in diameter). One micrometre is a thousandth of a millimetre. Humans inhale particles smaller than 10 micrometres, which end up deep in our airways.

Recent studies have presented well-founded assumptions concerning the biological mechanisms involved and the groups of people that are probably more susceptible to PM. Particulate Matter is a generic term for a complex mixture of large and small airborne particles. However, the causal factors within this complex mixture are difficult to disentangle and have not yet been identified. The second Section of this report looks at the different types of PM, their atmospheric behaviour and the methods of measuring them. The health effects associated with PM are also presented. Section 3 discusses the most recent epidemiological, toxicological and human clinical findings and their mutual relationships.

On the basis of epidemiological studies it has been estimated that in the Netherlands some 1,700 to 3,000 people per year die prematurely as a result of inhaling ambient PM. These figures reflect only the effects of acute exposure to air pollution. If the long-term effects of chronic exposure are taken into account, premature mortality could affect 10,000–15,000 people a year in the Netherlands. These last estimates for chronic exposure are more uncertain, because chronic effect studies are much fewer in number. The estimate of the chronic effects was based on foreign studies, which are not completely comparable with the Dutch situation.

Section 4 gives an overview of the most recent information relating to sources of PM and emissions in the Netherlands, while the last Section presents a critical evaluation of the current and future EU standards.
It is recommended that PM_{10} be retained as a standard for the time being, as it covers the effects of both fine and coarse particles. In view of the emerging evidence implicating fine particles in health effects, it is recommended that a standard for fine PM and/or a source-related fraction be developed as well.

Even with PM concentrations well below European Union (EU) standards, people's health will still be affected because no threshold has been found for the occurrence of health effects. PM is a complex mixture containing fractions that are to a greater or lesser extent health-relevant. This differentiation in potency has profound implications for an efficient and effective reduction of health impacts through PM emission abatement.

PM abatement can be justified by the precautionary principle. Further source- oriented actions could focus on reduction of the total PM_{10} aerosol mass or, first of all, on those PM fractions that are expected to be more health-relevant. This last option is preferred. These fractions are probably transport-related (diesel soot) and, more generally, combustion-related primary PM emissions. Abatement should therefore focus on these sources. In this respect, the abatement of uncontrolled shipping emissions has been identified as one of the more cost-effective control options.

Abatement of other combustion sources such as industrial combustion, wood burning in fireplaces, and off-road machinery are also possible, but less cost-effective.

The European Union has decided on two standards for PM, a daily and an annual average value. The current EU standards for daily and annual average values are not equivalent, as was originally intended. In the Netherlands the following options are equivalent to the EU annual standard of 40 $\mu g/m^3$: a daily level of 50 $\mu g/m^3$ with 80 exceedances (while the EU allows 35 exceedances) or a daily level of 100 $\mu g/m^3$ with 7 permitted exceedances per year. For practical reasons a daily standard of 100 $\mu g/m^3$ is preferred. Although the EU has proposed two standards for PM, there are several arguments that only one standard would suffice – annual mean concentrations being the best choice. However, for reasons of communication to the public daily standards may be appropriate.

Compliance with the annual average EU standard seems feasible for PM_{10} in the Netherlands in 2005, although local exceedances at 'hot spots' cannot be ruled out. Compliance in 2010 with the indicative annual average EU standard of 20 $\mu g/m^3$ is not feasible, even at high cost. Expectations are that there will still be 36–40 exceedances per year of the EU daily standard of 50 $\mu g/m^3$ even after all planned abatement measures (Current Legislation of Emissions (CLE)) have been taken in 2010. Therefore, compliance with the current EU daily standards for 2005 and 2010 does not seem feasible in the Netherlands and adverse health effects will continue to occur.

Samenvatting

Fijn stof in de lucht kan leiden tot gezondheidsklachten en zelfs vroegtijdige sterfte. Dat blijkt uit een honderdtal epidemiologische studies. Hoe die effecten precies ontstaan is nog niet duidelijk. Vast staat echter dat de gezondheidseffecten door fijn stof zo ernstig en omvangrijk zijn dat nadere actie geboden is. In de wetenschappelijke literatuur staat fijn stof bekend als 'deeltjesvormige luchtverontreiniging' (Engels: Particulate Matter, ofwel PM.). Afhankelijk van de doorsnee van de stofdeeltjes wordt gesproken van PM_{10} (voor deeltjes met een doorsnee tot 10 micrometer) of $PM_{2,5}$ (doorsnee tot 2,5 micrometer). Een micrometer is een duizendste millimeter. Deeltjes kleiner dan 10 micrometer worden door mensen ingeademd en dringen door in de luchtwegen.

Dankzij recente studies zijn er gegronde vermoedens over de biologische mechanismen die in het spel zijn en welke groepen mensen waarschijnlijk gevoelig zijn voor blootstelling aan fijn stof. Maar aangezien 'fijn stof ' een verzamelnaam is voor een complex mengsel van allerhande grote en kleinere stofdeeltjes in de luchtverontreiniging blijft het lastig om oorzakelijke verbanden te ontrafelen. In hoofdstuk 2 van dit rapport komen de verschillende fijn stof deeltjes, hun onderlinge wisselwerking in de atmosfeer en de diverse meetmethoden aan bod. Ook wordt een overzicht gegeven van de gezondheidsklachten die fijn stof kan veroorzaken. In hoofdstuk 3 worden de nieuwste epidemiologische, toxicologische en medische inzichten in onderlinge samenhang besproken.

Op grond van epidemiologische studies wordt geschat dat in Nederland jaarlijks zo'n 1700 tot 3.000 mensen vroegtijdig overlijden door het inademen van fijn stof. En dan hebben we het alleen nog over de acute gevolgen van blootstelling aan luchtverontreiniging. Nemen we ook de lange-termijneffecten van chronische blootstelling aan fijn stof in beschouwing, dan zouden in Nederland mogelijk zelfs 10.000 tot 15.000 mensen jaarlijks vroegtijdig overlijden. De laatste schattingen zijn met meer onzekerheid omgeven, aangezien chronische effecten in minder studies gekwantificeerd zijn dan acute effecten. Bovendien is de berekening het resultaat van een vertaalslag van internationale onderzoeksresultaten naar de Nederlandse situatie en die is niet helemaal vergelijkbaar.

Hoofdstuk 4 van dit rapport geeft een overzicht van de meest recente informatie over bronnen en emissies van fijn stof in Nederland. Aansluitend wordt in hoofdstuk 5 de huidige en toekomstige Europese normstelling kritisch beoordeeld.
Aanbevolen wordt om voorlopig PM_{10} te blijven hanteren als Europese standaard voor luchtverontreiniging door grove èn fijnere stofdeeltjes. Daarnaast zou er voor het fijnste stof een aparte normstelling of een meer brongerichte normstelling ontwikkeld moeten worden omdat er steeds meer aanwijzingen komen dat kleinere stofdeeltjes de gezondheid bedreigen.

Overigens is nooit aangetoond dat de gezondheidseffecten pas boven een bepaalde drempelwaarde optreden. Zelfs van fijn stof concentraties ver onder de huidige Europese normen zijn gezondheidseffecten in de bevolking te verwachten. Fijn stof is een complex mengsel van allerlei fracties die meer of minder van belang zijn voor de gezondheid. Die verschillen in toxische potentie wegen zwaar mee bij een doeltreffend emissiebeleid.

Bestrijding van de uitstoot van fijn stof valt te rechtvaardigen vanuit het voorzorgbeginsel.

Door verdere brongerichte maatregelen kan men de totale massa PM_{10} aërosol in de luchtverontreiniging terugdringen, of eerst die fracties aanpakken die vermoedelijk het meest relevant zijn voor de gezondheid. Waarschijnlijk behoren tot de relevante fracties het dieselroet uit de vervoerssector en fijn stof afkomstig van overige verbrandingsprocessen. Dergelijke bronnen verdienen prioriteit in het beleid voor uitstootbeperking van fijn stof. Bestrijding van de ongecontroleerde scheepvaartemissies blijkt bijzonder kosten-effectief. De aanpak van andere verbrandingsprocessen, zoals industriële verbranding, open haarden en mobiele werktuigen is ook mogelijk, maar minder kosten-effectief.

De EU heeft voor fijn stof twee normen vastgesteld, namelijk een dag- en een jaargemiddelde. Deze beide normen zijn niet gelijkwaardig, hoewel dat oorspronkelijk wel de bedoeling was. De Europese jaargemiddelde PM_{10} norm bedraagt 40 microgram fijn stof per kubieke meter lucht ($\mu g/m^3$). In Nederland kunnen we dat vertalen naar een dagelijkse norm van 50 $\mu g/m^3$ met 80 toegestane overschrijdingen per jaar (terwijl de EU-norm maar 35 overschrijdingen toestaat) of een dagelijkse norm van 100 $\mu g/m^3$ met 7 toegestane overschrijdingen per jaar. Om praktische redenen verdient die laatste norm de voorkeur. Overigens zijn er goede argumenten om maar één norm, en dan liefst een jaargemiddelde, te hanteren. Een daggemiddelde norm kan echter van pas komen bij publieksvoorlichting.

In 2005 lijkt de jaargemiddelde EU norm van 40 $\mu g/m^3$ voor fijn stof in Nederland in het algemeen haalbaar. Lokale overschrijdingen op ‘hot spots’ zijn echter niet uit te sluiten. In 2010 is de indicatieve jaargemiddelde waarde van 20 $\mu g/m^3$ in Nederland echter niet haalbaar, zelfs niet tegen hoge kosten. Zelfs als in 2010 alle voorgenomen stofbestrijdingsmaatregelen zijn uitgevoerd zullen vermoedelijk nog steeds 36 tot 40 maal per jaar daggemiddelde concentraties boven de 50 $\mu g/m^3$ voorkomen. De dagelijkse EU normen voor 2005 en voor 2010 lijken voor Nederland dan ook niet haalbaar en gezondheidseffecten zullen blijven bestaan.

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Summarised Conclusions

Assessment of PM health risks
Epidemiological studies present worldwide evidence for particulate matter (PM) associated serious health effects in the general population, which may lead to hospital admissions and premature mortality. Dutch observations are in line with the international scientific literature. In spite of the ongoing scientific debate and prevailing uncertainties concerning the quantification of acute and chronic health effects, the overall conclusion is that PM-associated health effects are so extensive and serious that further action is warranted.

Epidemiological studies could not identify a threshold for exposure levels related to PM health effects. This precludes regular standard setting, with a No Observed Adverse Effect Level (NOAEL) and safety factors. It implies that for any PM standard a certain level of impact on health will have to be accepted.

Because there is no threshold, adverse health effects are less effectively avoided by reducing episodic high concentrations than by reducing annual average concentrations (which will reduce the magnitude of occasional peak concentrations as well).

Overall, health effects are consistently associated with PM_{10} and $PM_{2.5}$. (These terms refer to ambient particles with diameters of up to approximately 10 and 2.5 micrometre). These associations are found in spite of the local differences in air quality, sources and the proportion of the susceptible sub-population. However, there seems to be heterogeneity between locations within the various epidemiological time-series studies for PM-associated health effects. This heterogeneity is manifested in differences in the size of the effects and may probably be influenced by local ambient and population-related circumstances. As for the future, a gradually ageing population and an increasing proportion of asthmatics or people with circulatory problems will proportionally enlarge the potentially susceptible sub-population.

Choice of PM indicators
Support is emerging for supplementing the current PM standard with other (smaller sized or source-related) indicators than PM_{10}. There is currently a lack of reliable information on ambient levels of these smaller sized or source-related fractions which is representative of the situation in the Netherlands. At the moment the available toxicological and epidemiological evidence is insufficient for regulating ultrafine (UF) particulate concentrations, though this is another field that needs more research as the potential health implications of UF may be considerable. It is recommended that PM_{10} be retained as a standard for the time being as it covers the effects of both fine and coarse particles. In view of the emerging evidence implicating fine particles in health effects, it is recommended that a standard for fine PM or a source-related fraction be developed as well.

Monitoring of PM
The levels of PM_{10} measured by a stationary site monitor seem to be representative of the personal exposure of the general public to ambient PM_{10}. Accurate measurement of PM is complicated. In the Netherlands a substantial fraction of the PM is ammonium nitrate, a salt that is in dynamic equilibrium with the gas phase of ammonium and nitrate. While PM is measured, the sample is heated to minimise interference from water. However, this volatises some of the PM, especially ammonium nitrate. The automatic PM_{10} monitoring network in the Netherlands therefore corrects for losses of semi-volatile material by using a factor of 1.3, as an

approximation. The accuracy of PM measurements needs to be increased, as semi-volatile ammonium nitrate is a principal component of PM in the Netherlands. More information on the specific chemical composition and size distribution of PM representative for typical situations in the Netherlands is needed to test relevant hypotheses concerning health effects, source contributions and possible atmospheric influences.

Risk reduction with current PM emission control
Current policies will reduce emissions of PM_{10} by about 20% from 1998 to 2010. The fraction of PM_{10} that is combustion-related and suspected of being health-relevant will show an even larger reduction of 40%, based on projections of energy use, transportation developments and performance of new technology in real-world conditions. Dutch emissions of $PM_{2.5}$ will decrease by about 30%. Traffic is an important source of carbonaceous PM (which can be broken down into Elemental Carbon (EC) and Organic Carbon (OC), the mixture of which comprises diesel soot) and ultrafines, which are emitted at breathing height, close to a large part of the population in the Netherlands.

Whether a reduction in PM levels leads to a proportional reduction in health effects is still uncertain. PM is a complex mixture with fractions that are to a greater or lesser extent health-relevant. Changes in the composition of this mixture might change the health impact. So, the most cost-effective policy will be to reduce that part of PM that causes the health problems. Unfortunately, there are currently only suggestions for the causal fractions as they have not yet been identified.

These health-relevant fractions are probably transport-related (diesel soot) and, more generally, combustion-related primary PM emissions. Certain fractions of ambient PM probably do not cause significant health effects. These include particle-bound water and probably sea salt particles. A number of epidemiological studies suggest that the crustal fraction is less health-relevant than combustion-related fractions. Toxicological studies with pure ammonium sulphate and nitrate (Secondary Inorganic Aerosol (SIA)) have not established overt toxicity of these components, even at concentrations considerably above ambient levels. In contrast, epidemiological studies continue to find strong associations between adverse health effects and secondary aerosol components such as sulphates and nitrates. This divergence of results has not yet been resolved.

The daily PM standard revisited
In 1999, the European Union promulgated PM standards for 2005 and 2010. The values for 2010 are indicative. This means that the values for 2010 become definitive after the evaluation in 2003. In this evaluation the experience of Member States in meeting the standards for 2005 will be taken into account, as well as the most recent scientific insights. The current EU PM standards can be found in the following table:

EU standards for PM_{10}.

	Phase1 1 January 2005	Phase2* 1 January 2010
Annual average	40 µg/m^3	20 µg/m^3
Daily average (24-hour)	50 µg/m^3	50 µg/m^3
Number of exceedances per year	35	7

*indicative value

The EU's original position paper envisaged promulgating two equivalent PM standards. However, the current EU annual average standard (40 µg/m^3) and daily standard (50 µg/m^3, 35 exceedances) for 2005 are not equivalent in the Netherlands.

With **80** permitted exceedances per year, a daily average of 50 µg/m^3 would be equivalent to an annual average of 40 µg/m^3, and a daily standard of 100 µg/m^3 PM_{10} with 7 exceedances. For practical reasons a standard with a value of 100 µg/m^{3}and 7 exceedances is preferred to a value of 50 µg/m^3 that may be exceeded on 80 days. In general, the public is able to comprehend a standard with a small number of exceedances better. Although the EU has proposed two standards for PM, there are several arguments that only one standard would suffice – annual mean concentrations being the best choice. However, for reasons of communication to the public, daily standards may be appropriate. Whether or not two averaging times are needed for an EU PM standard is a policy decision.

Dutch compliance with air quality standards with current control policy
Annually averaged values in the Netherlands obtained through modelling are consistent with measurements of PM_{10} here. Compliance with the annual average value of 40 µg/m^3 seems feasible for PM_{10} in the Netherlands, although local exceedances at 'hot spots' cannot be ruled out. However, compliance with the daily average value of 50 µg/m^3 with 35 permitted exceedances is probably not feasible in 2005. Because of the relatively large contribution of foreign PM in a small country like the Netherlands, combined with our substantial natural background levels caused by sea salt, crustal and other natural material, a daily level of 50 µg/m^3 will easily be exceeded. Expectations are that there will still be 36–40 exceedances per year of the EU daily standard of 50 µg/m^3 even after all planned abatement measures (Current Legislation of Emissions (CLE)) have been taken in 2010. Compliance with the indicative annual average value of 20 µg/m^3 for PM_{10} and with the indicative daily average value of 50 µg/m^3 with 7 permitted exceedances in 2010 is not possible in the Netherlands. If abatement measures are implemented in neighbouring countries (as one might expect they will), the number of exceedances will decrease.

The prospect of additional PM abatement
The ultimate potential for reducing primary PM_{10} emissions (on top of currently agreed measures, CLE) could be 60% in the Netherlands. This abatement package is called 'MFR_{ult}': ultimate Maximum Feasible Reduction. The cost of achieving the 'MFR_{ult}' is about 6000 million euro per year. The 'MFR_{ult}' reduction of 60% in primary PM_{10} emissions in the Netherlands will result in a 1.1 µg/m^3 lower PM_{10} concentration averaged over the country. An emission reduction up to a cost-efficiency of 55 euro/kg PM_{10} will lead to a reduction by a quarter (abatement package: '$2010_{quart\ red}$'). This can be achieved at a cost of 210 million euro per year and will result on average in a 0.3 µg/m^3 lower PM_{10} concentration. From the absolute value of the PM_{10} levels one can conclude that, averaged on a national level, these reductions seem fairly

small. Locally, however, higher reductions in PM_{10} levels of 1 to 5.5 µg/m^3 are modelled in the '$2010_{quart\ red}$' abatement package. The maximum reductions will be achieved in Rotterdam, which is densely populated. It is interesting to note that the measures directed at *transport* in the '$2010_{quart\ red}$' abatement package focus on the shipping sector only. When concentrating on probably more health-relevant fractions of PM, like traffic-related diesel soot, modelled reductions are relatively higher even. The presented abatement packages ('$2010_{quart\ red}$' and 'MFR_{ult}') correspond to a decrease of 20% and 50% respectively in average traffic-related diesel soot concentration levels of Dutch origin. These effects would increase even further if similar reduction technologies were to be applied to traffic in foreign countries also.

Supplementary PM abatement can be based on the precautionary principle. Further source-oriented actions could focus on the more cost-effective reduction of the total PM_{10} aerosol mass, or could first of all focus on those PM fractions that are expected to be more health-relevant. This last option is preferred. These fractions are probably transport-related (diesel soot) and, more generally, combustion-related primary PM emissions. In this respect, the abatement of uncontrolled shipping emissions has been identified as one of the more cost-effective control options. The abatement of other combustion-related sources such as industrial combustion, wood burning in fireplaces, and off-road machinery is also possible, but is less cost-effective. Additionally, climate change mitigation strategies may reduce combustion-related PM emissions.

Residual risk with improved PM air quality
A substantial part of the PM_{10} levels in the Netherlands cannot be influenced by policy measures, as natural sources are responsible for their ambient concentrations. Because future abatement measures will further reduce the anthropogenic fraction, the contribution of the natural fraction will increase proportionally. More insight into the chemical composition (specific tracers) and contribution of different sources to the currently ‘non-modelled’ and generally natural part of PM_{10} is necessary to find out how much of the current PM levels may eventually be influenced by abatement measures.

PM air quality will improve in the future. Despite the air quality, it could be conjectured that the health impact associated with PM will nevertheless become more pronounced. In the Netherlands the gradual ageing of the population and other demographic developments could lead to a more than proportionate rise in the susceptible sub-groups. However speculative the previous remark, continuing vigilance seems required for this only partially understood problem of PM.

Samenvattende conclusies

Beoordeling van gezondheidsrisico's van fijn stof
Epidemiologische studies uit de hele wereld wijzen op een verband tussen fijn stof (Eng.: *Particulate Matter* of *PM*) en ernstige gezondheidsklachten, die tot ziekenhuisopname en vroegtijdige sterfte kunnen leiden. Dat beeld wordt bevestigd door Nederlands onderzoek. Over ernst en omvang van de acute en chronische gezondheidseffecten woedt nog een wetenschappelijk debat. Er zijn nog veel onzekerheden. Vast staat echter dat de gezondheidseffecten door fijn stof zo ernstig en omvangrijk zijn dat nadere actie geboden is. In epidemiologische studies is geen drempelwaarde aangetoond waaronder géén gezondheidseffecten meer met fijn stof in verband gebracht kunnen worden. Er is dan ook geen klassieke grenswaarde of normstelling met een zogenoemde "No Observed Adverse Effect Level (NOAEL)" voor fijn stof. Welke normstelling men ook kiest, de bijbehorende gezondheidseffecten in de bevolking zullen nooit helemaal uit te sluiten zijn.

Omdat er voor gezondheidsklachten door fijn stof, geen drempelwaarde bestaat, zijn dergelijke gezondheidseffecten doeltreffender te verminderen door de jaargemiddelde concentraties fijn stof te verlagen dan door incidentele piekconcentraties te bestrijden. Bovendien zal het verlagen van de jaargemiddelde concentraties fijn stof ook tot vermindering van de incidentele piekbelastingen leiden.

Wereldwijd kunnen de gezondheidseffecten van fijn stof worden gekoppeld aan PM_{10} en $PM_{2.5.}$ (Dat zijn stofdeeltjes met een diameter tot ongeveer 10 respectievelijk 2,5 micrometer). Dit verband wordt steeds opnieuw gevonden ondanks lokale verschillen in luchtkwaliteit, wisselende bronnen en een wisselend aandeel van gevoelige groepen in de bevolking. Toch komen er in de diverse epidemiologische studies ook verschillen tussen lokaties aan het licht. De omvang van de gezondheidseffecten is vermoedelijk afhankelijk van lokale omgevingsfactoren of bevolkingsomstandigheden. In de toekomst zal een groter deel van de bevolking extra gevoelig zijn voor fijn stof. Dat komt door de toenemende vergrijzing en door het stijgende aantal astmatici en mensen met hart- en vaatstoornissen.

Keuze van indicatoren voor PM
Naast de huidige Europese PM_{10} normstelling voor fijn stof groeit de behoefte aan normstellingen, gericht op fijnere deeltjes of fracties afkomstig uit specifieke bronnen. Over de Nederlandse situatie ontbreekt echter voldoende betrouwbare informatie. Aanbevolen wordt om voorlopig PM_{10} als normstelling voor grove èn fijnere stofdeeltjes te handhaven. Daarnaast zou er voor fijnere stofdeeltjes een aparte normstelling of een meer brongerichte normstelling ontwikkeld moeten worden omdat er steeds meer aanwijzingen komen dat kleinere stofdeeltjes de gezondheid kunnen schaden.
De huidige toxicologische en epidemiologische informatie is onvoldoende om regelgeving op te stellen voor de ultrafijne (UF) deeltjes, die kleiner zijn dan 0,1 micrometer. Een flink deel van die ultrafijne deeltjes is afkomstig van het verkeer. Op dit gebied is meer onderzoek nodig, want de gezondheidsschade door ultrafijne deeltjes zou aanzienlijk kunnen zijn.

Meten van PM
De PM_{10} niveaus in de buitenlucht die op een vast meetpunt worden gemeten, blijken in de praktijk ook representatief te zijn voor de persoonlijke blootstelling van het algemene publiek aan PM_{10}. Het nauwkeurig meten van fijn stof is echter lastig. In Nederland bestaat een flink deel van het fijn stof uit ammoniumnitraat, een zout dat in

een dynamisch evenwicht verkeert met de gasfase van zowel nitraat als ammonium. Tijdens de metingen van het fijn stof in de stofmonitor worden de luchtmonsters verwarmd om storing door water te minimaliseren. Daarbij vervluchtigt echter een deel van het monster, vooral ammoniumnitraat.Het huidige automatische meetnet in Nederland hanteert daarom een factor van 1,3 om te corrigeren voor de verliezen van het semi-vluchtige deel van het fijn stof. Het is wenselijk dat de precisie van deze PM metingen vergroot wordt.
Ook is meer informatie nodig over de specifieke chemische samenstelling en deeltjesgrootteverdeling voor representatieve situaties in Nederland. Daarmee kunnen hypotheses over gezondheidseffecten, bronbijdragen en mogelijke atmosferische invloeden worden getoetst.

Risicoreductie en de bestrijding van emissies
Door het al vastgelegde fijn stof beleid gaan de emissies van PM_{10} tussen 1998 en 2010 in Nederland met zo'n 20 procent omlaag. De fijn stof fractie die verbranding gerelateerd is zal zelfs met 40 procent afnemen, en de $PM_{2.5}$ emissies met 30 procent. Verkeer is een belangrijke bron van ultrafijne deeltjes en koolstofhoudend fijn stof. Dieselroet is een mengsel van elementaire en organische koolstof. Verkeersemissies komen op leefniveau in woonwijken terecht en dicht bij belangrijke bevolkingsconcentraties.

Of een vermindering van fijn stof concentraties ook tot een evenredige vermindering van de gezondheidseffecten zal leiden is nog onzeker. Fijn stof is immers een complex mengsel, waarin sommige fracties meer gezondheidsrelevant zijn dan andere. Veranderingen in de samenstelling van dit mengsel kunnen van invloed zijn op de omvang en aard van de gezondheidseffecten. De meest kosten-effectieve maatregelen richten zich met name op de meest toxische fracties in het fijn stof. Helaas weten we momenteel nog niet met voldoende zekerheid welke fracties dat zijn. Waarschijnlijk behoren tot de voor de gezondheid relevante fracties het dieselroet uit de vervoerssector en fijn stof afkomstig van overige verbrandingsprocessen. Bepaalde fracties fijn stof in de buitenlucht, zoals zeezoutdeeltjes of het water in de stofdeeltjes, veroorzaken waarschijnlijk geen gezondheidseffecten. Een aantal epidemiologische studies doet vermoeden dat bodemstofdeeltjes minder relevant zijn voor de gezondheid dan stofdeeltjes afkomstig van verbrandingsprocessen. Toxicologische studies met zuiver ammoniumsulfaat en -nitraat (secundaire anorganisch aërosol) wijzen niet op een hoge toxische potentie van deze beide componenten, zelfs niet bij aanmerkelijk hogere concentraties dan in de buitenlucht. Wèl wordt in epidemiologische studies keer op keer een samenhang gevonden tussen gezondheidsklachten en aanwezigheid van sulfaat en nitraat als bestanddelen van secundaire anorganisch aërosol. Deze tegenstrijdigheid valt nog niet te verklaren.

Europese normstelling opnieuw bekeken
In 1999 heeft de Europese Unie de fijn stof normstelling voor 2005 en 2010 vastgesteld. De normstelling voor 2010 betreft een zogenoemde indicatieve waarde. Dat wil zeggen dat deze normen pas definitief worden vastgesteld na een evaluatie in 2003 van de ervaringen die in de diverse lidstaten zijn opgedaan met de normstelling van 2005. Bovendien zal rekening worden gehouden met de nieuwste wetenschappelijke inzichten. Deze EU normen zien er als volgt uit:

EU normen voor fijn stof (PM_{10})

	Fase 1 1 Januari 2005	Fase2* 1 Januari 2010
Jaargemiddelde	40 µg/m^3	20 µg/m^3
Daggemiddelde (24-uur)	50 µg/m^3	50 µg/m^3
Aantal overschrijdingen per jaar	35	7

*indicatieve waarde

In het document dat oorspronkelijk ten grondslag lag aan de EU normstelling werd aangekondigd dat beide EU fijn stof normen (daggemiddeld en jaargemiddeld) gelijkwaardig ofwel equivalent zouden zijn. In de praktijk blijkt echter dat in Nederland de huidige EU jaargemiddelde PM_{10} norm voor 2005 van 40 µg/m^3 niet equivalent is aan de daggemiddelde norm van 50 µg/m^3 met 35 toegestane overschrijdingen per jaar. Pas met **80** toegestane overschrijdingen per jaar zou een daggemiddelde norm van 50 µg/m^3 equivalent zijn aan de jaargemiddelde norm van 40 µg/m^3. Datzelfde geldt voor een daggemiddelde norm van 100 µg/m^3 met 7 overschrijdingen.
Om praktische redenen wordt de voorkeur gegeven aan een daggemiddelde norm van 100 µg/m^3 met 7 overschrijdingen boven een daggemiddelde norm van 50 µg/m^3 met 80 overschrijdingen. In het algemeen zal ook het publiek een hogere norm met een geringer aantal overschrijdingen beter kunnen begrijpen.
Hoewel de EU twee normen heeft voorgesteld, is er een aantal argumenten waarom voor fijn stof één norm toch voldoende is. De jaargemiddelde norm is dan de beste keuze. Om redenen van risicocommunicatie kan een daggemiddelde norm toch nuttig zijn. Aan beleidsmakers de keuze of er twee EU normen nodig zijn voor fijn stof.

Voldoet Nederland aan de luchtkwaliteitsnormen bij het huidige beleid?
De jaargemiddelde fijn stof concentraties die we in Nederland modelleren zijn consistent met onze metingen. We verwachten dat het voldoen aan een jaargemiddelde PM_{10} norm van 40 µg/m^3 in Nederland in 2005 wel haalbaar zal zijn, hoewel plaatselijke overschrijdingen op een aantal met name verkeersbelaste 'hot spots' niet uit te sluiten zijn. Daarentegen is het niet waarschijnlijk dat we in 2005 overal de daggemiddelde norm van 50 µg/m^3 met 35 toegestane overschrijdingen halen. Vooral in zo'n klein land als Nederland met een relatief grote buitenlandse bijdrage en een relatief hoge achtergrondsbelasting door zeezout wordt een daggemiddeld niveau van 50 µg/m^3 gemakkelijk overschreden.
De verwachting is dat zelfs als alle overeengekomen bestrijdingsmaatregelen in 2010 zullen zijn uitgevoerd volgens het '*current legislation scenario*' (CLE) er in Nederland toch nog 36 tot 40 overschrijdingen van de EU daggemiddelde norm van 50 µg/m^3 zullen zijn. De indicatieve EU normen voor 2010 van 20 µg/m^3 als jaargemiddelde en een daggemiddelde norm van 50 µg/m^3 met 7 overschrijdingen zijn dan ook voor Nederland geen van beide haalbaar.

Perspectief voor een verdere vermindering van fijn stof
Uiteindelijk is het technisch mogelijk om de Nederlandse PM emissies met nog eens 60 procent extra te verminderen (bovenop de al voorgenomen maatregelen volgens het *current legislation scenario*). Dit maatregelenpakket wordt "MFR_{ult}" genoemd: de ultieme Maximaal bereikbare Reductie. Dit maatregelenpakket kost jaarlijks ongeveer 6 miljard Euro. Gemiddeld over Nederland zal een extra reductie van de fijn stof

emissies met 60 procent leiden tot een 1.1 µg/m^3 lagere jaargemiddelde concentratie aan PM_{10}.
Daarnaast is nog een ander maatregelenpakket doorgerekend, dat aanmerkelijk goedkoper uitpakt. Hierbij wordt de emissiereductie beperkt tot maatregelen met een marginale kosten-efficiëntie van 55 Euro per bespaarde kg $PM_{10.}$ Dit maatregelenpakket, waarbij de Nederlandse emissies met ongeveer 25 procent ofwel een kwart afnemen, wordt "$2010_{quart\ red}$" genoemd. Het pakket kost jaarlijks 210 miljoen Euro. Gemiddeld over Nederland zal een reductie van de fijn stof emissies met 25 procent leiden tot een 0.3 µg/m^3 lager jaargemiddelde concentratie aan PM_{10}. Dat lijkt maar een kleine vermindering van de huidige concentraties fijn stof in Nederland. Plaatselijk worden echter forsere reducties verwacht. Zo leidt het pakket "$2010_{quart\ red}$" plaatselijk tot verminderingen van 1 tot 5.5 µg/m^3. De maximale reducties kan men bij dit pakket verwachten in het dichtbevolkte Rotterdam. Interessant is ook dat de op het verkeer gerichte maatregelen in het pakket "$2010_{quart\ red}$" alleen betrekking hebben op de scheepvaart. Als we ons concentreren op de waarschijnlijk meer verkeersgerelateerde fracties van fijn stof, zoals dieselroet, dan zijn de gemodelleerde verminderingen zelfs relatief belangrijker. De al genoemde pakketten "$2010_{quart\ red}$" en "MFR_{ult}" leiden tot een afname van het Nederlandse dieselroet afkomstig van de transportsector van respectievelijk 20 procent en 50 procent. Als ook in het buitenland vergelijkbare maatregelen worden getroffen, wordt de aanpak nog effectiever.

Aanvullende fijn stof bestrijding kan gebaseerd worden op het 'voorzorgbeginsel'. Bij verdere brongerichte acties kan men zich richten op het zo kosten-efficiënt mogelijk terugdringen van ofwel de totale PM_{10} massa ofwel op de waarschijnlijk meer gezondheidsrelevante fractie daarvan. Aan de laatste optie wordt de voorkeur gegeven. Deze fracties zijn waarschijnlijk verkeersgerelateerd dieselroet of meer in het algemeen verbrandinggerelateerde PM emissies. Daarom is het bestrijden van de nu nog vrijwel onbestreden scheepvaart emissies een bij uitstek kosten-effectieve optie. Het bestrijden van andere verbrandinggerelateerde bronnen zoals industriële verbranding, stoken van openhaarden en mobiele werktuigen is ook mogelijk, maar minder kosten-effectief. Ook aanvullende maatregelen in het kader van het klimaatbeleid kunnen de verbrandinggerelateerde fijn stof emissies helpen terugdringen.

Welke risico's blijven nog over bij een verbeterde luchtkwaliteit?
Een aanzienlijk deel van de PM_{10} niveaus in Nederland is niet door beleidsmaatregelen te beïnvloeden, aangezien het afkomstig is van natuurlijke bronnen. Naarmate de door mensen veroorzaakte emissies verder worden teruggedrongen, stijgt het aandeel van de natuurlijke bronnen. Er is meer inzicht nodig in de chemische samenstelling en in de bijdragen van de diverse bronnen aan de merendeels natuurlijke en tot nog toe meestal niet gemodelleerde fijn stof fracties in de lucht. Daaruit valt af te leiden in hoeverre het fijn stof probleem uiteindelijk door milieumaatregelen kan worden aangepakt.

Ook al wordt het fijn stof probleem aangepakt, de bijbehorende gezondheidsklachten zullen niet van de agenda verdwijnen, integendeel. In Nederland worden mensen steeds ouder en wellicht leiden ook andere demografische ontwikkelingen tot een meer dan evenredige toename van extra gevoelige bevolkingsgroepen. Hoe dat uitpakt is de vraag, maar bij een nog zo slecht begrepen fenomeen als fijn stof blijft voortdurende waakzaamheid geboden.

1. Introduction

In 1998, a proposal was made for a daughter directive for *inter alia* PM_{10} based on the European Union (EU) Framework Directive on Ambient Air Quality, c.f. Table 1.

Table 1 EU standards for PM_{10}.

	Phase 1 1 January 2005	Phase 2* 1 January 2010
Annual average	40 µg/m^3	20 µg/m^3
Daily average (24-hour)	50 µg/m^3	50 µg/m^3
Number of exceedances per year	35	7

*indicative value

This PM_{10} standard is to be evaluated in 2003 and brought into line with new scientific developments in knowledge about the effects of PM_{10} on health and the environment. In addition, the practical experience of Member States in applying the standards, as well as the feasibility of meeting the standards, are to be considered.

The association of ambient PM with serious health effects caused the World Health Organisation (WHO) to provide guidelines for PM (WHO, 2000). Because of the health risks, the European Union (EU) provided standards for PM in a daughter directive (1999/30/EC) of the EU Air Quality Framework Directive (96/62/EC). The values and averaging times are presented in Table 1.

For the Dutch government these air quality standards gave rise to five questions concerning PM, to which it would like an answer before evaluation of the EU daughter directive on PM in 2003.

- How do the various indicators of PM compare as relevant for the causation of health effects?
- What is the relationship between concentrations of ambient PM and health effects in order to make a substantiated choice of PM standard?
- What are the actual PM concentrations in the Netherlands and how big are the contributions of the different source categories to these concentrations?
- Which indicator of PM is preferable if, besides health relevance, risk management considerations are also taken into account?
- What is the quantification of the total source-risk chain now and in the foreseeable future?

These five questions will be answered in subsection 7.2 of this report, to the extent our current knowledge permits.

The EU daughter directive on PM naturally forms the basis of this project. To answer the five questions put by the Ministry, we need to establish which information was available when the daughter directive was drawn up and how this information may have changed over the last five years. The then available scientific information was set down in a position paper by a group of EU experts under the chairmanship of Bernt Seifert: ‘Ambient air pollution by particulate matter’, final version dated 8 April

1997. (http://europa.eu.int/comm/environment/air/pp_pm.pdf) The original scientific information covered the pollutant description, risk assessment, measurement and cost. The first of these is the pollutant description (PM_{10}). New research suggests that there are currently other descriptors like finer PM, ultrafine (UF) or source-related PM that also need to be considered for purposes of standard setting. The second aspect is that of the averaging time. Risk estimates based on recent measurements in the Netherlands suggest that either standard would lead to similar risk estimates. The monitoring of PM is the third aspect. In the position paper and the accompanying documents the need for the use of a correction factor was indicated because the semi-volatile fraction of PM is only partially measured in the currently used automatic measuring devices. Compliance assessment is difficult in these circumstances. The fourth aspect is that of cost. For this last, but certainly not least, aspect more information has again become available since 1997. The EU has decided that a new position paper is warranted in 2003 and is working on it.

This report has been prepared for a number of Dutch Ministries in the context of the Netherlands Aerosol Programme to facilitate a Dutch position in the evaluation process of the EU PM directive in 2003. The Netherlands Aerosol Programme was instigated at the request of three Ministries, that of Housing, Spatial Planning and the Environment (VROM), that of Transport, Public Works and Water Management (VW) and that of Economic Affairs (EZ). It is being conducted jointly by the Netherlands Institute of Public Health and the Environment (RIVM, Bilthoven), the National Organisation for Applied Scientific Research (TNO, Apeldoorn), the Energy Research Foundation (ECN, Petten) and the Institute for Risk Assessment Studies (IRAS, Utrecht).

In July 2001, the Netherlands Aerosol Programme distributed a discussion document on health risks of particulate matter in ambient air. This document, also known as the 'orange document', was discussed *inter alia* at a speciality workshop held on 6 September 2001 following the annual ISEE conference, which took place that year in Garmisch Partenkirchen, Germany. Some 30 experts from Europe and the US attended this speciality workshop and discussed the ideas presented in the discussion document. A second formal opportunity for international discussion arose during a two-day workshop in June 2002 with some 20 experts from the Netherlands and US-EPA in Research Triangle Park (North Carolina) in the United States. Judging by the discussions that followed, including those outside the regular workshops, the 'orange document' fulfilled its purpose well. The feedback we received helped us improve the original document.

However, on a number of issues the scientific evidence available regarding the role of PM is as yet indecisive. Weighing the current evidence, experts sometimes arrive at different conclusions. This report, therefore, by no means contains definitive answers. The whole process we went through has resulted in the common ground presented here. This position will constitute a major element in the Dutch contribution to evaluation of the EU daughter directive on ambient PM in 2003.

In this report we present the answers to the Ministry's five questions on PM in relation to the review of the EU daughter directive based on the current level of our knowledge. Though a number of other questions will undoubtedly remain after this report has been read, in some respects the report is quite unique. It is the first PM risk

evaluation in which the annual average PM_{10} concentration based on emission inventories, assessments and modelling is brought into balance with PM_{10} measurements. It is the first risk evaluation of PM based on epidemiological data for seven years of PM_{10} measurements and health effects occurring in one country. It comprises more than 100 million person years of epidemiological data for one country. It is the first report to have this data set, allowing us to look at the quantitative behaviour of PM risks in space (urban versus rural) and in time.
The report comprises the following Sections:

- Section 2 is devoted to the physico-chemical information relating to air quality in the Netherlands. The problems involved in measuring PM are briefly described and an overview presented of the typical chemical composition, mass balance and typical size distribution of PM in the Netherlands. This section on air quality goes on to describe the emissions, modelling and current ambient concentrations of PM. Complementary to the anthropogenic sources, the natural sources of PM and their contribution to ambient levels in the Netherlands are also described. The overview concludes with information on the typical ambient average levels and the levels that can be seen during episodes in the Netherlands. As a bridge to the next section, the relevance of the central site monitor to personal exposure is described.
- Section 3 covers the epidemiological evidence for PM-associated public health problems. First of all, a brief overview is given of the different health risks associated with ambient PM. The most recent information on the form of the concentration-response relationship in the Netherlands is also presented. The possibility of a risk to a specific population is explored, as are the health effects in the Dutch population based on short-term exposures, daily average values and the effects to be expected from chronic exposure to air pollution. The health risks for the Netherlands are quantified on the basis of the most recent concentration-response relationships. This section concludes with a paragraph on the relevance of PM as a proxy or causal factor by way of an introduction to the next section on toxicology.
- Section 4 is devoted to the available toxicological evidence. It describes the different chemical fractions of PM and the corresponding doses in airways and lungs. In this section, space is given to the *in vitro* experiments with PM and the resulting dose-effect relationships. Next, the *in vivo* experiments are presented, leading on to a paragraph on human clinical evidence concerning PM and the resulting dose-response relationships. On-site toxicology using an Ambient Fine Particle Concentrator (AFPC) is also described. Concluding this section, the available toxicological information from foreign scientific literature concerning possible mechanisms and routes of action is evaluated for its relevance in the light of the knowledge of concentrations of these specific fractions in the Netherlands.
- Section 5 gives a description of the standards that could be used to prevent or mitigate PM-associated health effects. The scientific evidence for a specific size-related PM standard or for a specific averaging period is evaluated.
- Section 6 is devoted to the reduction of PM health problems. Triggered by the indicative annual average value for PM_{10} in 2010 of 20 $\mu g/m^3$ compared with the 40 $\mu g/m^3$ standard in 2005, the question of whether a reduction of PM levels will also lead to a similar reduction in health risks is discussed. Reducing emissions of PM and the cost of abatement measures are treated jointly, together with the question concerning the PM levels that may be expected in the future. Because of

their possible significance for heath risks, some additional information is presented on ultrafines (UF) and their sources. It goes without saying that specific measures for traffic are also elaborated.

- Finally, Section 7 presents the summarised conclusions and the answers to the five questions put by the Ministry based on the available evidence. As more results of Dutch and foreign research become available in the future, the current conclusions will undoubtedly need to be revised in line with advancing understanding.

2. Physico-chemical characteristics of ambient PM

2.1. General information

Particulate Matter (PM) has both a *primary* component, which is emitted directly by sources, and a *secondary* component, which is formed in the atmosphere via chemical reactions of gases. Examples of primary man-made sources are traffic and industry, while natural sources include soil particles blown by the wind, and sea salt. The most important secondary particulate matter derives from sulphur dioxide, oxides of nitrogen, ammonia and volatile organic carbon compounds. Both primary and secondary PM can be of natural or man-made (anthropogenic) origin. PM can also be the vehicle for heterogeneous chemical reactions of gases, which produce new and sometimes highly reactive components like nitrous acid.

PM can be emitted and exist in the atmosphere in a wide range of particle sizes. Particles with an aerodynamic diameter of 10 µm or less are called inhalable, as this is the fraction that is inhaled by humans. Particulate Matter within a specific size range, a 50% cut-off diameter of 10 µm, is referred to as PM_{10}. Particles with a 50% cut-off diameter of less than approximately 4 µm can end up deep in the lungs even. From a human health point of view, PM generally larger than 10 µm in diameter and forming a large part of TSP (Total Suspended Particulates) is probably of less concern, because we do not inhale it. From the point of view of nuisance dust, large particles or TSP can be a problem, though. Information on the PSD (Particle Size Distribution) is essential to understand the potential health effects of PM. The deposition of PM in the respiratory tract and lungs (inhaled dose) varies with particle size. Apart from its size distribution, the chemical speciation and bioavailabilty of PM is information that is also needed to understand the health effects of PM. For an understanding of the current health-related problems of PM, a distinction is generally drawn between three size ranges. The smallest particles are the ultrafine particles or ultrafines (UF), which are less than 0.1 µm in diameter. The next size class is that of the particles smaller than 2.5 µm (but bigger than 0.1 µm) and the largest is the coarse fraction of PM_{10} in the range between 2.5 and 10 µm.

Other particle measures are sometimes reported as well, and as they will be presented in this report, too, they need to be mentioned here. In the USA, measurements taken using nephelometers have been used to estimate $PM_{2.5}$. In the past, Black Smoke measurement, based on the decreased reflectance or increased light absorbance of filters loaded with soot, was used to indicate the total particulate matter concentration. The US alternative, the coefficient of haze (CoH), seems to be a less reliable parameter (Allen and Koutrakis, 1999). BS particles are mostly smaller than a few µm. The calibration curve for BS is based on the measured concentrations of PM in the UK in the 1950s. Although the ambient mix of PM has changed, the original calibration curve is still used, as nothing better is available.

2.1.1. Size, composition and sources

Primary fine PM emitted by diesel engines is of a carbonaceous nature. It is also known as DEP (Diesel Exhaust Particulate) or ‘soot’. Coarse particles, and to a lesser extent fine particles as well, are linked more to primary emissions from mechanical

processes or the handling of dusty materials. Crustal material re-suspended as a result of turbulence caused by traffic, wind-blown soil particles or PM caused by sea spray and wood burning also result in fairly coarse particles. However, there is no strict relationship between size distribution and source of emissions, as natural emissions of sea salt, for example, contain both fine and coarse particles. The high ambient relative humidity in the Netherlands and the hygroscopic nature of the SIA here are the reasons for the droplet shape being the most common form of the average $PM_{2.5}$ particles in this country.
Ultrafine particles are formed during high-temperature combustion processes of either mobile sources or fossil fuel-based power production. However, in the summer UF also forms as a result of photochemical reactions (Birmilli and Wiedensohler, 2000; Tuch *et al.*, 1997; O'Dowd *et al.*, 1998).
The atmospheric residence times and hence ranges of travel of these different size fractions vary considerably. Table 2.14 gives the average residence times and travelling distances of the five size classes of aerosol differentiated in the OPS model. They range from more than sixty hours to less than two hours for different size classes.

Apart from its size range and emission sources, PM can also be characterised by its chemical composition. The chemical composition of PM in ambient air depends on the contribution made by both anthropogenic and natural sources. The natural sources mainly consist of primary emissions of sea salt, wind-blown soil dust and some secondary organic particulate matter. The chemical composition of PM varies in accordance with these different emission sources. One can distinguish carbonaceous PM or PM_{carb} (elemental carbon (EC) and organic particulate matter (OC)), secondary PM or PM_{sec} (ammonium salts of nitrates and sulphates), also known as SIA (Secondary Inorganic Aerosols), and natural PM or PM_{nat} (sea salt, crustal material, sulphate, nitrate and secondary organic particulate matter from natural emissions of precursors). The generic term SOA (Secondary Organic Aerosol) may comprise aerosols formed from natural as well as man-made emissions. Elemental carbon and ammonium salts are chemically well-defined, while organic PM, sea salt and crustal material consist of mixtures of chlorides, oxides of metals and silicates, and a wide range of organic compounds. Heterogeneous atmospheric reactions of ambient PM with gases can in the right circumstances result in highly reactive products like nitrous acid, which in turn is a source of the reactive OH. Owing to their high reactivity, these components are hard to measure. These considerations need to be taken into account when crude particle measures like concentrations of PM_{10} or $PM_{2.5}$ are compared with health effects.

A substantial part of the $PM_{2.5}$ in the Netherlands consists of semi-volatile components like ammonium nitrate. This causes sampling and measurement problems, as will be discussed in detail in subsection 2.2.1.1.

2.1.2. PM in the Netherlands

The Netherlands Aerosol Programme (NAP), which is the subject of this report, studied ambient PM concentrations in the Netherlands. Of importance in this respect were measurements of and model calculations made for PM_{10} and $PM_{2.5}$ at rural, urban and industrial locations during the one-year period 1998–1999 (Visser *et al.*, 2001). Relevant emission sources were identified using source receptor modelling in

combination with dispersion modelling and national and European emission inventories for PM. In addition, special studies focused on estimating contributions to PM concentrations in the Netherlands made by emission sources outside Europe (Weijers *et al.*, 2000), wind-blown soil particles and re-suspended road dust (Keuken *et al.*, 1999) and on background information relating to EC/OC and ultrafine (UF) particles (Ten Brink and Keuken, 2000). Some of the research connected with the NAP is still ongoing.

The physical and chemical aspects, as well as the concentrations, of PM in ambient air vary in accordance with the relative contribution made by different emission sources. Consequently, PM concentrations and the physico-chemical properties of PM show temporal and spatial variation even in a relatively small country like the Netherlands. The average results of PM research in the Netherlands conducted on three spatial scales and in six locations – street, urban background and rural background – during the period from August 1998 to September 1999 were presented by Visser *et al.* (2001). The estimate for the annual average of PM_{10} in the Netherlands was compiled from this project and from the above references and is presented in Table 2.1 below. A default value of 10% unknown and/or water was assumed.

Table 2.1 Estimate of the annual average PM_{10} contribution to aerosol levels in the Netherlands from various sources in the late 1990s.

	Subtotal µg/m^3	PM_{10} µg/m^3	Source
Primary natural			
Sea salt		4-7	(Visser et al., 2001)
Crustal material		2	(Visser et al., 2001)
Subtotal natural	6-9		
Primary anthropogenic			
EU and NL contribution (non-traffic)		4-7	(Visser et al., 2001)
Traffic (EU and NL)		2-5	(Visser et al., 2001)
Netherlands road dust		1-2	(Keuken et al., 1999)
Subtotal primary anthropogenic	7-14		
Secondary anthropogenic (SIA)			
European contribution		7-9	(Visser et al., 2001)
Dutch contribution		3-5	(Visser et al., 2001)
Subtotal SIA	10-14		
Other			
Northern Hemisphere background		1	(Weijers et al., 2000)
Unknown 10%		3-4	Default
Subtotal	4-5		
Total PM_{10}	27-42		

Natural secondary organic aerosol is not included as it represents only a small fraction (<< 10%) of the anthropogenic contribution.

The range of 27– 42 µg/m^3 in annual average PM_{10} reflects the uncertainties and spatial variation in the estimated contribution made by various sources to PM_{10} in the Netherlands. Measurements in the air quality monitoring network for 1998 and 1999 resulted in annual PM_{10} levels in the non-urban (regional) parts of the Netherlands of

33 and 32 μg/m^3 respectively (Hammingh, 2001). In the large cities, assessed urban background concentrations still exceeded 40 μg/m^3 in the 1990s. All the NAQMN values of PM_{10} were corrected by a factor of 1.3 for volatilisation losses; see subsection 2.2.1.1. The different PM concentrations on the three spatial scales – regional, urban background and street – will be treated in 2.3.
The light blue shaded areas in Table 2.1 are the only sources for which emission data are currently known and included in European and Dutch emission databases. Only for those sources the dispersion models can present calculated yearly average values. The other sources can not be modelled in the classical dispersion models because of lack of data and have to be estimated by way of a non-modelled fraction, for the Netherlands this fraction has been determined at 18 μg/m^3 as a country average. This value will be used later on to bridge the gap between dispersion models and actual measurements of ambient PM_{10}.

2.1.3. Introduction to the chapter on physico-chemical characterisation

This subsection is structured such that the measurement of PM_{10} and its ensuing problems are presented first in Section 2.2. Section 2.3 presents the typical ambient levels of PM_{10} in the Netherlands on three spatial scales: regional, urban background and urban street. These have been divided into two time domains: annual average and smaller time scales. In 2.4 the natural contribution to PM_{10} levels in the Netherlands is explored and an estimate is made of the part of PM_{10} that cannot be influenced by European abatement measures. In 2.5 the modelling of PM_{10} is presented. This section starts with an overview of emissions both in the Netherlands and in the countries within the modelling area mentioned. Then, the models for calculating the annual average (OPS) and daily average values (EUROS/LOTOS) are presented, as well as the results they produced. Section 2.6 compares the results of models and measurements, showing that a previously observed 'gap' between measurements and modelling has been adequately closed. In Section 2.7 the divide between ambient air quality and epidemiologically relevant data will be bridged via a discussion of the relationship between PM levels as measured at measuring sites and personal exposure.

2.2. Mass measurement of PM

2.2.1. General methods

The standard procedure in Europe for measuring PM consists of sampling the desired fraction using an appropriate inlet and a filter. The filters are equilibrated for constant weight at a prescribed temperature (20 ^{0}C) and relative humidity (50%) before and after sampling, and the mass of aerosol is determined as the difference in weight before and after sampling.

There is a standard procedure for sampling PM in the EU, which is CEN standard EN12341. Unfortunately, the reference instrument, the WRAC (Wide Range Aerosol Classifier), is so cumbersome to use that for daily monitoring of PM other instruments are employed. These have, of course, been compared with the reference instrument. Another fact that should be considered is that the prescribed filter material gives rise

to large artefacts in the collection of PM, as described in a special review paper on measurement methods for compliance in the US (Chow *et al.*, 1995).

RIVM's National Air Quality Monitoring Network (NAQMN) in the Netherlands performs continuous PM measurements using an FAG-type β-dust monitor (Van Elzakker, 2000). Other countries (UK, France) use the 'TEOM'. Both instruments (TEOM and FAG) have pre-treatment of the aerosol by heating, which introduces losses of semi-volatile components; see 2.2.1.1. below. To correct for these losses the measured concentrations were increased by 30%. This factor of 1.3 was also presented by Williams *et al.* (2001) in a guidance note on PM_{10} monitoring to EU member states and in intercomparisons with the reference method.

2.2.1.1. Uncertainties in PM measurements

A problem in measurements of mass concentrations of PM is the behaviour of the so-called semi-volatile compounds, both of inorganic and organic nature. An important inorganic species by mass in the Netherlands aerosol is ammonium nitrate. For instance, Figure 2.1 (Schaap *et al.*, 2002) shows that nitrate is a major compound in the aerosol of a rather large area of Europe. For the Netherlands, ammonium is also an important component on account of our extensive husbandry on a relatively small surface. These factors therefore make ammonium nitrate a highly relevant contributor to aerosol mass levels in the Netherlands. Ammonium nitrate is especially liable to losses.

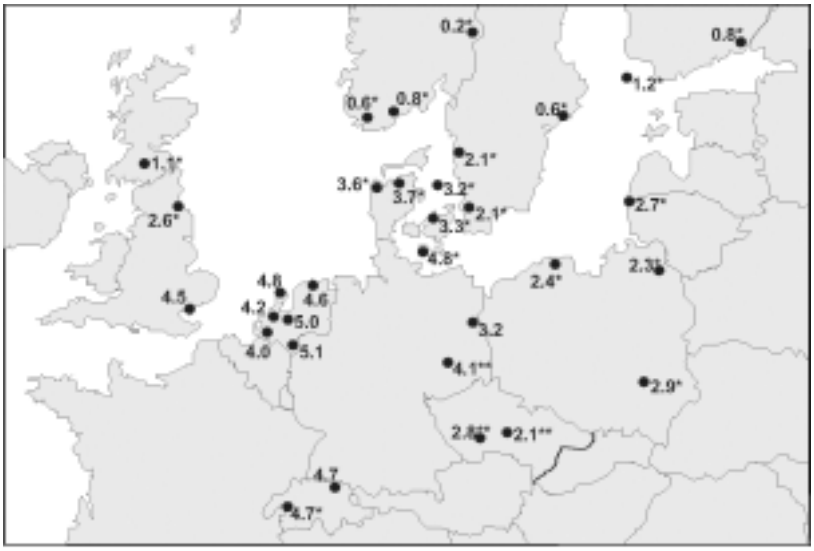


Figure 2.1 Nitrate in aerosol (μg.m^{-3}), annual average, (Schaap et al., 2002).

Losses of ammonium nitrate may influence the measured PM levels not just in the Netherlands, but in a large area of northwest Europe, Switzerland and Italy. The losses occur because ammonium nitrate is in equilibrium with nitric acid and ammonium. This equilibrium shifts during sampling. As sampling in the Netherlands is carried out for a 24-hour period, this procedure may result in an unpredictable rate of loss of ammonium nitrate (e.g. Herring and Cass, 1999 and Keuken 1989). Losses depend to a great extent on the filter material. Glass fibre and cellulose will retain

nitrate, Teflon filters will not, while quartz fibre filters (as prescribed by CEN) will show losses.

The annual PM_{10} average – established in the National Air Quality Monitoring Network in the Netherlands – was corrected by a factor of 1.3. In view of the advantages of an automatic monitoring network, this is an acceptable approach for long-term and national surveys. However, in the case of local surveys aimed at establishing PM_{10} and $PM_{2.5}$ concentrations near roads and highways, a factor of 1.3 may not be adequate to correct for losses of ammonium nitrate due to *spatial and temporal* variations in this component. This issue is relevant for the PM_{10} standard in the EU regulations, which involves a 24-hour 90- or 98-percentile standard. The effect of ammonium nitrate on PM_{10} and $PM_{2.5}$ gravimetric measurements was illustrated by sampling with a denuder/filter pack to remove interfering nitric acid and to collect all evaporating ammonium nitrate. The results are presented in Table 2.2 (Spoelstra and Keuken, 2002). This sampling method allows 'true' PM_{10} and $PM_{2.5}$ concentrations and the size of the 'loss' of ammonium nitrate to be established.

Table 2.2 Average PM_{10} and $PM_{2.5}$ 48-hour concentrations at Overschie (50 m from a highway) and in the Pleinweg (street canyon) plus the ammonium nitrate concentrations in April, June and October 2001 in µg/m^3 and number of measurements.

April

	PM_{10}	$PM_{2.5}$	NH_4NO_3	N
Highway	23	18	2	4
Street canyon	30	17	2	4

June

	PM_{10}	$PM_{2.5}$	NH_4NO_3	N
Highway	34	26	10	2
Street canyon	37	29	10	2

October

	PM_{10}	$PM_{2.5}$	NH_4NO_3	N
Highway	46	31	20	3
Street canyon	50	38	20	3

Ammonium nitrate is a secondary aerosol, the spatial concentrations of which in the region of Rotterdam do not vary significantly at the locations Overschie and Pleinweg. However, the temporal variations – related to ammonia emissions from agricultural activities – range from 2 to 20 µg/m^3 for different months. Consequently, the percentage of ammonium nitrate in PM_{10} and $PM_{2.5}$ varies from 10% in one month to more than 50% in another. Hence, in some months the PM_{10} and $PM_{2.5}$ concentrations need to be corrected by a factor of 2 in order to obtain a 'correct' measurement of PM_{10} and $PM_{2.5}$. As a consequence, this correction might result in exceedance of the EU 24-hour PM_{10} standard of 50 µg/m3. This indicates that the use of a general correction factor is not adequate for accurately assessing concentrations on a shorter time scale than annual averages. More on correction factors can be found in 2.6.1.1.

2.2.2. Continuous monitors

The standard instrument used in the NAQMN is the FAG (β-dust monitor). The FAG works on the principle that any mass sampled on the filter substrate will more or less uniformly attenuate β-radiation. The sampling air is heated (to 50 ^{0}C) in order to remove water from aerosol particles. However, this heating also removes semi-volatiles and influences quantification of the PM levels. The resulting error depends on the chemical composition of the aerosol and is compensated for by applying a constant correction factor of 1.3. For the Netherlands these losses have been studied by Van Arkel *et al.* (2001a and 2001b) and Van Putten *et al.* (2001a and 2001b).

The TEOM measures the mass by impacting aerosol particles on a piezo-electric crystal. Again, water interferes, so water from aerosol particles is removed by heating the sampled air, with at least similar losses as described previously for the FAG. Van Putten *et al.* (2001a and 2001b) described that for Dutch urban aerosols the correction factor of a TEOM may, under specific urban circumstances, even rise to 1.6 when compared with a KFG (Klein Filter Gerät) as a reference instrument. Recent intercomparisons, over a whole year, of TEOMs with other instruments in the Netherlands (in the Rijnmond area around Rotterdam) have indicated that for average Dutch circumstances the TEOM correction factor is 1.9 (Voerman *et al.*, 2001 and Van den Elshout *et al.* 2001).

A very promising new development to avoid evaporation and correct for adsorption by gases is the 'differential' TEOM. In this monitor the air in one sample line is dried without heating and, in addition, the aerosol is periodically removed using an electrostatic precipitator. The advantage of this is that it does not adsorb gases. Very few data have as yet been gathered using the instrument and additional research is required to show how well it works as a regular monitoring device.

2.2.3. Black Smoke and carbonaceous PM

The elemental and the organic carbon contents of PM are considered to be relevant for the health effects of PM. The so-called 'black smoke' method is a *qualitative* method developed in the 1950s to measure the 'blackness' of particles by a relatively simple, light-reflectance method. For the then prevailing *coal* smoke, the degree of blackening of a sampled filter was calibrated to the mass concentration. However, particles emitted by different combustion sources have a different blackening effect, which besides composition also depends on the particle size. For example, the blackening effect of diesel emissions is three to four times greater than that of coal emissions. Hence, for ambient PM there is no quantitative relationship between reflectance (of a sampled filter) and particle mass. The 'black smoke' (BS) method is still widely applied as a 'screening' method for black particles in ambient air and for trend analysis, due to its long history in ambient air measurement. The original calibration curve for BS (OECD method) is based on the measured concentrations of PM in the United Kingdom (UK) in the 1950s. However, the ambient mix of PM has changed, so the original calibration curve has lost its meaning. The current reflectance measurement of BS filters has been found to correlate highly with elemental carbon and is therefore a useful surrogate for primary traffic-related emissions.

Since 1980, the measurement of elemental carbon (EC) has been used as a surrogate for soot. The method involves the temperature-controlled heating of a sample and distinguishing between organic compounds (OC) and elemental carbon (EC). Differentiation into carbon fractions is not straightforward and apportioning has been the subject of several intercomparison trials. Differences of up to a factor of three were common. However, it seems that in the most recent intercomparison, for urban aerosol, agreement between the most accepted methods has improved (Schmid *et al.*, 2001).

The dominant problem in the assessment of particulate organic compounds (OC) in ambient air is the sampling artefacts resulting from the adsorption of adsorptive organic gases and the evaporation of semi-volatile aerosol OC.

OC is a mixture of primary organic compounds (e.g. incompletely burned diesel and lubricants) *and* secondary particles formed by the photochemical oxidation of hydrocarbons (e.g. terpenes). OC is measured as mass *carbon* content. It is assumed that the ratio of the mass of organic compounds to their carbon content is around 1.4. The results of the analysis are therefore multiplied by 1.4 to estimate the mass of OC. However, there are others using factors ranging from 1.2 to 1.9, so OC data should be regarded as semi-quantitative at best.

2.2.4. Other methodologies for PM analysis

All methodologies based on sampling by way of filters followed by the extraction of sampled aerosol material and analysis are subject to errors due to losses of semi-volatile organic or inorganic material. To overcome these problems, ECN has developed a Steam Jet Aerosol Collector (SJAC). Slanina *et al.* (2001) have described this instrument. The SJAC measures the composition of aerosol as a function of the size of the aerosol particles. The results of measurements taken in the Netherlands are given in Figure 2.2.

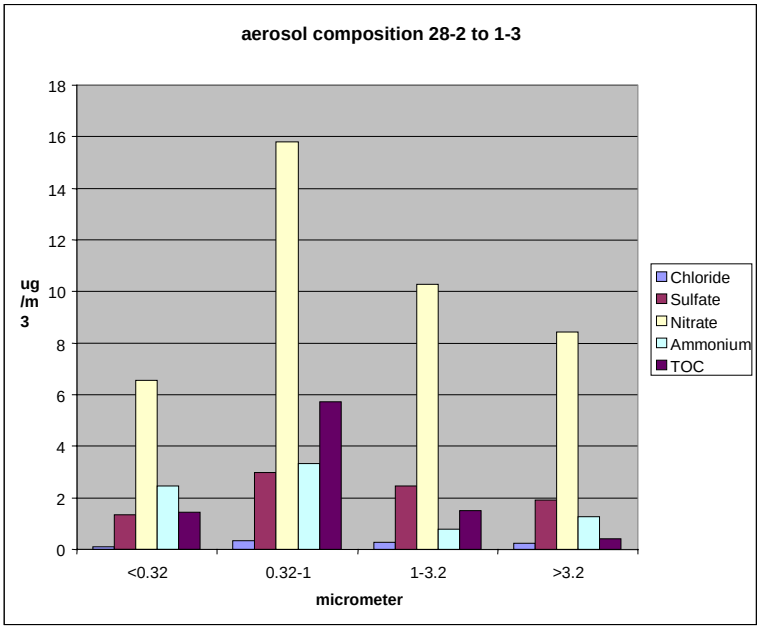


Figure 2.2 Average ammonium, sulphate, nitrate, chloride and TOC concentrations in 4 size classes, measured during a two-day period of easterly winds at ECN (28 February and 1 March 2000).

The sum of nitrates and total organic carbon (TOC) during the two-day period presented in Figure 2.2 is 40 μg/m^3 and 9 μg/m^3, versus 6 μg/m^3 for sulphate. About 80% of the nitrate is in fine particles (< 2.5 μm). These results illustrate the importance of ammonium aerosol in the Netherlands. Only systematic monitoring with artefact-free methods can show the extent of the uncertainties caused by volatilisation.

2.3. Ambient levels of PM

In this section the ambient levels of PM_{10} in the Netherlands are presented by showing the spatial distributions and trends of annual average PM_{10} concentrations, and ambient daily and episodic levels. Meteorology has an impact, too.

The measurements presented here were collected via the Dutch National Air Quality Monitoring Network (NAQMN) operated by the RIVM. The equipment and methods used to monitor particulate matter are described in several reports (Van Elzakker, 2000; Visser *et al.*, 2001). The PM_{10} data from the RIVM network are multiplied by a factor of 1.33 to correct for a systematic underestimation by the sampling equipment (Hammingh, 2001). Figure 2.3 shows the locations and the number of PM_{10} monitoring sites in the Netherlands in 2000. The National Air Quality Monitoring Network (NAQMN) comprises three types of station:

- Regional stations (green in Fig. 2.3).
- Urban background stations chosen so that the number of passing vehicles within a radius of 35 m from the station does not exceed 2,750 per 24 hours (black in Fig. 2.3).

- Street stations are located close to a major road so that the number of passing vehicles within a radius of 35 m from the station is at least 10,000 per 24 hours (red in Fig. 2.3).

In the near future the number of PM_{10} locations will be substantially increased due to implementation of the first EU daughter directive on air quality (Van Breugel and Buijsman, 2001).

Figure 2.3 Monitoring sites for the determination of PM_{10} (Van Elzakker, 2000). The blue points are urban background stations, the green points regional background stations and the red points street stations. The numbers and corresponding names of stations can be found in Table 2.3.

In the first European daughter directive for air quality, the standard for PM_{10} is set at 40 $\mu g/m^3$ as the annual average. The daily standard is set at 50 $\mu g/m^3$ and may not be exceeded more than 35 times a year. A standard with 35 exceedances per year is virtually identical to a 90-percentile. These limit values are to be met in 2005. With this in mind, the ambient levels presented here are related to these values.

2.3.1. Annual average of PM

The assessed spatial distribution on 5 x 5 km^2 grids of annual average PM_{10} in 2000 is shown in Figure 2.4, while the values actually measured are presented in Table 2.3. The spatial distribution in Figure 2.4 was assessed on the basis of measurements taken by the NAQMN and knowledge of spatial patterns, which are used to interpolate the measurements.
Figure 2.4 shows that the annual average PM_{10} concentrations increase from the north to the south, partly because of the increasing influence of the Dutch sources, which are predominantly located in the southwest part of the Netherlands. Added to this,

sources from outside the Netherlands contribute more in the south and the east of the country. In 2000, the value of 40 µg/m^3 was assessed to have been exceeded in a limited number of urban and industrial areas near Amsterdam, Haarlem, Rotterdam and The Hague. Spatially averaged over the Netherlands, the annual average PM_{10} concentration was 31 µg/m^3 in 2000, cf. Figure 2.4. In the summer of 2000 (from April 1 to October 1), average PM_{10} concentrations were 3 µg/m^3 lower than in the winter of that year.

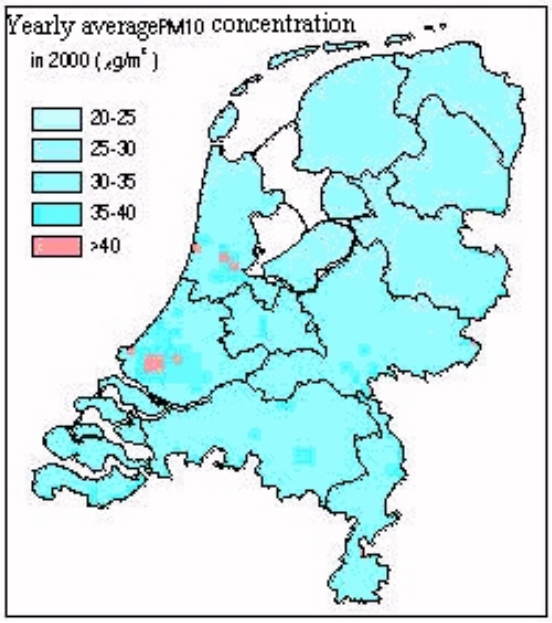


Figure 2.4 Spatially averaged, assessed annual concentrations of PM_{10} in 2000.

The annual average PM_{10} concentrations measured are presented in Figure 2.5. The nationwide average based on regional stations is decreasing (statistically significant with a 95% confidence level), as is the annual average PM_{10} concentration at three regional stations (Vredepeel #131, BiestHoutakker #230 and Witteveen #928). These trends and the variations in the annual meteorological conditions are discussed in more detail in subsection 2.3.1.2.

Table 2.3 Annual average PM_{10} levels in $\mu g/m^3$ measured in NAQMN.

National Air Quality Monitoring Network									
Year	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**
# **Regional station**									
131 Vredepeel	50	43	38	43	40	37	35	35	36
133 Wijnandsrade	39	37	42	39	38	34	29	30	31
230 BiestHoutakker	46	48	48	47	45	40	37	34	36
318 Braakman	44	41	42	46	45	42	33	34	33
437 Westmaas	42	36	39	42	41	37	35	34	32
444 DeZilk		31	37	41	38	34	30	31	29
538 Wieringerwerf	32	34	34	43	33	30	33	31	25
722 Eibergen	36	37	37	44	40	33	29	28	28
724 Wageningen	43	42	39	43	44	36	34	33	32
928 Witteveen	36	33	32	35	32	27	27		
929 Valthermond								29	30
# **Urban background station**									
404 The Hague	41	41	42	47	41	44	43	34	36
418 Rotterdam		41	42	47	42	42	38	36	37
441 Dordrecht		37	40	43	39	33	32	32	29
520 Amsterdam	41	37	41	49	39	38	35	33	29
# **Street station**									
236 Eindhoven	41	42	45	49	42	39	36	35	35
433 Vlaardingen	40	39	41	49	43	39	36	32	32
641 Breukelen		42	41	46	43	38	40	37	31
639 Utrecht	42	40	47	54	49	36	37	35	37
728 Apeldoorn	39	39	42	46	42	36	34	33	37

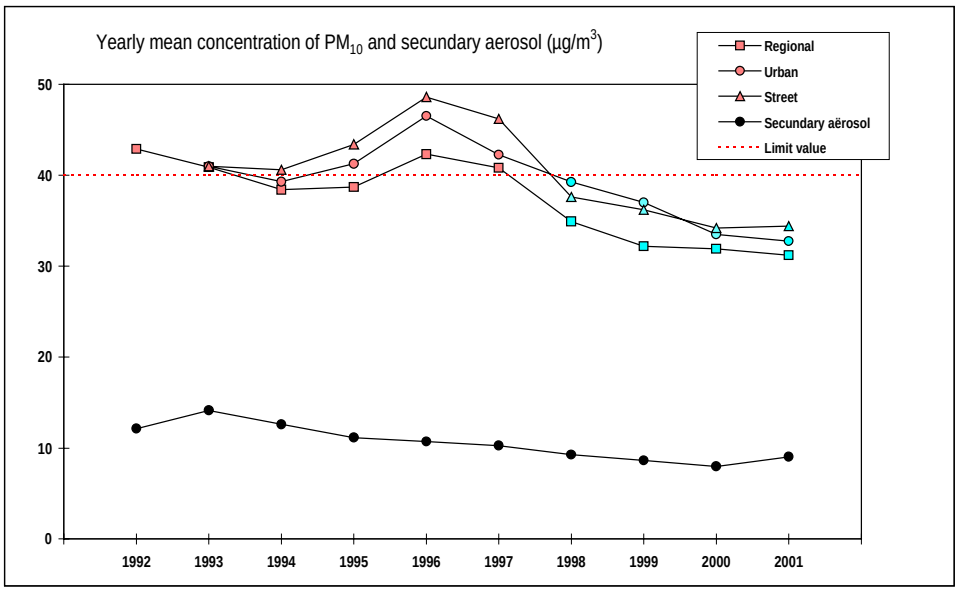


Figure 2.5 Annual average PM_{10} concentration in the Netherlands based on measurements from regional, street and urban background sites. Averaged regional concentrations of Secondary Inorganic Aerosol (SIA) are also presented.

2.3.1.1. Historical trends

Regular nationwide measurements of PM_{10} in the Netherlands only started in 1992. However, in order to assess the long-term health impact of PM it would be helpful to have some insight into the trends in historical levels of PM in the country. Owing to the lack of a real nationwide network of monitoring stations, the PM picture that will be presented here will be of a kaleidoscopic nature and only available for specific fractions of PM: black smoke (BS) and total suspended particulates (TSP). If inferences have to be made from pre-1960s PM concentrations, this will have to be based on longer-term measurements taken in the UK. Obviously, going back further in time results in larger uncertainties in the PM levels.

In the Netherlands, black smoke (BS) and TSP have to be used as a surrogate for PM. The first measurements of BS date back to 1962 (see Figure 2.6). The BS concentrations in the more densely populated west of the country fell from more than 30 to 5 µg/m^3, the current average of the 50-percentiles of 10 regional sites. This decrease is the result of the shift from coal and oil to gas for residential heating in the Netherlands. Although more coal-fired power was generated in the Netherlands in the 1980s than in the preceding decades, aerosol emissions from power production were drastically reduced due to the introduction of electrostatic precipitators and do not have any measurable influence on the nationally averaged BS levels. From the mid-1970s, road traffic became a major source of black smoke in the Netherlands. In the 1980s, the BS emitted by light-duty diesel vehicles was reduced by a factor of 2, although this was offset by the growing number of vehicles (Donkelaar, 1995). In the 1990s, the introduction of three-way catalytic converters further reduced aerosol emissions from light-duty vehicles.

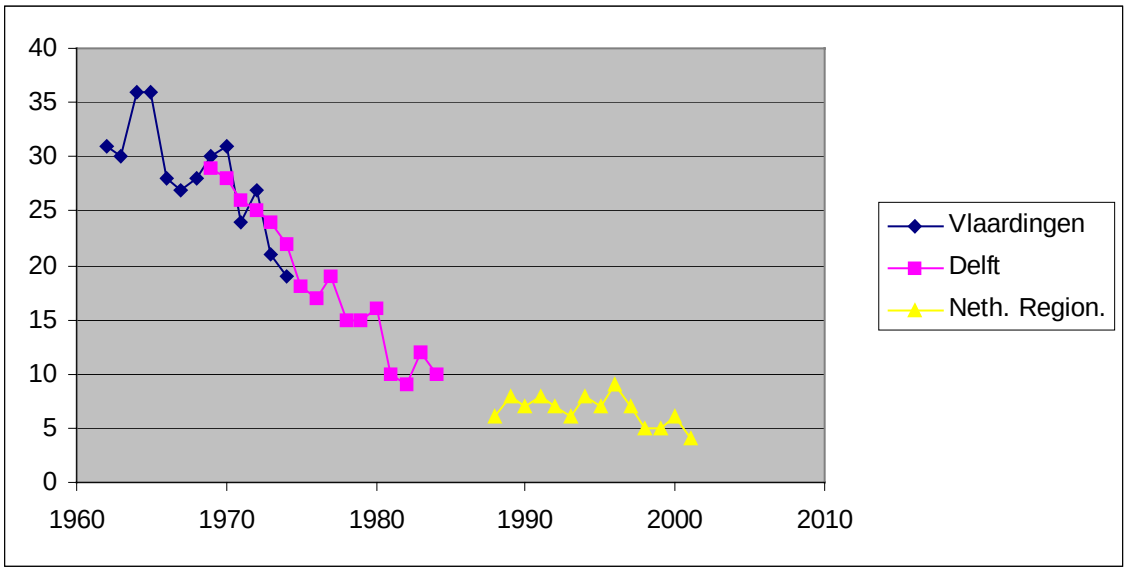


Figure 2.6 50-percentiles of BS levels in µg/m^3 in Vlaardingen and Delft 1962–1984 and Netherlands regional average of 10 sites 1988 –2001 (Sources: Lanting, 1986 and Jaaroverzicht Luchtkwaliteit, 2001 and before).

Because of the gradual changes in composition of BS and more in particular to changes in the 'blackening capacity' of PM after a shift from predominantly coal to sub-micron diesel soot, there is no simple linear relationship between historical BS levels and historical PM_{10} levels.

A trend similar to the one for BS can be found in the TSP concentrations in Rotterdam from 1970 to 2000, as is shown in Figure 2.7. This coarser part of PM has decreased considerably in the Netherlands over the past decades.

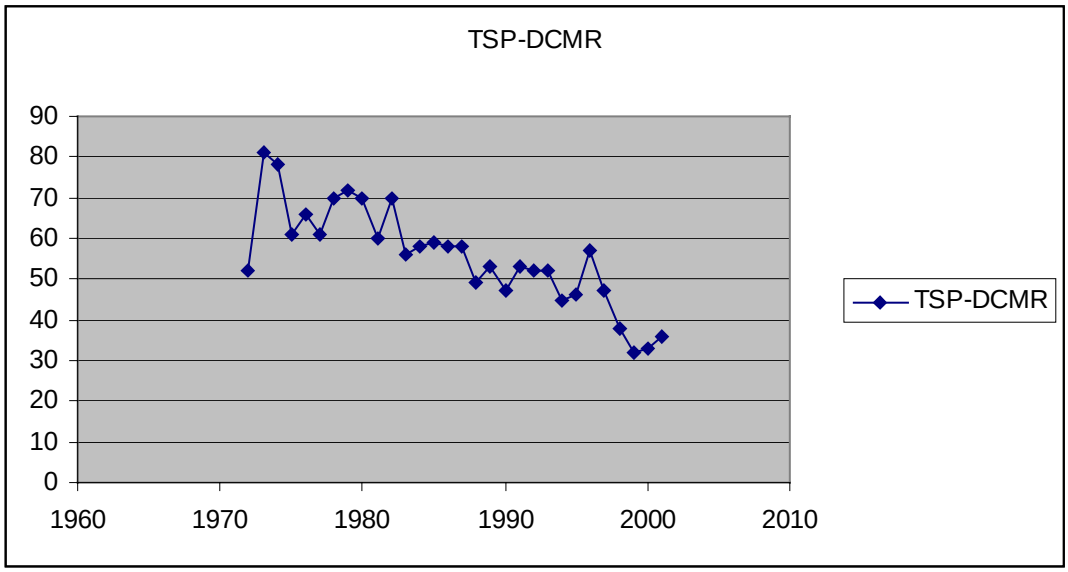


Figure 2.7 TSP in µg/m³ in Rotterdam (Source: DCMR, 2002; Lucht in cijfers 2001).

It was not possible to find a longer time-series of PM measurements in the Netherlands than the two presented in Figures 2.6 and 2.7. Despite its patchwork appearance, it can be concluded that the ambient levels of PM_{10} – lying somewhere between BS and TSP – will have shown a similar downward trend during the last few decades.

2.3.1.2. Current trends corrected for meteorology

Series of measurements are generally analysed in order to show trends in concentrations that are affected by human activities. The point is to clarify how emissions have an impact on ambient PM concentrations. On the one hand, continuing economic growth results in higher production and emissions, while on the other environmental abatement policies and the abatement of emissions results in the mitigation of concentrations. A long-term time-series of measurements may produce an understanding of the impact of environmental policy.

A further influence on ambient concentrations is that of meteorology. The influence of weather is not constant over time. For PM we know that a cold winter results in higher than average concentrations in the Netherlands. These higher PM concentrations are partly the result of more transport from abroad, as sub-zero temperatures are usually accompanied by a continental air flow that is generally more polluted than air masses from the Atlantic Ocean. During a period of frost the general dispersion conditions are

such that inversions also occur more often. Furthermore, emissions tend to be higher during these periods as the heating of homes consumes more energy and cold starts of cars produce more pollution. During a long cold spell, high wind speeds may also re-suspend airborne crustal material from the then barren and dry fields. A year with more rain will result in lower concentrations of PM, not only because of rain-out or wash-out, but also because it is harder for crustal material to become re-suspended when the soil is moist.

At RIVM a method has been developed for analysing the influence of meteorology on air pollution trends (Dekkers and Noordijk, 1997). This method uses the ambient concentrations in the Netherlands in combination with daily-average meteorological values. It has been further expanded to include the analysis of PM. The method has four steps:

(i) classification of the meteorology,
(ii) calculation of correction factors and the addition of missing meteorological classes,
(iii) calculation of an annual average PM value corrected for meteorological conditions, and
(iv) model validation by diagnostic checks, crossvalidation and comparison to rival models.

The meteorological factors that influence the concentrations of PM are divided into different classes by way of Regression Tree Analysis (RTA, Breiman *et al.*, 1984). This is a statistical technique that distributes the set of measurements into two subsets, based on meteorological criteria. The criterion for the division of the subsets is the minimisation of the variance of the two subsets. After this first step in the Regression Tree Analysis, one of the subsets itself is again divided into two new subsets, again using the criterion of minimisation of variance. This eventually produces a ‘tree’ of classes that describe the influence of meteorology on PM concentrations. An important difference between Regression Tree Analysis and, for example, Multiple Regression (MR) is that RTA can deal with highly non-linear relations between concentrations and meteorology.
The meteorological correction of the trend is effected by feeding meteorological class correction factors per year into the model (Stoeckenius, 1991). The frequency of occurrence of a meteorological class is determined for each year. This frequency determines the actual value of the correction factor. For example, if the meteorological ‘days of tropical heat’ class normally occurs three times a year and in a specific year there are six of these days, the calculation of these tropical days in the annual average is less than average and the correction factor becomes 0.5. However, if for some reason a certain meteorological class does not occur, an estimate is made of the expected PM concentrations by using concentrations from other years in that specific class.
Finally, a low-pass filter is applied to the meteorologically corrected concentrations, allowing the estimate of 90% confidence limits. In this case the fit is based on a second order polynomial (a 'parabola fit'). The filter separates an emission-related concentration trend and noise, and assumes consistency in the measurements and a smooth change in emissions.

The PM time-series from RIVM's NAQMN regional stations for the years 1992–2001 were used for this analysis. These results are therefore representative of the regional background concentrations of PM in the Netherlands. The meteorological information was provided by the Royal Dutch Meteorological Institute (KNMI) and contains variables such as temperature, rainfall, sunshine, wind direction and wind speed, radiation intensity, relative humidity, etc.

An initial analysis was performed on the daily average PM_{10} levels. This, however, resulted in an explained variance of only 35% in the PM_{10} data. For the next analysis, based on the monthly average PM_{10} values, the variance explained by meteorology rose to 70%. This remarkable difference is most probably explained by the phenomenology of PM_{10}. PM_{10} stays in the ambient air for a couple of days until it is removed by either dry or wet deposition. The size of the Netherlands and the average wind speed mean that ambient concentrations are influenced to a large extent by the situation in neighbouring countries. The meteorology, however, is only that for the Netherlands, and its variance on a daily basis does not say very much about the situation in the rest of northwest Europe. Monthly average meteorology in the Netherlands is influenced more by large-scale meteorology and so is a better basis for the meteorological correction of PM_{10}.

The main meteorological factors that explain the variance are rainfall, temperature and wind direction. Continental wind directions result in higher concentrations, as do days with subzero temperatures. At the coastal stations, temperature seems to be the main variable; in the east and south of the Netherlands rainfall is a stronger explanatory variable for the local concentrations of PM_{10}. The lowest monthly averages varied from 20 to 30 $\mu g/m^3$ and the highest values reached were 60 to 70 $\mu g/m^3$.

After calculation of Regression Trees for all regional stations individually, we averaged concentrations and meteo-corrected concentrations for all nine stations. In this way we end up with two nation-wide regionally-averaged curves, presented in Figure 2.8.

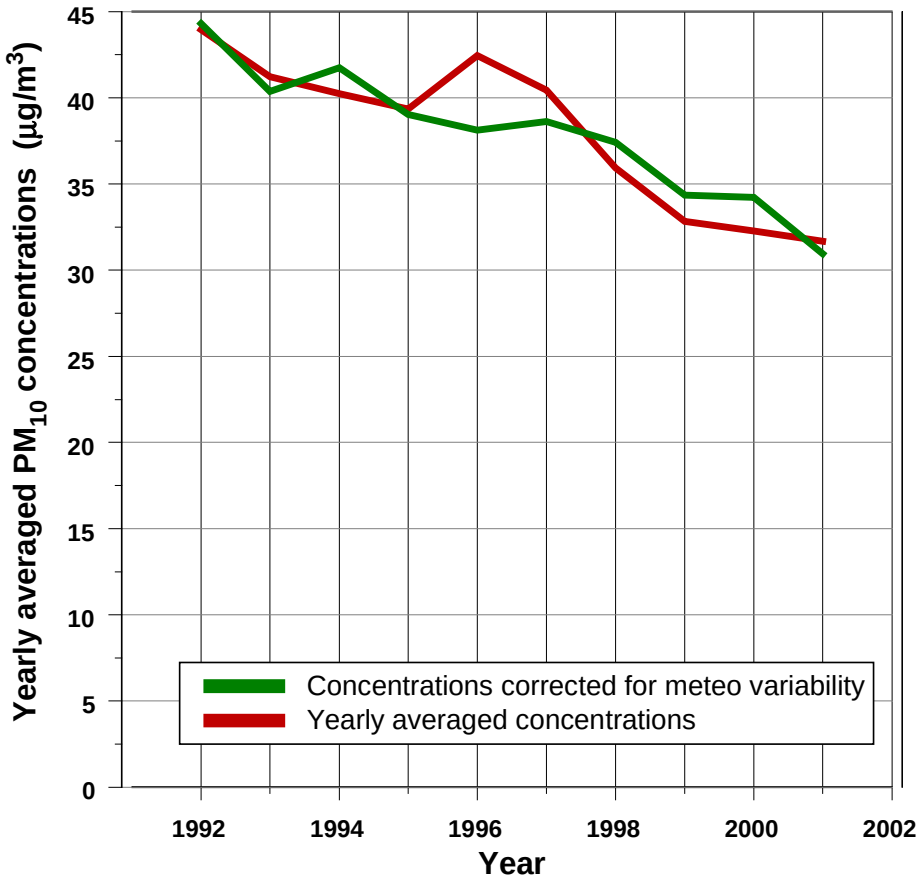


Figure 2.8 *Yearly-averaged concentrations for nine regional PM_{10} stations in the Netherlands and the corresponding meteorologically corrected concentrations.*

The annual-averaged concentrations clearly show a downward trend over the past decade. Concentrations decrease from 43.3 µg/m^3 in 1992 to 31.7 µg/m^3 in 2001 (red curve in Figure 2.8). In other words, a decrease of 11.6 µg/m^3 occurs in a ten-year period; relative to the year 1992, this decrease accounts for 27%. The year 1992 was the starting year of the PM_{10} monitoring network. We have estimated that concentrations in this specific year were probably ~1 µg/m^3 too high. Therefore the decrease is slightly lower: **~25%.**

After correction for meteorology there seems to be difference in the PM concentrations for the period 1992-1998 and the post-1998 period. Up to 1998 the decrease in PM levels seems limited. Afterwards, the decrease in concentrations is somewhat speeded up.

Due to the cold and dry winter of 1996, and to a lesser extent of 1997, PM levels are elevated, yielding a lower concentration than was to be expected. An extremely cold winter may lead to PM concentrations, which, on an annual-average basis, are elevated by approximately 4 µg/m^3. For the individual stations this elevation varies from 2 to 7 µg/m^3.

The fact that PM_{10} concentrations are somewhat lower (~ 2 µg/m^3) at the end of the nineties (1998, 1999, 2000), can be attributed to meteorological circumstances too. There were fewer days with sub-zero temperatures than usual, and winters were predominantly wet. At the same time wind speeds in the spring-summer-autumn period were relatively high.

We conclude that

- the decrease of PM_{10} over the past 10 years accounts for ~11 µg/m^3. Relative to 1992 this implies a decrease of ~25%;
- this decrease is not influenced by meteorological variability;
- in individual years the annual average PM_{10} concentration may be *lower* by ~2 µg/m^3 for wet and mild years with wind speeds slightly higher than average. In years with extreme cold and dry winter months the annual average PM_{10} concentrations may be *elevated* by ~ 4 µg/m^3.

2.3.2. Smaller scale temporal variations in PM

2.3.2.1. Hourly variations

The hourly data from all 19 stations from 1998 to 2001 were checked to see whether or not they had a log-normal distribution by means of a goodness-of-fit test. Data from 16 of the 19 monitoring stations deviated from this distribution for a small range of highest concentrations. In practice, however, almost any goodness-of-fit test will reject the null hypothesis if the number of observations is large, since 'real' data are never distributed according to any theoretical distribution (Conover, 1980). The PM_{10} distribution for all the stations was estimated to be 'close enough' to the theoretical distribution, so that fairly accurate results would be provided as the standard variation (GSD) by assuming a log-normal distribution. This avoided possible analysis problems which otherwise might have been caused by the FAG's high detection limit of 10 µg/m^3.

In the Netherlands there is a high correlation (R=0.8) between the daily maximum hourly value and the 24-hour average value of PM_{10}. This finding may be relevant for future epidemiological results, as data have been presented orally at conferences in which there also seems to be an association with shorter term variations in PM and health effects in susceptible subgroups.

a. Wind direction and PM_{10}

Three-hourly moving averages of monitored PM_{10} concentrations were used to analyse the correlation of aerosol levels with wind direction and wind speed. Hourly PM_{10} concentrations measured by FAG fluctuate randomly and contain less information than the three-hourly averaged values. Wind directions from six meteorological stations in the KNMI monitoring network were used; data from the meteorological stations were linked to the nearest NAQMN monitoring stations, after time-synchronisation.

Figure 2.9 Average PM_{10} concentrations, classified by wind direction, for all monitoring stations between January 1998 and January 2001. The circle is the value of the highest PM_{10} concentration at all stations. The blue wedge at the top denotes average PM_{10} concentrations in conditions with widely differing wind directions.

The average PM_{10} concentration classified by wind direction between January 1998 and January 2001 is presented in Figure 2.9. The large circle represents the highest concentration level recorded for all stations and can be used to facilitate comparison of concentration levels between monitoring stations. The position of the sites can be found in Figure 2.3 and the names in Table 2.3.

The main wind direction is SW and the associated wind speeds highest. The highest PM_{10} concentrations, however, are consistently found for most of the Netherlands with easterly to southeasterly winds and low wind speeds. This suggests that subcontinental-scale PM_{10} transport makes a considerable contribution. This is in line with modelling, which calculates a substantial contribution of foreign anthropogenic PM_{10} to ambient Dutch PM_{10} concentration levels. The blue wedge, supposedly pointing north in Figure 2.9, is indicative of the influence of local PM_{10} sources in periods of low wind speeds or fluctuating wind directions.

2.3.2.2. Daily variations

The spatial distribution of the number of days in 2000 when the daily average PM_{10} concentration exceeded 50 µg/m^3 is presented in Figure 2.10. This value is exceeded on more than 35 days in large parts of the Netherlands. The largest numbers of days with concentrations above 50 µg/m^3 are found in the central southern part of the Netherlands.

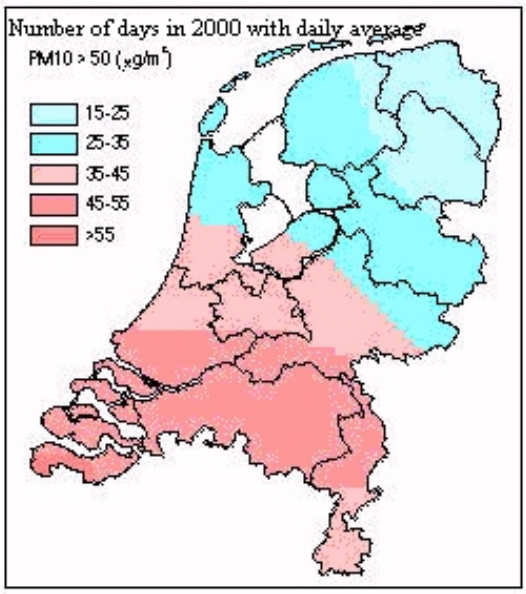


Figure 2.10 Spatial distribution of the number of days when the PM_{10} concentration exceeded 50 µg/m^3 in 2000.

The absolute number of days in 2000 when the daily standard of 50 µg/m^3 was exceeded is shown in Figure 2.11. This number is currently falling as a consequence of reductions in emissions of SIA precursors (i.e. SO_2) and a downward trend in PM_{10} emissions.

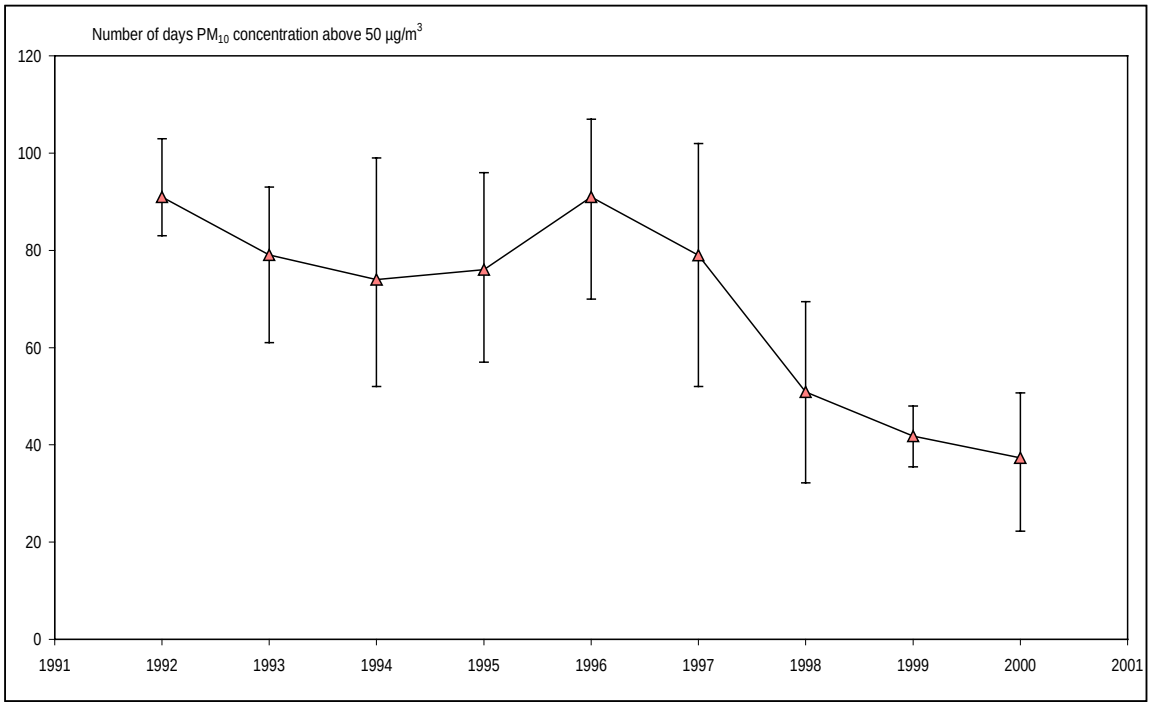


Figure 2.11 The annual average number of days when the PM_{10} value of 50 µg/m³ was exceeded at stations. (**Note**: the bars denote the highest and lowest values at the NAQMN stations and **not** a 95% CI).

a. Description of PM_{10} 'episodes' > 100 µg/m³

As an indicator of PM 'episodes', exceedances of the daily PM_{10} level of 100 µg/m³ were collected at the regional stations Vredepeel (# 131), Wieringerwerf (# 538), De Zilk (# 444) and Wageningen (# 724) for the period 1992–2001; see Table 2.4.

Table 2.4 Number of days when the PM daily average of 100 µg/m³ was exceeded at the regional stations Vredepeel (# 131), De Zilk (# 444), Wieringerwerf (# 538) and Wageningen (# 724). Blank means no data available.

	#131	#444	#538	#724
1992	18[1]	.	6[1]	14
1993	18	.	6	16
1994	12	0[1]	6	8
1995	3	4	0	6
1996	15	14	16	9
1997	9	4	2	6
1998	3	3	2	4
1999	1	0	3	1
2000	1	0	0	0
2001	3	2	1	3

[1] Monitoring started in March of the year.

The numbers of days when daily PM_{10} averages of 100 $\mu g/m^3$ were exceeded decreased for all stations during this period, as did the annual averages. Also, the magnitude of the exceedances appears to be decreasing. In the period 1992–2001, a daily average PM_{10} value of 150 $\mu g/m^3$ was exceeded at stations # 131, # 444, # 538 and # 724 summed over the whole period on respectively 11, 3, 7 and 9 days. All these exceedances occurred before 1998. The duration of the exceedance is on average 1.5 days, with an observed maximum of five days. It should be noted that a number of exceedances in the Netherlands are due to fireworks around New Year's Eve.

Meteorology of episodes

All the 67 occurrences of a PM_{10} level of 100 $\mu g/m^3$ and above at the Wageningen # 724 monitoring station were analysed to discover whether there were similar meteorological conditions which increased PM levels. The meteorological data from the Royal Dutch Meteorological Institute (KNMI) in De Bilt, a distance of 40 km from Wageningen, were used for this analysis. The results show that levels in excess of 100 $\mu g/m^3$ correspond with conditions in which daily average wind speeds are low to very low (average of 2.4 m/s). Most days had no or very little precipitation, relatively high atmospheric pressure (average of 1022 hPa, where the climatological average atmospheric pressure is 1013 hPa) and high cloud cover. The prevailing wind direction was east to south. No days with levels in excess of 100$\mu g/m^3$ occurred with wind directions between west and northwest.

For the years between 1992 and 2001 the days with PM_{10} levels of 100 $\mu g/m^3$ and above were also categorised per month. In the different months of the year, from January to December, the numbers of exceedances were 12, 17, 10, 6, 2, 1, 0, 3, 2, 4, 7 and 3 respectively. In the three months from January to March, 39 of the 67 days with levels exceeding 100$\mu g/m^3$ occurred, while in the six months of the period from May to October only 12 days were observed when levels exceeded 100$\mu g/m^3$.

Typical episode

A period of higher PM levels with daily averages above 100 $\mu g/m^3$ occurred at four regional stations between 18 and 20 January 2001. This period is now described as an example of a PM episode; see Figure 2.12. The buildup and the breakdown of the episode is described using meteorological data from the KNMI.

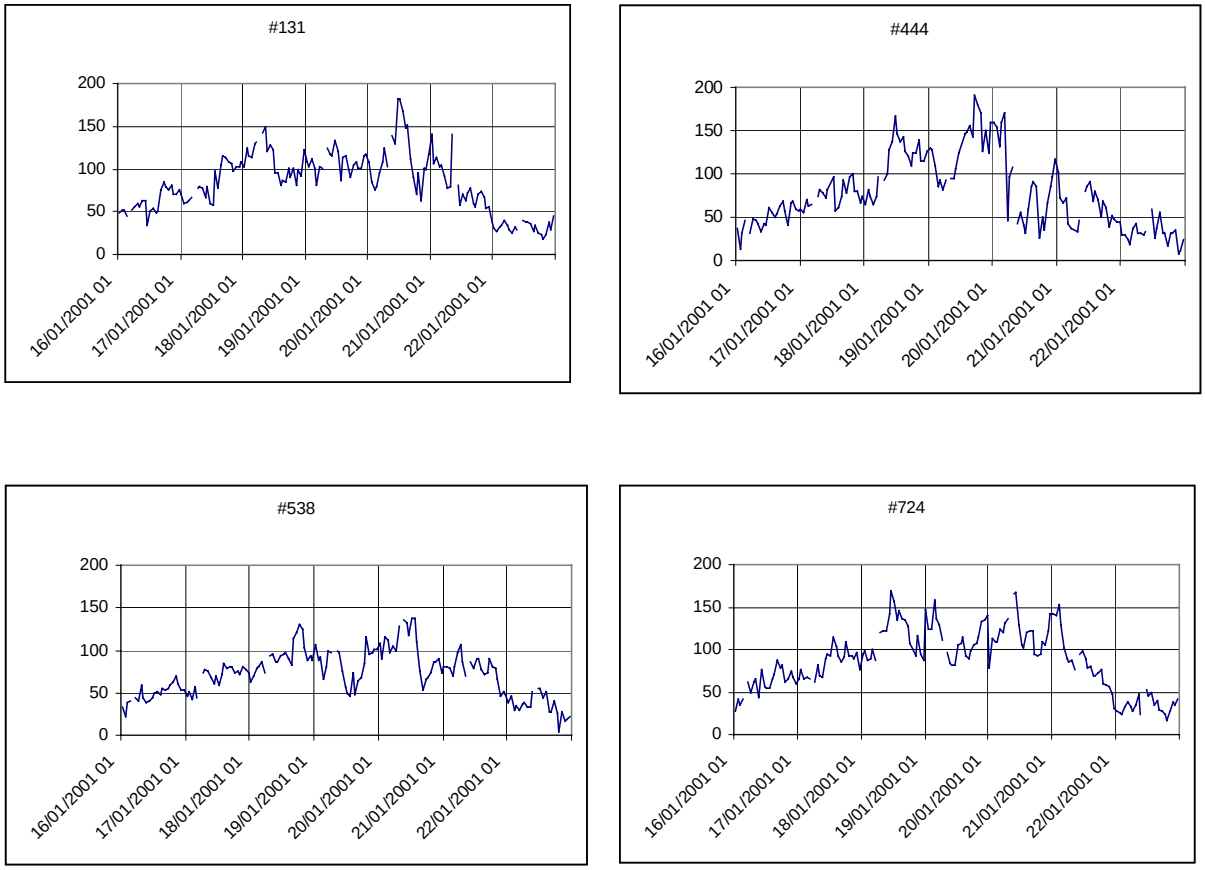


Figure 2.12 The hourly PM_{10} concentration in µg/m^3 at the regional stations Vredepeel (# 131, situated in the southeast of the Netherlands), De Zilk (# 444, situated in the west near the coast), Wieringerwerf (# 538, situated in the northwest) and Wageningen (# 724, situated in the centre of the Netherlands) for the period 16–22 January 2001.

In the period 16–17 January, easterly winds brought dry and cold air into the Netherlands. The period of 18–20 January was characterised by a low pressure system, which travelled from Germany across the Netherlands. Wind speeds were low and PM_{10} levels increased up to 150 µg/m^3. There was also dense, sometimes very dense fog. The four regional stations observed persistent high PM_{10} levels during this period. In the period of 21–22 January, southwesterly winds arose. With wind speeds increasing, PM levels fell back to more normal levels of 30–40 µg/m^3.

b. Frequency distribution of daily averages

According to the first EU daughter directive (99/30/EC), the PM_{10} limit value for the daily average of 50 µg/m^3 may be exceeded on a maximum of 35 days a year by 2005. In 2000, about 75% of the Dutch population was exposed to more than 35 days in excess of the daily value of 50 µg/m^3 (Hammingh, 2001). Measurements show that the number of times the EU daily standard is exceeded is decreasing, from an average of 91 days in 1992 to on average 37 days in 2001.

An analysis was made of the PM_{10} frequency distribution of regional, street and urban stations in the years 1993, 1995, 1998, 2000 and the entire period 1993–2000 in order

to check for possible trends in the distributions of daily averaged PM_{10} concentrations over the years. For a valid comparison the frequency distributions need to be corrected for missing values. This was done by scaling the distribution to a full measurement year. It should be noted it was therefore assumed that the occurrence of missing values was random and independent of meteorological conditions.

In Figure 2.13 the frequency distribution of one site is presented as an example. The distributions of all monitoring stations were analysed by means of a Kolmogorov-Smirnov test. The two-sample Kolmogorov-Smirnov goodness-of-fit test was used to test whether two sets of observations could reasonably have come from the same distribution. The results show that the frequency distribution (p= 0.05) of daily average PM_{10} values for most of the monitoring stations from various years between 1992 and 2001 differed, implying that the character of the frequency distributions is slowly changing. It can therefore be concluded that gradual differences in PM_{10} frequency distributions exist over the years and can be quantified. As it has been demonstrated in previous subsections that a log-normal distribution for PM_{10} concentrations can be assumed, the PM_{10} frequency distributions can therefore be described using the variables Geometric Mean (GM) and Geometric Standard Deviation (GSD). These two variables are calculated by taking the logarithm of each daily measurement value, determining the mean and standard deviation, and then calculating the exponential value of these variables.

In the period 1992–2001, average PM_{10} concentrations decreased. Table 2.5 presents the geometric mean and geometric standard deviation, averaged for the regional (N = 11) and for the urban/street stations (N = 8). It can be concluded that the geometric means of both the regional and the urban stations are decreasing (statistically significant), consistent with PM emission reductions. Clear statistical trends in the geometric standard deviation in this period cannot be found, which is consistent with meteorology as a driving force behind daily concentration differences. An analysis of the influence of meteorology on the annual average trends of PM_{10} was presented in subsection 2.3.1.2. For the calculation of a 98-percentile for standard setting in Section 5.3, an average GSD in the Netherlands of 1.7 will be used.

Table 2.5 The geometric mean (GM) and geometric standard deviation (GSD), averaged for all the regional and urban/street stations between 1992 and 2001.

	Geometric Mean		Geometric Standard Deviation	
Year	Regional average	Urban average	Regional average	Urban average
1992	38.6	38.2	1.64	1.58
1993	34.3	33.5	1.86	1.88
1994	33.3	34.7	1.74	1.66
1995	34.4	38.5	1.65	1.56
1996	37.1	41.1	1.72	1.65
1997	35.7	37.2	1.61	1.58
1998	31.6	34.7	1.59	1.51
1999	30.0	32.9	1.55	1.48
2000	29.7	30.7	1.52	1.49
2001	28.5	30.2	1.55	1.50

Figure 2.14 shows the trend for the numbers of days in different classes of the frequency distribution for the average of all regional stations (N = 11) and for the urban/street stations (N = 8) between 1992 and 2001. These trends are in general consistent with those for the individual PM_{10} monitoring stations. It can be concluded that occurrences of daily average concentrations of between 20 and 30 $\mu g/m^3$ increased considerably between 1992 and 2001; occurrences of daily average concentrations of between 30 and 40 $\mu g/m^3$ increased slightly in this period. It is not possible to detect clearly any changes in occurrences of daily average concentrations of between 40 and 50 $\mu g/m^3$, while occurrences of daily average concentrations in excess of 50 $\mu g/m^3$ decreased over this period. There are no clear differences in trend behaviour of the distributions of regional stations compared with that of urban/street stations.

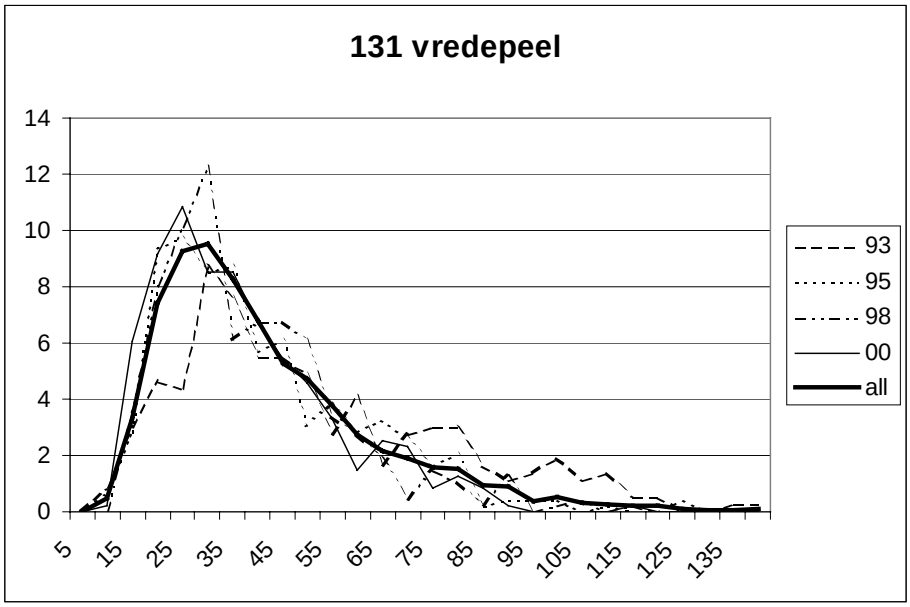


Figure 2.13 Example of a frequency distribution of daily PM_{10} measurements for Vredepeel (# 131) station between 1993 and 2000.

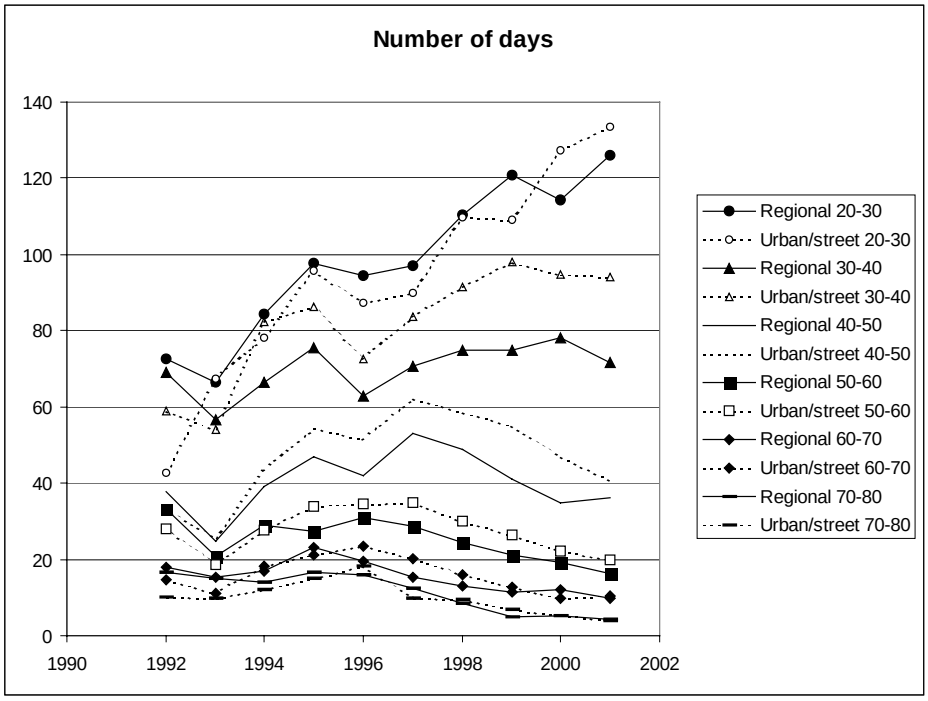


Figure 2.14 Trend of the numbers of days in different classes of the frequency distribution between 20 and 80 µg/m^3, averaged for all regional stations (N = 11) and for all urban stations (N = 8) between 1992 and 2001.

The frequency distribution can also be used to estimate the effect of a possible future change in the daily standard to the expected number of days exceeding this standard.

The EU daily standard for 2005 may not be exceeded on more than 35 days a year. In 1998, exceedance of the daily standard occurred on 61 days, averaged over all the monitoring stations. Specified, the regional stations recorded on average 53 days above 50 µg/m^3, whereas the urban stations and street stations recorded 75 and 68 days respectively on which the daily standard was exceeded. In 2001, the daily standard was exceeded on average on 37 days. Regional stations recorded 35 days above 50 µg/m^3, whereas the urban stations and street stations recorded 32 and 43 days respectively.

It should be clear, however, that the total numbers of days on which the daily standard was exceeded was not limited to this relatively small number. If the whole of the Netherlands is considered, the standard was exceeded on 166 days in 1998 at one monitoring station at least. In 1999, 170 days were recorded on which the 24-hour standard of 50 µg/m^3 was exceeded at a monitoring station somewhere in the Netherlands.

The relationship between the annual average PM_{10} concentrations and the number of days daily average concentrations exceed the standard of 50 µg/m^3 for data from all PM_{10} monitoring stations between 1992 and 2001 is presented in Figure 2.15. The dotted line shows the upper and lower 95% confidence limit respectively. The resulting relationship is $y = 5{,}2\ x - 127$, with a correlation coefficient of 0.91. It can be concluded that although the correlation between the EU annual and daily standards is high, the EU daily standard is not equivalent with the EU annual standard.

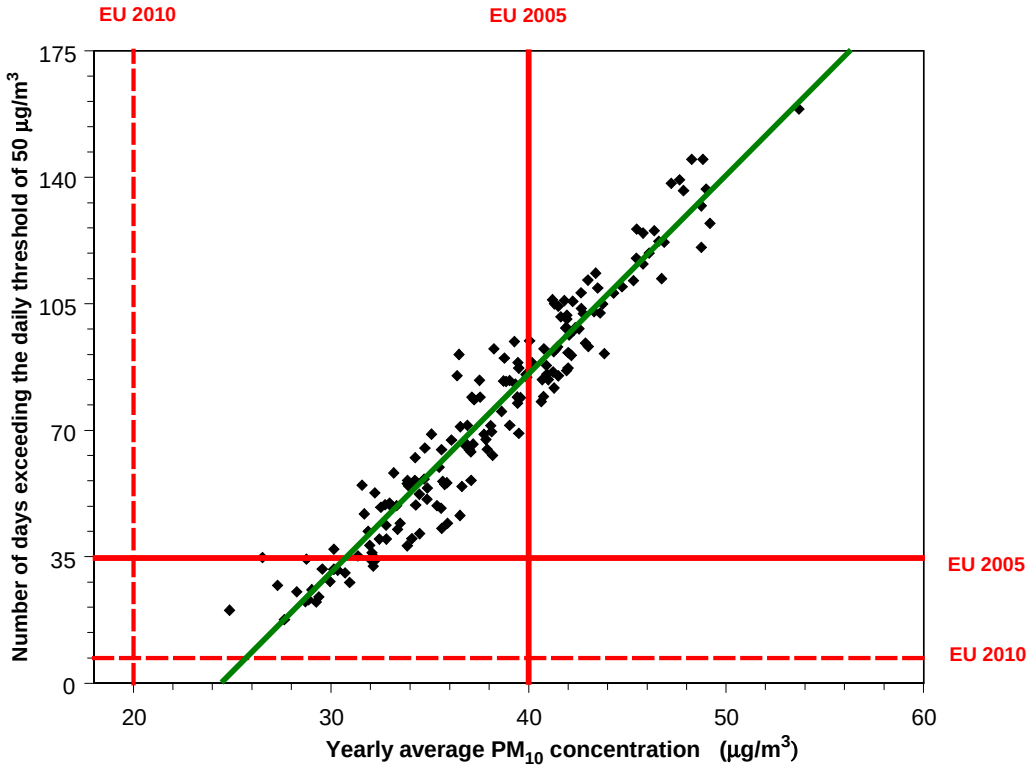


Figure 2.15 The annual average PM_{10} concentration level and the number of days the EU standard of 50 µg/m^3 was exceeded for all the different PM_{10} monitoring stations between 1992 and 2001.

The horizontal and vertical red lines in Figure 2.15 show the EU daily standard for 2005 (35 days permitted on which 50 µg/m^3 is exceeded) and the annual limit value (40 µg/m^3). The red dotted lines represent indicative EU standards for 2010.

2.3.2.3. Weekly variations

Average PM_{10} concentrations are lower during weekends than on workdays, as can be seen in Figure 2.16. Systematic differences between workdays and weekends are caused by anthropogenic emissions. Street stations monitor the largest differences (3.6 µg/m^3) between Monday–Friday workday averages and weekend averages; urban stations monitor on average differences of 3.4 µg/m^3, while regional stations monitor the smallest differences (1.9 µg/m^3); see Table 2.6. The highest PM levels are monitored on a Wednesday for regional and street stations, and on a Friday for urban stations. The lowest levels are monitored on a Sunday for all three types of station. For more information see Van der Wal and Janssen (1996).

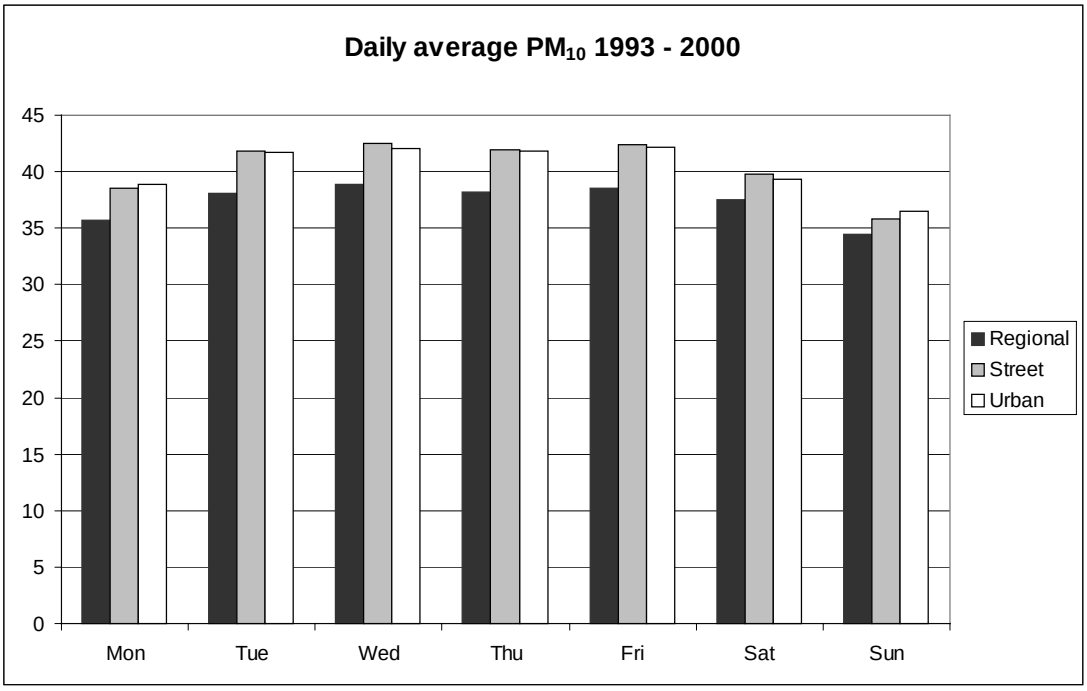


Figure 2.16 Average weekday PM_{10} concentration in µg/m^3 from 1993 to 2000 for the three types of station.

Table 2.6 Average levels of PM_{10} in µg/m^3 for weekdays and weekends (1993–2000). The standard deviations in the concentration levels of the different monitoring stations are given between brackets.

	Monday–Friday		Saturday–Sunday	
	Average	stdev.	Average	stdev.
Regional	37.9	(± 3.7)	36.0	(± 3.7)
Street	41.4	(± 1.8)	37.8	(± 2.4)
Urban	41.4	(± 2.5)	37.9	(± 2.4)

2.3.2.4. Monthly variations

Monthly averages at the rural background site Vredepeel #131 were analysed. Figure 2.17 shows the monthly averaged concentrations between 1996 and 2001. The annual variation in the monthly averages is large, typically with higher concentrations in January, February and March. The numbers of days on which a 24-hour average of 50 µg/m3 is exceeded in the different months of the year can be found in Figure 2.18.

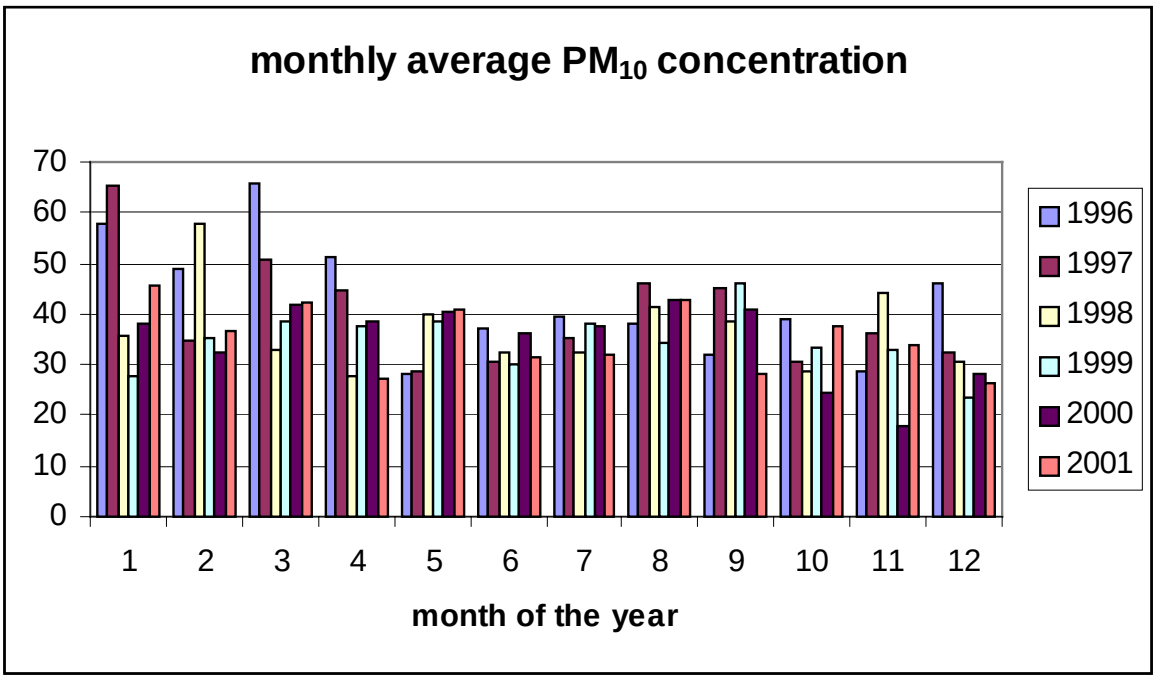


Figure 2.17 Monthly average concentration at the regional monitoring site Vredepeel #131 between 1996 and 2001.

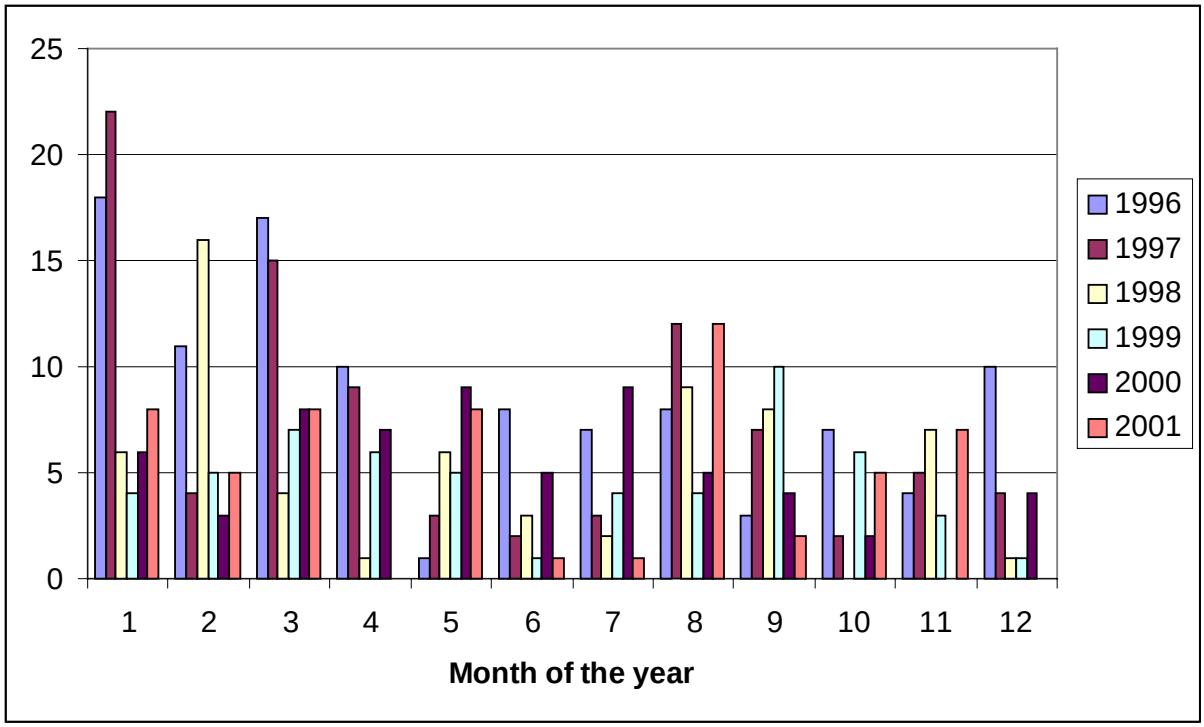


Figure 2.18 Number of days per month when daily average of 50 $\mu g/m^3$ is exceeded at Vreedepeel #131 between 1996 and 2001.

2.3.2.5. Seasonal variations

The average PM_{10} concentration for summer (June, July and August) and winter (December, January and February) was calculated for the years 1994 to 1999. As an example, the season-specific average PM_{10} levels are presented in Figure 2.19. The three types of station (**R**egional, **U**rban and **S**treet) are shown on the x-axis.

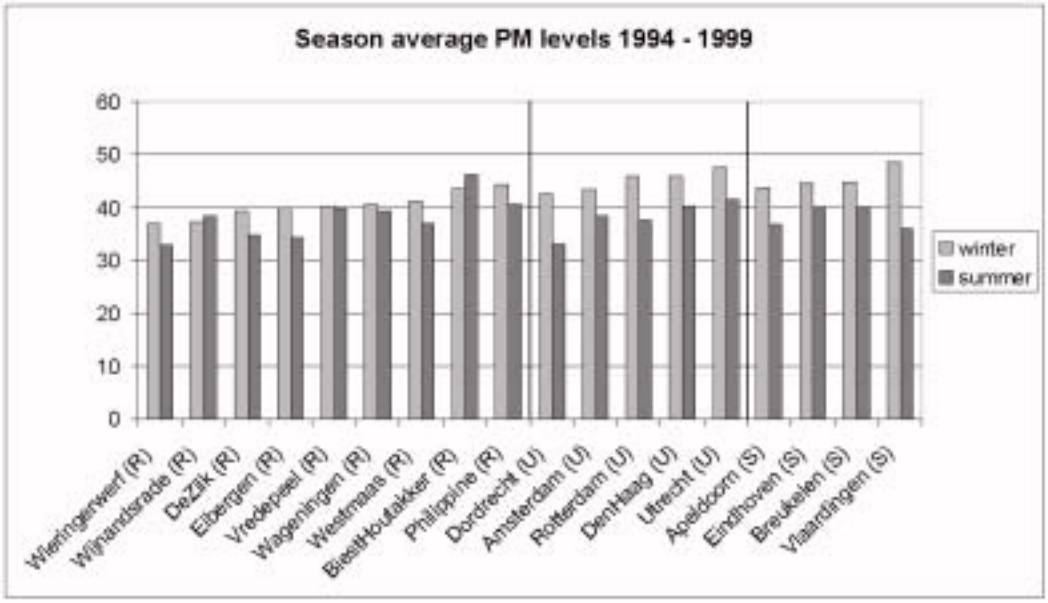


*Figure 2.19 Season-specific average PM_{10} levels between 1994 and 1999 from all the PM_{10} monitoring stations. The type of station is indicated between brackets (**R**egional, **U**rban and **S**treet).*

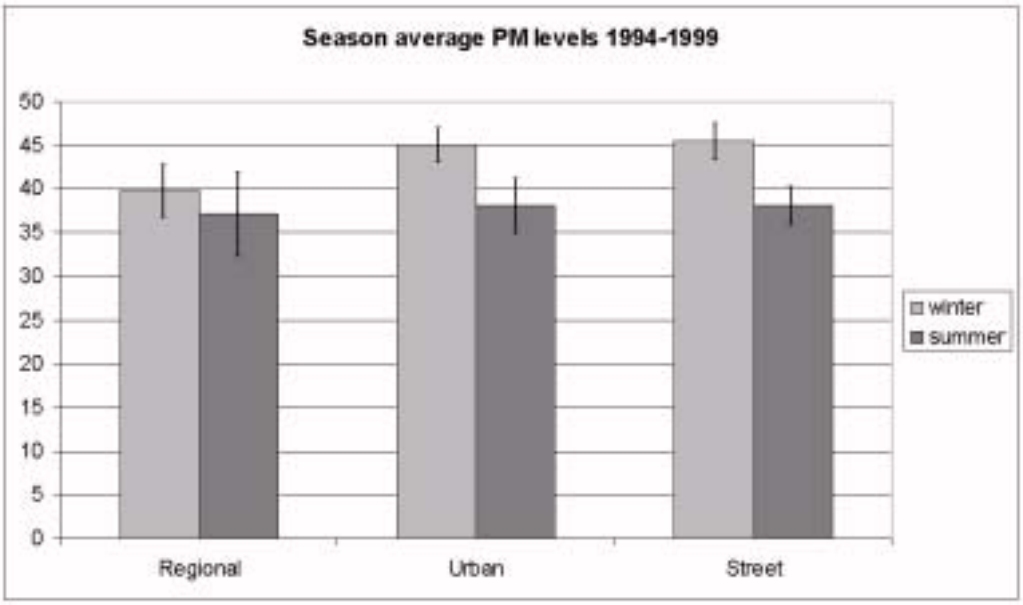


Figure 2.20 Season average PM_{10} levels between 1994 and 1999 from all the PM_{10} monitoring stations, categorised by type of station.

The differences in average PM_{10} concentration between summer and winter can be attributed to less atmospheric dispersion in the winter due to lower mixing heights and

a more stable atmosphere. What is striking in Figure 2.19 is that average seasonal differences at regional stations appear to be less pronounced than in cities, though the same weather systems pass over both types of station.

Table 2.7 gives the measured Secondary Inorganic Aerosol (SIA) as the sum of concentrations of nitrate, sulphate and ammonium measured daily using an LVS at regional NAQMN stations between 1994 and 1999. The SIA concentrations in the summer are 87% of the winter concentrations.

Table 2.7 Measured average concentrations of SIA (nitrate, sulphate and ammonium) in μg/m^3 between 1994 and 1999 (average of Witteveen #928, De Zilk #444, Vredepeel #131 and Wieringerwerf #538).

	Winter	Summer
Nitrate	4.1	3.5
Sulphate	3.6	3.3
Ammonium	2.4	2.0
SIA total	10.1	8.8

Sea salt concentrations were determined using a Dichote with Na^+ or Cl^- concentrations as tracer (Visser *et al.*, 2001). With Na^+ as tracer, an average concentration of 3.8 μg/m^3 was estimated in the summer and 11.6 μg/m^3 in the winter at the coastal monitoring station De Zilk (# 444), resulting in a sea salt summer/ winter ratio of 32%. With Cl^- as tracer, an average concentration of 3.1 μg/m^3 was measured in the summer and 5.8 μg/m^3 in the winter, resulting in a summer/winter ratio of 54%.

The relatively higher PM_{10} concentrations in summer can be observed most clearly at the regional stations (cf. Figure 2.19), in their most extreme form at stations with more farming activities such as Vredepeel #131 and BiestHoutakker #230. This indicates that seasonal variations in emissions could also be due to a combination of natural and farming activities. In size, the PM could very well be quite coarse material, possibly even with diameters of more than 10 μm. This material has a short atmospheric residence time and a transport distance of only 35 to 110 km (see Table 2.14), so it is more of a local phenomenon. It has to be remembered that the cut-off curve of the TEOM PM_{10} monitors is designed not to be very steep, but follows the cut-off of the human airways, implying that by definition a specific percentage of the particles of 15 μm even is considered to be part of PM_{10}.

The phenomenon of seasonal variation in natural and anthropogenic sources not listed in the emission inventories has been substantiated by Visser *et al.* (2001). In their report they showed (Table 15B) that the average contribution of crustal material to PM_{10} is approximately 50–100% higher during summer than during winter in the Netherlands, despite the fact that general dispersion is higher during summer. In summer, the contribution of crustal material in a rural area like Vredepeel #131 is estimated to be 2–3 μg/m^3 higher than in winter.

In the report from Visser *et al.* (2001), organic carbon (OC) was only analysed in PM with diameters smaller than 2.5 μm. From measurements in Germany it is known that approx. 25% of OC is in the coarse size category, with diameters between 2.5 and 10 μm (Israel *et al.*, 1992). In Vredepeel #131, Visser *et al.* (2001) established approx. 4 $\mu g/m^3$ OC with a diameter smaller than 2.5 μm, implying a 'missed' mass of approx. 1 $\mu g/m^3$ OC > 2.5 μm if the German results are applied to the situation in the Netherlands. This would produce an annual average value of 5 $\mu g/m^3$ OC in a rural area like Vredepeel. In Vredepeel #131 the OC levels are 44 % higher during summer than in winter, corresponding in absolute terms with an extra contribution of 2–3 $\mu g/m^3$ in the summer, again despite the generally higher dispersion during summer.

Seasonal variation in emissions is also observed for Secondary Organic Aerosol (SOA) formation. Absolute values cannot be presented for the Netherlands. Even and Ten Brink (2000) made a very rough estimate of seasonal SOA aerosol levels, arriving at a percentage of the PM_{10} of 3 to 15% in urban areas and 3 to 30% in rural areas. For SOA a seasonal variation is assumed. Relative to total PM_{10} this variation in SOA is minor. In episodes of photochemical smog, the contribution may be appreciably larger than the annual average values (Even and Ten Brink, 2000).

An accurate estimate of the seasonal variation in the chemical composition of PM_{10} is important enough to have it better substantiated. This will require a more specific measurement programme. When this is combined with careful analysis of the chemical composition and size distribution of PM_{10}, in particular the EC and OC part, the tentative results presented above may be rendered less uncertain and, it is hoped, will be replaced by more factual information.

2.4. Natural sources and global background

Background concentrations commonly refer to the concentrations observed in remote areas that are relatively unaffected by local pollution sources. However, for regulatory purposes it refers to an estimate of the contributions from natural and other sources that cannot be controlled by abatement measures. The background concentration is therefore defined here as the sum of the contributions resulting from anthropogenic sources outside the EU and the contribution of all natural sources. A quantification of this fraction is relevant as it cannot be influenced by EU regulations.

2.4.1. Northern hemisphere background

Background concentrations commonly refer to the concentrations observed in remote areas relatively unaffected by local pollution sources. Background concentrations from the northern hemisphere are defined here as the sum of the contributions resulting from anthropogenic and natural emissions in the northern hemisphere but outside the area under study (an area of 2,000 x 2,000 km^2 in Europe, including the Netherlands) and the contribution of natural sources within this area. A precise estimation of the natural and non-natural background level determines the fraction of ambient PM that can potentially be influenced by regulations.

Contribution from the northern hemisphere to PM_{10} levels in western Europe is the result of emissions in North America, Asia and Africa. The natural contribution

results from sea salt. Fugitive dust (crustal material) is an only partially controllable source, consisting of particles that are of natural and anthropogenic origin. The contribution to PM in the Netherlands is estimated as follows:

- *North America and the Atlantic Ocean:* The most likely annual average contribution to the PM_{10} concentrations arising from emissions in North America is estimated to be in the order of 0.7 $\mu g/m^3$, with a bandwidth of 0.6–0.9 $\mu g/m^3$ (Weijers *et al.*, 2000).
- *Eurasia:* Hardly any reliable data are available on PM_{10} emissions or concentration levels near the eastern border of the EMEP region or further to the east. The maximum contribution of Eurasian sulphur emissions to Dutch PM_{10} levels is 1.3% (Weijers *et al.*, 2000).
- *Africa:* In suitable meteorological conditions, Saharan dust can also be transported across the Mediterranean, especially into southern European countries, or it can end up east of Central America. The supply of Saharan dust to Europe is of a highly episodic nature. For this reason, its contribution to the annual average concentration of PM_{10} is relatively small: less than 1 $\mu g/m^3$ for the Netherlands (Weijers *et al.*, 2000).

2.4.2. Wind-blown crustal material

The contribution of wind-blown dust to PM_{10} concentrations can be assessed by means of Artificial Neural Networks (ANNs) as described by Vrins and Schulze (2000). Using ANN, it is possible to distinguish a slowly varying background concentration from high frequency variations caused by local sources and/or changes in weather conditions. In general, contributions due to high frequency variations show a characteristic relationship with wind speed, being high at a low wind speed and low with increasing wind speed. However, above a certain threshold wind speed this contribution increases again as a result of the wind-blown crustal material.

ANNs have been established for six monitoring stations of the RIVM National Air Quality Monitoring Network. At these stations, PM_{10} concentrations were measured using a FAG (β-dust monitor). These concentrations were corrected for loss of volatile particles using a factor of 1.3. The variations in PM_{10} concentrations measured with a FAG at the various stations could be explained for 48 to 58% by ANN. At the regional monitoring sites of the NAQMN in Westmaas (#437), De Zilk (#444) and Witteveen (#928), a characteristic profile was found in relation to wind direction and wind speed. For the location of these sites see Figure 2.3. At low wind speeds, the high frequency variations were highest with a continental wind direction (i.e. southeasterly). At high wind speeds, a contribution of 2.6 $\mu g/m^3$ made by wind-blown dust was calculated for all three stations. At the street station in Vlaardingen (#433), located near a busy road, the contribution made by wind-blown dust was found to be 2.5 $\mu g/m^3$, whereas in the urban background site Rotterdam (#418) it was found to be 1.5 $\mu g/m^3$. The relative standard deviation of the calculations was estimated at 30%. As the ANNs are not able to speciate the PM levels chemically, all differences were attributed to crustal material. Of course, this is an overestimation, especially in the coastal area of the Netherlands, as sea salt is also part of the PM levels measured by a FAG. With higher wind speeds the sea salt contribution is also likely to increase.

It should be borne in mind that the average correction factor of 1.3 used for the FAG makes it difficult for the ANNs in this analysis to detect crustal material based on wind frequency variations. The factor of 1.3 is based on the average losses occurring with the higher temperatures in the FAG for ammonium nitrate and OC especially, but not for crustal material. By definition, this analysis will therefore result in an overestimation of the crustal contribution using the ANN.

Based on the fact that TEOMs have a shorter response time than the FAG and that TEOM ANN analyses conducted by Vrins *et al.* (2000) resulted in slightly lower estimates of the crustal contributions, plus the fact that Visser *et al.* (2001) estimated the wind-blown crustal material from non-anthropogenic sources at 2 $\mu g/m^3$, the overall mean value is estimated at 2 $\mu g/m^3$.

Using the OPS dispersion model, it was calculated that an emission of wind-blown dust of 1 ktonne/year, evenly distributed over the Netherlands, results in a contribution of 0.05 to 0.1 $\mu g/m^3$ to the annual average PM_{10} concentration. Consequently, based on a mean contribution of 2 $\mu g/m^3$, 20–40 ktonnes/year is the input of crustal material needed for long-term modelling to estimate an annual average regional increase of 2 $\mu g/m^3$ in the Netherlands.

2.4.3. Sea salt

The contribution from sea spray to the ambient levels of PM_{10} is substantial in the Netherlands. On a day-to-day basis this contribution can be considerably greater than the annual average values presented by Visser *et al.* (2001). Using sodium and chloride as tracers for sea salt, these authors estimated that the average sea salt contribution in the Netherlands was between 4–7 $\mu g/m^3$ on an annual basis. Slightly more than 50% of this sea salt was in the coarse fraction. Compared with the estimates of sea salt contribution in the UK, the annual average values presented in the Netherlands seem to be quite large, but this may be a consequence of our geographical position and the flat nature of the country, with virtually no hills. A discussion with literature references is presented in Visser *et al.* (2001).

Based on the preceding three sections, the resulting natural and global background contribution to the Dutch annual average PM_{10} levels is estimated to be in the order of 7–10 $\mu g/m^3$.

2.5. Modelling

2.5.1. Primary emissions of PM

2.5.1.1. The Netherlands

This subsection gives a brief overview of anthropogenic sources of PM in the Netherlands. Emission estimates for the size ranges PM_{10} and $PM_{2.5}$ are presented. The main focus in this chapter will be on the PM_{10} size fraction. Presented estimates for $PM_{2.5}$ are indicative. Natural sources are not treated here (see the previous section for more details), since they are not included in Dutch emission inventories.

Tables 2.8 and 2.11 show national emission estimates for PM_{10} and $PM_{2.5}$ for the period 1980–1998. The different sources have been grouped into seven larger socio-economic sectors commonly used in Dutch policymaking. Figure 2.21 is a pie chart illustrating the contribution of the socio-economic sectors. Almost all major anthropogenic sources in the Netherlands have been considered, with two exceptions: re-suspended dust due to turbulence induced by the motion of vehicles along roads and wind erosion in agricultural areas. The re-suspended dust caused by vehicles may originally come from transport (tailpipe and wear of tyres, brakes and road surface) or from other sources (crustal material and deposited dusts from other source categories).

First, some general remarks will be made concerning the Dutch PM_{10} and $PM_{2.5}$ emissions inventories. Next, a brief outline is presented of the main source categories of PM_{10} emissions in the Netherlands for the year 1998. After that, the contribution made by various sectors to $PM_{2.5}$ emissions will be discussed, with a focus on the dissimilarities that can be observed in contrast to PM_{10}. Finally, some remarks are made on the uncertainty of reported emission estimates.

a. Dutch PM inventory

For this study, emissions of PM_{10} were taken from the 2001 version of the Dutch Pollution Emission Register (PER). The PER is used as the national instrument for monitoring emissions. PM_{10} has been included in the PER for many years now. Apart from PM_{10}, the PER also includes emission estimates for TSP, which are not presented here. It does not yet include $PM_{2.5}$ emissions.

Emission estimates included in the PER are reported for each year in the 2001 Dutch emission inventory report (CCDM, 2001) and the 2001 Dutch Environmental Balance report (RIVM, 2001a). The methods used for estimating PM_{10} emissions, including all underlying activity levels, emission factors and references, are not documented here. However, more details regarding these data will be presented in a background report to this report (Smeets *et al.*, 2002).

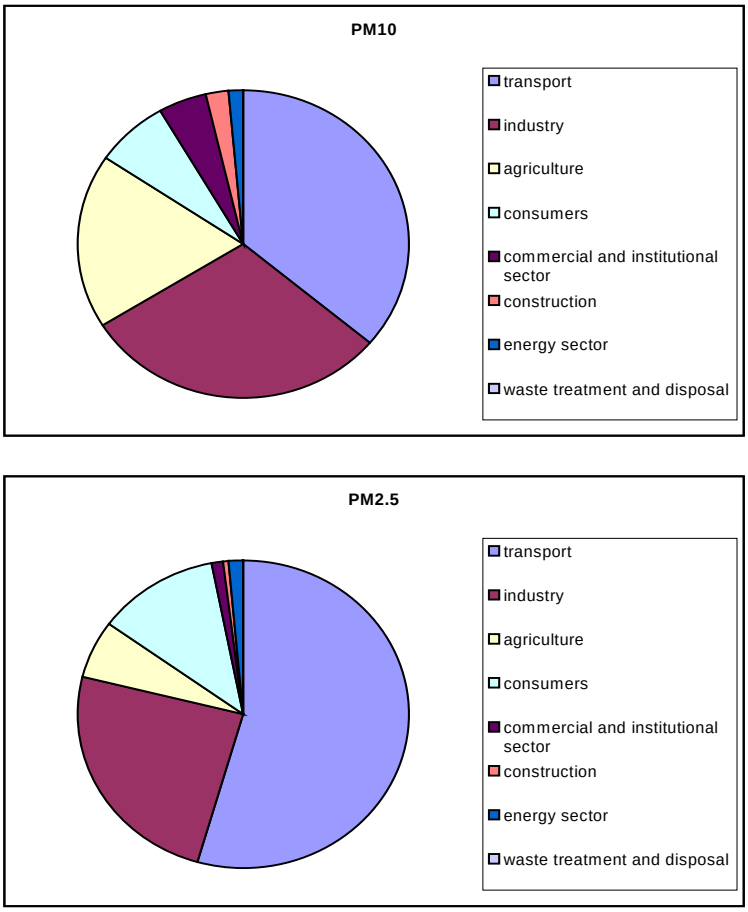


Figure 2.21 PM_{10} emissions and $PM_{2.5}$ emissions in 1998 by source sector.

The 2001 version of the PER did not include emission estimates for agriculture and diffuse industrial sources. Because it is the explicit intention of the current report to present a complete PM_{10} emissions inventory based on the latest knowledge, the PER 2001 estimates have been supplemented by emission estimates for these two missing sources. These estimates were prepared by RIVM (Smeets *et al.*, 2002) based on the results of Dutch research projects (Chardon, 1999; Groot Koerkamp, 1996; Haskoning, 2000). These new estimates will be included in future versions of the PER.

It should be borne in mind that the Fifth National Environmental Outlook (RIVM, 2000b) published in 2000 was based on an outdated PM_{10} data set. National emissions in 1998 were estimated in this old data at 41 ktonnes, i.e. about 25% lower than the current estimate of 54 ktonnes. The old data for PM_{10} is summarised in Table 2.9.

Emission estimates shown in Table 2.8 and Figure 2.21 include some of the international shipping emissions. Maritime shipping emissions in Dutch ports and inland shipping in the Netherlands are included. Emissions presented throughout this report are consistent with January 2002 reporting to EMEP.

As already mentioned, PER estimates for $PM_{2.5}$ are not yet available. RIVM estimated these emissions (Table 2.11) from PM_{10} emission estimates using the source-specific $PM_{2.5}$ size fractions presented in Table 2.12. For transport, the profiles are based on

Van den Brink (1996). For industry, size fractions are based on a detailed Dutch study conducted by the Netherlands Organisation for Applied Scientific Research TNO (TNO, 2002). The reported size profiles in Table 2.12 are indicative.

b. Primary PM_{10} emissions

National emissions of PM_{10} in 1998 were approximately 54 ktonnes. Since 1980, PM_{10} emissions have fallen by approximately 55% due to effective national as well as EU policies. From1990, the decrease in emissions was approximately 35%. At present, major sources of PM_{10} are transport, industry, agriculture, consumers and the commercial and institutional sector. Emissions sources within these sectors are covered in detail in the following paragraphs.

Transport

Emissions of PM_{10} for the transport sector in 1998 are estimated at 19 ktonnes, i.e. 36% of total national emissions. The main contributors to traffic emissions were the exhaust from diesel-powered road vehicles, off-road mobile machines and ships; but petrol-driven road vehicles also emit small amounts of PM_{10}. In 1998, diesel vehicles accounted for about 80% of the calculated transport emissions; petrol-fuelled engines contributed about 5%.

PM in exhaust gas has an average aerodynamic diameter of between 0.1 and 1.0 µm. Non-exhaust emissions from tyres, brakes and road wear are also part of transport-related PM. In 1998, approximately 15% of transport-related PM_{10} emissions were explained by wear. These particles are in general larger than 2.5 µm in diameter. PM emissions in the Netherlands from air and rail transport are negligible.

Shipping and off-road machinery are major sources of PM_{10}. In 1998, about 7 ktonnes of the national total of 54 ktonnes came from these non-road sources. This is about 35% of the total of transport emissions of 19 ktonnes: 19% of the transport total comes from off-road machines, 9% from inland shipping and 9% from sea shipping in Dutch ports. These nautical sources are important for total PM_{10} emissions because engine design is less advanced and the level of emission control is poor compared with road vehicles. Also, the shipping sector uses fuels with a maximum sulphur content of 0.2% for gas oil and 5% for heavy fuel oil. A high sulphur content in the fuel results in high emissions of sulphate and particulate matter. For road vehicles, current requirements for the maximum sulphur content of diesel oil is 0.035%, resulting in considerably reduced emissions

Table 2.8 PM_{10} emissions in the period 1980–1998 (latest knowledge on agricultural emissions and ventilation emissions from industrial buildings is included).

	1980	1985	1990	1995	1998	1999	2000	*share in 1998 emissions*
million kg								
transport	**32,6**	**28,7**	**26,6**	**21,2**	**19,3**	**18,9**	**18,9**	***36%***
combustion	*30,6*	*26,6*	*24,1*	*18,4*	*16,3*	*15,8*	*15,6*	*31%*
process	*1,9*	*2,1*	*2,5*	*2,8*	*3,0*	*3,2*	*3,3*	*6%*
industry	**52,8**	**36,5**	**35,0**	**21,3**	**16,1**	**n.e.**	**n.e.**	***30%***
combustion			*7,4*	*6,0*	*3,7*			*7%*
process, other			*24,8*	*12,6*	*9,6*			*18%*
process, building venting			*2,7*	*2,7*	*2,7*			*5%*
consumers	**4,5**	**4,5**	**4,4**	**4,0**	**3,9**	**3,7**	**3,6**	***7%***
combustion	*2,7*	*2,6*	*2,6*	*2,2*	*2,2*	*2,0*	*1,9*	*4%*
process	*1,8*	*1,8*	*1,8*	*1,7*	*1,7*	*1,7*	*1,7*	*3%*
agriculture	**7,7**	**8,5**	**8,8**	**9,5**	**9,7**	**10,0**	**9,8**	***18%***
combustion	*0,2*	*0,2*	*0,2*	*0,2*	*0,1*	*0,1*	*0,1*	*0%*
process	*7,5*	*8,3*	*8,6*	*9,4*	*9,7*	*9,9*	*9,7*	*18%*
commercial and institutional sector	**2,0**	**2,0**	**3,0**	**2,8**	**2,5**	**n.e.**	**n.e.**	***5%***
combustion			*0,5*	*0,5*	*0,1*			*0%*
process			*2,5*	*2,3*	*2,5*			*5%*
construction	**0,8**	**0,8**	**1,1**	**1,1**	**1,2**	**n.e.**	**n.e.**	***2%***
combustion			0,1	0,1	0,1			*0%*
process			1,0	1,0	1,1			*2%*
waste treatment and disposal	**4,3**	**1,2**	**0,9**	**0,1**	**0,1**	**n.e.**	**n.e.**	***0%***
combustion			*0,0*	*0,0*	*0,0*			*0%*
process			*0,9*	*0,1*	*0,0*			*0%*
energy sector	**11,0**	**1,7**	**1,6**	**0,7**	**0,6**	**n.e.**	**n.e.**	***1%***
combustion			*1,5*	*0,4*	*0,4*			*1%*
process			*0,1*	*0,3*	*0,2*			*0%*
TOTAL	**115,7**	**83,9**	**81,4**	**60,7**	**53,4**	**n.e.**	**n.e.**	***100%***
combustion			*36,4*	*27,8*	*22,8*			*43%*
process			*44,9*	*32,9*	*30,6*			*57%*

Part of international shipping is included, i.e. international maritime transport in Dutch ports and international shipping on Dutch inland waterways.

Table 2.9 Outdated PM_{10} emissions in the period 1980–2030 for the EC and GC scenario. Old data according to Fifth Environmental Outlook (reported in CCDM, 2000; RIVM, 2000a; RIVM, 2000b).

	1980	1985	1990	1995	1998	2010		2020		2030	
						EC	GC	EC	GC	EC	GC
million kg											
transport	25,7	23,0	22,3	19,8	18,1	9,9	10,2	9,8	10,6	11,0	12,2
industry (incl. refineries)	50,1	33,8	32,7	19,1	13,3	9,9	10,3	10,0	11,9	10,2	12,5
consumers	4,4	6,7	7,5	6,7	6,0	6,5	6,5	6,6	6,5	6,7	6,5
agriculture	0,1	0,1	0,1	0,1	0,1						
storage and handling	2,0	2,0	2,6	2,4	2,5						
construction	0,1	0,1	0,1	0,0	0,1						
public waste incineration	4,3	1,2	0,9	0,1	0,1						
energy sector	11,0	1,7	1,5	0,6	0,6	0,4	0,6	0,3	0,3	0,3	0,3
other[1]						0,7	0,7	0,7	0,7	0,2	0,2
TOTAL	97,7	68,7	67,8	48,7	40,7	27,5	28,1	27,4	29,9	28,4	31,6

As mentioned earlier, emissions due to re-suspension are not incorporated in the inventory. It should be mentioned here that results from PM_{10} emission inventories in Switzerland (BUWAL, 2001) and Germany indicate that re-suspension makes an important contribution to transport-related PM_{10}. These predominantly crustal re-suspension contributions have been assessed by Visser *et al.* (2001) and are of a similar magnitude to the primary PM contribution made by traffic.

Industry

The emission of PM_{10} from industry is estimated at 16 ktonnes, i.e. 30% of total national emissions. The most important source categories are refineries (3.4 ktonnes), basic metals production (4.6 ktonnes), the production of food and animal feed (2.4 ktonnes), the production of fertilisers and other chemical products (2.1 ktonnes) and the production of cement and other building materials (1.8 ktonnes).

Approximately 75% of industrial emissions are from non-combustion process sources, including diffuse emissions from the ventilation of buildings. Combustion emissions in industry are low because natural gas is the main fossil fuel used in the Netherlands. The only combustion source in industry with a significant amount of PM_{10} emissions is refineries. Combustion emissions from oil refineries were 3.7 ktonnes in 1998 compared with 0.2 ktonnes from process sources in refineries. For other industrial sectors, combustion emissions amount to only 0.3 ktonnes, compared with 12.6 ktonnes from process sources.

Process-related emissions from industry cover a wide range of different activities all with their own specific characteristics. Emissions registered in the PER are based on estimates of hundreds of different individual emission points. These estimates are based on a number of different sources: annual environmental reporting by industrial companies, estimates by TNO or estimates from licensing authorities. In addition, emissions from non-individually registered enterprises are estimated collectively using a top-down approach based on statistics and emission factors.

The PER 2000 incorporates the estimates for pollutant releases from waste gas flows through pipes and ducts. There is only scarce registration of diffuse industrial emissions. Two important diffuse sources should be considered, i.e. PM emitted due to the mechanical or natural ventilation of industrial buildings and PM emitted due to storage and handling activities in the open air. In 2000, a study was performed to estimate the emissions from the ventilation of industrial buildings based on data from workplace exposures and on ventilation flows. These diffuse emissions for PM_{10} were estimated at 3.7 ktonnes, which amounts to about 30% of total industrial emissions in 1998. This emission figure has been included in the new emissions table (Table 2.8). No new estimates were made for storage and handling activities in industry. It should be remembered that the estimates presented are still incomplete and are part of a permanent process of revision and improvement.

PM emissions from point sources, including plant ventilation, are strictly controlled in the Netherlands through permits. Table 2.10 presents an overview of current emission control limits contained in the Netherlands emission Regulations (NeR) and the Order governing combustion plant emission requirements (BEES). The NeR applies to process emissions in ambient air and is based on Best Available Technology (BAT). It is implemented by the competent licensing authorities. The NeR incorporates

guidelines for all process units, including those with low mass flows. Thanks to the current Dutch system of emission permits, actual emissions of TSP for many industrial processes in the Netherlands are as low as 5 mg/m^3 (Tauw, 1996). BEES is applicable to new large-scale combustion plants. Under BEES regulations, dust concentrations depend on the fuel used.

The national guidelines NeR and BEES have now been almost fully implemented. Allowing for the NeR boundary condition on costs, this means that all industrial measures costing less than 2.3 euro/kg of avoided total particulate matter have been taken.

In the Netherlands, technology for the abatement of PM emissions from industrial process sources is probably situated somewhere at the top end of the EU control spectrum. This can be concluded from a comparative review for 1994 of national emission guideline standards used as a basis for setting the emission limits contained in the official facility permits (ERM, 1996). In 1994, emission standards in the Netherlands were more stringent than in Belgium, Germany, France, Italy, Luxembourg and the United Kingdom. However, to what extent national emission standards in different countries have converged since 1994 is currently unknown.

On a European scale, Reference Documents on Best Available Techniques (BREFs) are under development by the European Integrated Pollution Prevention and Control (IPPC) Bureau. These BREF documents reflect the information exchange conducted under Article 16 (2) of Council Directive 96/61/EC. They include subsections on applied processes and techniques, present emission and consumption levels, and techniques to consider in the determination of Best Available Technologies (BAT).

In Table 2.10 we have compared the guidelines given in the national NeR/BEES with the information in BREF documents for new installations for the branches of industry for which BREF documents are at the final stage. Note that these documents are not available for the majority of sectors.

As can be seen from Table 2.10, most concentration levels according to NeR/BEES are stricter than in the BREF. Only for the sinter strand in the iron and steel industry is the concentration in the BREF lower, but NeR is expected to be lowered shortly to the BREF level because emission concentrations in the Netherlands are already below this level. The concentration level in the non-ferrous industry is also lower in the BREF.

For all other sectors where BREF documents are available, the plants that comply with the NeR and BEES will be operating in compliance with the BREF guidelines.

Consumers

Consumers are estimated to emit 4 ktonnes $PM_{10,}$ i.e. 7% of the national total. Domestic burning of wood in stoves and fireplaces (1.9 ktonnes) is the major source, contributing 50% to consumer emissions. Besides refineries, it is the only significant stationary combustion source in the Netherlands. Other consumer emissions are mainly caused by cigarette smoking (1.6 ktonnes). Domestic emissions due to the combustion of fossil fuels are negligible in the Netherlands because natural gas is used for domestic heating.

Agriculture
Agricultural activities are estimated to emit approximately 9.7 ktonnes $PM_{10,}$ i.e. 18% of total national PM_{10} emissions. The major contribution originates from animal housing systems (9.3 ktonnes), of which 5.3 ktonnes is from poultry, 3.1 ktonnes from pigs and 0.9 ktonnes from cattle (Smeets *et al.*, 2002). The magnitude of emissions of other agricultural sources have also been estimated (Chardon, 1999; Smeets *et al.*, 2002). This study included emission estimates for the delivery of animal feed concentrates, the application of fertilisers and pesticides, and the harvesting of arable crops and hay. At 0.4 ktonnes, the size of emissions from all these sources together is far lower than the emissions from animal housing systems.

Emission estimates for agricultural activities presented in this study have not yet been included in the Dutch Environmental Balance report 2001or in the PER 2001.

As mentioned earlier, the re-suspension of crustal material by wind in agricultural areas has been excluded from this chapter, because wind-blown crustal material in the Netherlands was treated in subsection 2.4.2. These emissions can be a major source, especially in dry periods and, more particularly, in winters characterised by high wind speeds.

Commercial and institutional sector
About 2.5 ktonnes of PM_{10} emissions are estimated to come from the commercial and institutional sector (Dutch: 'HDO'), i.e. 5% of the national total. Companies specialising in the storage and handling of dusty materials are the principal emitters of this PM. Only 0.1 ktonne is emitted through the combustion of fossil fuels in this sector. Emissions from storage and handling have been estimated using the available measurement data from the port authorities of the Rijnmond environmental protection agency (DCMR) and individual companies (measurements from several years) (Vrins, 1999).

The emission estimates presented for storage and transshipment do not include similar activities that take place in industry. The industrial emissions are included under the category industry.

Construction
About 1.2 ktonnes of PM_{10} is emitted as a result of construction activities (Dutch: 'bouwsector'), i.e. 2% of total national emissions. Estimates are based on a Dutch study of diffuse emissions (Haskoning, 2000).

Energy sector
The energy sector is estimated to emit about 0.6 ktonne PM_{10}, i.e. 1% of total national emissions. Coal-fired power stations in the Netherlands have been equipped with highly efficient electrostatic precipitators as well as flue gas desulphurisation units, which further reduce PM emissions. PM_{10} emission concentrations of less than 3 mg/m^3 are measured in the flue gas. The other fossil fuel used in the energy sector is clean natural gas. The use of fuel oil by the energy sector in the Netherlands is negligible.

Table 2.10 Comparison of concentration levels between national NeR and BEES requirements and EU BREF guidelines (new installations).

Industry	Process	Reduction Technique *	Dust concentration / emission		
			NeR [mg/m3]	BREF [mg/m3]	[kg/ton pr.]
Cement	Cement production	FF (ESP)	10	10–50	
	. Kiln systems	FF (ESP)	10	10–50	
	. Klinker cooler	FF (ESP)	10	10–50	
	. Cement mills	FF (ESP)	10	10–50	
Iron and Steel	Sinter strand	Advanced ESP	100	50	
	. idem	ESP + HES	100	50	
	. idem	ESP + lime + FF	10	10–20	
	Pelletisation				< 0.1
	. Grinding mills	ESP	10	< 50	
	. Drying / induration	FF (wet scrubber)	10	< 20	
	Coke oven plant				
	. Charging / pushing	Collection + FF	10	< 30	< 0.01
	. Quenching	Wet scrubber	< 0.06 kg/t		< 0.05
	Blast furnace				
	. Furnace gases	HES (wet ESP)	25	< 10	
	. Cast house	Collection + FF (ESP)	rend. > 99%	1–15	
	Basic Oxygen Steel	Collection + FF (ESP)	10	10–15/20–30	
	. Fugitive		< 5 g/t		5–15 g/t
	Electric steel making	Collection + FF	10	5–15	
Glass	Melting	FF or ESP	10 / 25	< 10–20	< 0.1
	. Container / Flat glass	FF or ESP	10 / 25	5–30	< 0.1
	. Fibre glass	FF or ESP	10 / 25	5–30	< 0.14
	. Glass wool	FF or ESP	10 / 25	5–30	< 0.1
Ferrous Metals	Hot rolling				
	. Dry dust	Enclosure + FF	10	< 5–20	
	. Wet fumes	Enclosure + ESP	25	< 10–50	
	Cold rolling				
	. Decoiling / levelling / welding	Collection + FF	10	< 5–20	
Non-Ferrous	Electrode baking	Dry alumina + FF	10	1–5	
	Other sources	Collection + FF	10	1–5	
Lime	Calcining of lime	FF / ESP	10 / 25	< 5–20	0.1–0.2
	Hydrating	FF / LES / HES	10 / 25	< 5–20	0.02
	Grinding and milling	FF	10		0.03

* Explanation:
- FF = fabric filter
- ESP = electrostatic precipitator
- LES = low energy scrubber
- HES = high energy scrubber

Public waste treatment and disposal

Emissions from public waste incinerators are negligible because of strict controls.

c. Primary $PM_{2.5}$ emissions

Indicative emissions for $PM_{2.5}$ in 1998 were estimated at 32 ktonnes; see Table 2.11. The uncertainties in the emissions for $PM_{2.5}$ are considerably higher than for PM_{10}.

With regard to the review of the current EU directive, which besides PM_{10} will also look at other indicators such as $PM_{2.5}$, it is important to know to what extent sources for PM_{10} and $PM_{2.5}$ correlate. Results show that specific source categories are less important for total $PM_{2.5}$ emissions than for PM_{10}. These categories are the wear of tyres, brakes and road, storage and handling companies, agriculture, and in industry the sectors food, building materials and other industry. Going from PM_{10} to $PM_{2.5}$, the contribution of wear caused by road vehicles decreases from 6% to 3%, of agricultural activities from19% to 6% and of storage and handling companies from 5% to 1%.

Furthermore, road transport exhaust dominates national $PM_{2.5}$ emissions more than emissions of PM_{10}. The share of road transport in total national emissions increases from 30% of national PM_{10} emissions to 50% of national $PM_{2.5}$ emissions.

Table 2.11 $PM_{2.5}$ emissions in the period 1980–1998 (latest knowledge on emissions from agriculture and ventilation emissions from industrial buildings is included).

	1980	1985	1990	1995	1998	1999	2000	*share in 1998 emissions*
million kg								
transport	**31,2**	**27,2**	**24,8**	**19,2**	**17,2**	**16,7**	**16,6**	***54%***
combustion	*30,6*	*26,6*	*24,1*	*18,4*	*16,3*	*15,8*	*15,6*	*52%*
process	*0,6*	*0,6*	*0,7*	*0,8*	*0,9*	*0,9*	*1,0*	*3%*
industry	**25,3**	**16,6**	**16,7**	**10,6**	**7,7**	**n.e.**	**n.e.**	***24%***
combustion			*5,7*	*4,5*	*2,9*			*9%*
process, other			*10,2*	*5,3*	*4,0*			*13%*
process, building venting			*0,8*	*0,8*	*0,8*			*3%*
consumers	**4,4**	**4,3**	**4,3**	**3,9**	**3,8**	**3,6**	**3,5**	***12%***
combustion	*2,5*	*2,5*	*2,5*	*2,1*	*2,1*	*1,9*	*1,8*	*6%*
process	*1,8*	*1,8*	*1,8*	*1,7*	*1,7*	*1,7*	*1,7*	*5%*
agriculture	**1,7**	**1,9**	**1,9**	**2,0**	**2,0**	**2,1**	**2,0**	***6%***
combustion	*0,2*	*0,2*	*0,2*	*0,2*	*0,1*	*0,1*	*0,1*	*0%*
process	*1,5*	*1,7*	*1,7*	*1,9*	*1,9*	*2,0*	*1,9*	*6%*
commercial and institutional sector	**0,2**	**0,2**	**0,7**	**0,7**	**0,3**	**n.e.**	**n.e.**	***1%***
combustion			*0,5*	*0,5*	*0,1*			*0%*
process			*0,2*	*0,2*	*0,2*			*1%*
construction	0,2	0,2	0,2	0,2	0,2	**n.e.**	**n.e.**	***1%***
combustion			0,1	0,1	0,1			*0%*
process			0,1	0,1	0,1			*0%*
waste treatment and disposal	**4,3**	**1,2**	**0,9**	**0,1**	**0,1**	**n.e.**	**n.e.**	***0%***
combustion			*0,0*	*0,0*	*0,0*			*0%*
process			*0,1*	*0,0*	*0,0*			*0%*
energy sector	**8,8**	**1,4**	**1,6**	**0,7**	**0,6**	**n.e.**	**n.e.**	***2%***
combustion			*1,2*	*0,3*	*0,3*			*1%*
process			*0,1*	*0,1*	*0,1*			*0%*
TOTAL	**75,9**	**52,9**	**50,0**	**37,1**	**31,7**	**n.e.**	**n.e.**	***100%***
combustion			*34,2*	*26,0*	*21,8*			*69%*
process			*15,8*	*11,1*	*9,8*			*31%*

Part of international shipping is included, i.e. international maritime transport in Dutch ports and international shipping on Dutch inland waterways.

Table 2.12 Mass fractions $PM_{2.5}$ of total PM_{10} emissions.

	PM2.5 of PM10
transport	
exhaust	1.00
wear: tyres	0.10
wear: brakes	0.50
wear: road	0.20
industry (incl. refineries)	
oil refining: combustion/process	0.80
other industry: process/building venting	
food products and beverages	0.15
chemicals and chemical products	0.55
mineral products	0.35
basic metals	0.50
metalworking	0.60
other	0.10
other industry: combustion	0.60
consumers	
combustion wood	0.95
combustion fossil fuels	0.95
cigarette smoking	1.00
other sources	1.00
agriculture	
livestock houses	0.20
combustion	1.00
other agricultural sources	0.20
commercial and institutional sector	
storage and handling	0.10
combustion	0.95
construction	
combustion	0.95
process	0.10
energy sector	
electricity/heat production: combustion	0.80
electricity/heat production: process	0.10
oil/gas extraction/distribution	1.00
waste treatment and disposal	
waste treatment: combustion	1.00
waste treatment: process	0.10

d. Uncertainty

Uncertainties in emissions from PM_{10} are large. The uncertainty for primary PM is considerably higher than for the precursors of SIA: SO_2, NO_x and NH_3. The reliability of emission estimates for PM_{10} source categories decreases in the following order:

1. energy sector, road transport exhaust, agriculture
2. other mobile sources, ships, wood stoves and fireplaces
3. large industrial process sources
4. small industrial process sources
5. wear of brakes, tyres and road surface

No quantitative estimates of the accuracy of emission estimates will be presented here. In regard to industry, some qualitative remarks will be made.

For industrial emissions, it should be noted that diffuse industrial sources (building venting, materials handling) become relatively more important due to the continued control of classical point source emissions (waste gas streams in pipes and ducts). Because emissions of diffuse industrial sources are less well-known, the reliability of industry estimates (expressed in relative terms as a percentage of total emissions) is expected to decrease in time as well.

From 2000, a change was made in the reporting and collecting of emission data for industry in the Netherlands. The reason for this change was the coming into force of a new reporting guideline, which obliges approximately 250 large industrial companies to report their annual emissions to the national government in the form of an environmental report. A verification study of the quality of the PM emissions revealed deviations in comparison with the data from previous years. The emissions could therefore not be guaranteed to an acceptable level, so it was impossible to present emission totals for PM_{10} for the years 1999 and 2000. The most recent year for which PM emission data is available in the Dutch Pollution and Emission Register is 1998.

Conclusions concerning the quality of the PM_{10} emission inventory can only be based on the emissions as registered for years preceding 1999. Up to 1999, large industrial companies reported how emissions were determined as well as the actual magnitude of PM_{10} emissions. This information was reported per individual emission point. Table 2.13 gives an overview by estimation methodology.

Table 2.13 Determination methods for PM_{10} emissions for industry (excluding ventilation emissions from industrial buildings) in 1995

Determination method	**% of total PM_{10} emissions**
Continuous measurements	3.9
Periodic measurements	14.9
Annual measurements	22.3
Calculation: emission factors	11.9
Calculation: other methods	35.6
Estimations	11.3

The main conclusion that can be drawn from this subsection is that the Dutch Pollution Emission Register currently does not present PM_{10} emission estimates for industry. Current emission reporting practices of industrial companies have been judged to be insufficient to allow an estimate of total Dutch PM_{10} emissions from the year 1999 onwards. However, PM_{10} estimates available for previous years also contain considerable uncertainty, as only 40% of these emissions are based on measurements. This situation gives cause for concern and its improvement should be given higher priority by the responsible ministries, competent authorities and industry. This situation can only be improved by enhanced communication, data exchange and co-operation between the parties involved, and by adequate and actual measurements in industry.

2.5.1.2. Europe

PM_{10} concentrations in the Netherlands are influenced to a large extent by emissions from other countries. The contribution of these foreign sources to concentrations in the Netherlands has been calculated using atmospheric models. Emission estimates for Europe used for this modelling were based on the CEPMEIP study performed by TNO for 1995 (TNO, 2001). This is an update of an older emission inventory for 1990, which was used for the Fifth National Environmental Outlook.

For agriculture, comparable emission factors were used for the Netherlands and foreign countries. However, for residential wood combustion, transport and industry there are differences. For industrial processes, emissions for the Netherlands were based on a detailed Pollution Emissions Register (PER), in contrast to foreign countries for which TNO emission estimates based on a more generalised set of emission factors were used. For transport, emission factors used by TNO for heavy duty vehicles in particular are lower than emission factors used in the Dutch inventory, which are based on TNO measurements. For residential wood combustion, emission factors used for the European CEPMEIP inventory are higher.

2.5.2. Emissions of precursor gases of SIA

Secondary inorganic contributions (SIA) to PM_{10} levels have different sources and can ultimately be traced back to emissions of SO_2, NO_x and NH_3 and to semi-volatile organic compounds.

Emission estimates for SO_2, NO_x and NH_3 for the Netherlands can be found in Van Wee (2001) and RIVM (2000a,b). Emissions for Europe are based on the official emission reports of countries to the UN/ECE Convention on Long-Range Transboundary Air Pollution (EMEP, 1998) and are consistent with data used for the Fifth National Environmental Outlook.

2.5.3. Dispersion modelling of PM

Two model approaches have been used to provide information on sources and sinks of PM on an annually averaged and on a daily basis. These two model approaches are long-term averages with OPS and diurnal concentrations (short-term) with EUROS and LOTOS.

2.5.3.1. Modelling annual averages

Annually averaged concentrations of particulate matter are calculated using the OPS model (Van Jaarsveld, 1995), a long-term Lagrangian transport and deposition model. It quantifies the relation between individual sources or source areas of air pollutants and individual receptors. The model is statistical in the sense that concentration and deposition values are calculated for a number of typical situations. The long-term value is obtained by summation of these values, weighted with their relative frequencies. All relations governing the transport and deposition process are solved analytically, allowing the use of non-gridded receptors and sources, and of variable grid sizes. Transport from a source to a receptor is assumed to take place in straight,

well-mixed sectors of a particular height and with horizontal angles of 30°. Corrections are applied close to the source to account for height of emission and vertical dispersion.

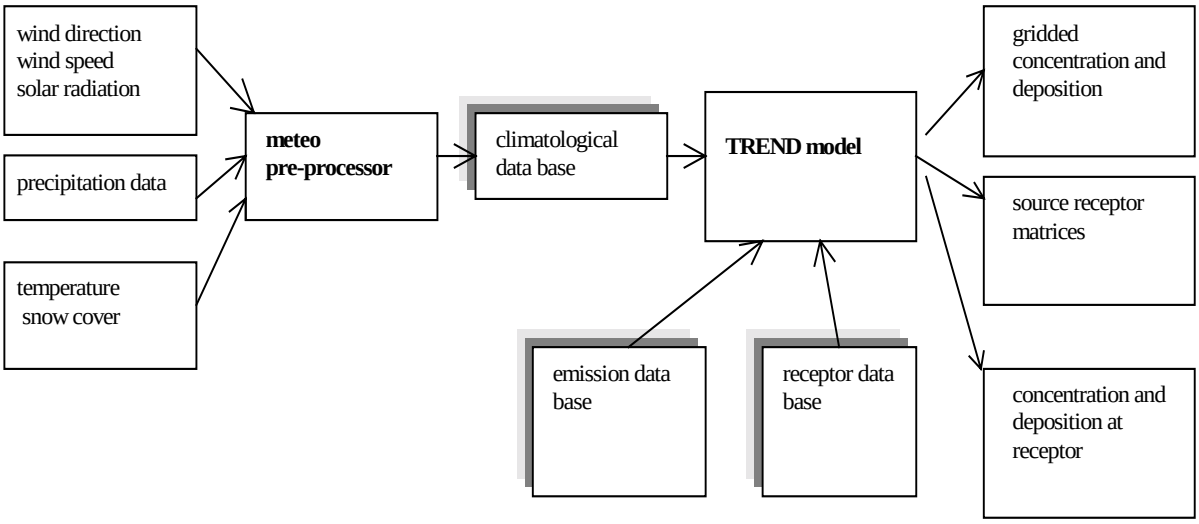


Figure 2.22 Schematic overview of the OPS model (Van Jaarsveld, 1995).

The OPS model consists of two main parts (See Figure 2.22):

- A special meteorology pre-processor calculates hourly transport trajectories arriving at a receptor based on wind observations. Other parameters like atmospheric stability and mixing-height classification, which define the atmospheric state along the trajectories, are also derived from the observed data. Together they provide the necessary frequency distribution statistics for the period of interest. These statistics are then stored in a climatological database for use in the main part of the model.
- The model itself carries out the actual calculations. The mathematical formulation of the model is based on the change of mass for a pollutant due to chemical conversion and deposition. These processes are based on variables including dry deposition velocities, wet scavenging coefficients, pseudo first-order chemical reaction constant for inorganic secondary aerosol precursors (formation of secondary organic aerosol (SOA) is not included), physically inert primary particles (no nucleation and agglomeration included) and the mixing height. The roughness length is based on a 5 x 5 km^2 grid map of land-use of the Netherlands. Formation of secondary inorganic aerosol (SIA) is modelled from primary emissions of sulphur dioxide (SO_2), oxides of nitrogen (NO_x) and ammonia (NH_3). The formation of sulphate, nitrate and ammonium aerosol includes contributions from the gas phase and oxidation in the heterogeneous and aqueous phase. Particle growth processes are not parameterised. The dry deposition of secondary aerosol is modelled assuming that these particles are all within the same lowest size range (<1 μm).

The dispersion of emissions of primary particulates is performed using four particle size classes. Because of the very different atmospheric residence times of particles of

different sizes, it is necessary to describe dry deposition as a dynamic process, i.e. as a function of particle size. Concentrations and depositions for each of the four classes are weighted with the percentage of the total particle mass allocated to the individual classes. This approach is particularly useful for the modelling of primary emitted particles, because they usually cover a broad size range of particle sizes, often including a significant fraction of large particles.

The particle size classes are listed in Table 2.14. As the larger particles tend to be removed faster than small ones, the actual size distribution calculated for a certain receptor site always shifts towards smaller diameters (Table 2.14). As a result, the deposition velocity for particles transported over long distances is always lower than the deposition velocity of freshly emitted particles.

The model can be applied for time resolutions varying between seasons and years. The spatial resolution of the receptors is determined by the spatial resolution of the emission inventory (typically 5 x 5 km^2). Emissions can be located over a large part of Europe, e.g. 2,000 x 2,000 km^2. The first of the four size classes in Table 2.14 (particles smaller than 0.95 µm) also comprises UF particles smaller than 0.1 µm. The class of particles below 0.1 µm in diameter is high in numbers but low in mass. The smallest UF particles have atmospheric residence times in the order of minutes instead of the hours presented in Table 2.14. However, because of their minor contribution to the mass of particles < 0.95 µm, the values presented in Table 2.14 are still indicative for the average mass of PM in the size class.

Table 2.14 Properties of the particle size classes with respect to dry deposition for land surfaces

Particle size class µm)	Median aerodynamic diameter µm)	Initial PSD for primary PM_{10} (%)	Mean dry deposition velocity V_d (m/s)[a]	Mean atmospheric residence time $T_{1/2}$ (h)	Mean transport distance during $T_{1/2}$ (km)[c]
< 0.95	0.2	42	0.00065	63	1,150
0.95–2.5	1.5	33	0.0025	11	200
2.5–4	1.5	33	0.0025	11	200
4–10	6	14	0.0071	8	145

[a] Dry deposition velocity for 50 m altitude and a roughness length of 0.15 m.
[b] $T_{1/2} = \ln 2 / (v_d/z_i + \square)$, where z_i is the mixing layer height and □ is the mean scavenging rate.
[c] Transport distance calculated using a 5 m/s wind speed.

For reasons of consistency with previous work, similar deposition parameters are used for the 0.95–2.5 µm and the 2.5–4 µm classes. This approximation leads to a maximum difference of 3.7% in modelled concentrations compared with a differentiated deposition scheme for these two size classes.

In earlier presentations of PM_{10} results of the OPS model we used two particle size classes, i.e. 10–20 µm and > 20 µm, in addition to those listed in Table 2.14. This was in accordance with the definition of PM_{10} given in EU Council Directive 1999/30/EC: 'PM_{10} shall mean particulate matter which passes through a size-selective inlet with a *50% efficiency cut-off* at 10 µm aerodynamic diameter'. However, the European

atmospheric modelling community treats PM_{10} as a mix of particles with a *maximum* diameter of 10 µm. The results presented in the current report are computed using the latter definition. Sensitivity runs showed that similar emissions lead to at most 8% higher concentrations of PM_{10} when all mass is distributed among the four size classes < 10 µm.

Parties to the UN Convention on Long-Range Transboundary Air Pollution are supposed to provide national emission estimates of primary particles for the year 2000. EMEP in collaboration with the EEA initiated the so-called Coordinated European Programme on Particulate Matter Emissions Inventories, Projections and Guidance (CEPMEIP) in order to assist various national experts. The results of this project for the year 1995 are now available at http://www.mep.tno.nl/emissions/. Compared with a previous estimate for 1990, total European emissions of PM_{10} and $PM_{2.5}$ in 1995 increased by 27% and 29% respectively. The changes in emissions estimates for 1990 and 1995 vary considerably between countries and source groups (EMEP, 2001).

a. Modelling results for PM_{10}

This subsection addresses the budget of primary and secondary inorganic PM_{10} that is calculated on the basis of known emission sources. In the mid-1990s the annual average contribution of Dutch sources (primary and secondary) was approximately 6 µg/m^3 to PM_{10}, whereas other countries supplied roughly 11 µg/m^3 (primary and secondary). Approximately 6 µg/m^3 of the annual average consists of modelled primary particles and half of this is the result of emissions from 'other' non-Dutch sources (see Table 2.15).

Approximately 10–11 µg/m^3 of the PM_{10} concentration consists of SIA, of which the contribution of nitrate aerosol is the major part. The source category 'transport' is the main contributor, ~6 µg/m^3 to the concentration of PM_{10}. 'Agriculture' 'Energy' and 'Others' each contribute approximately 3 µg/m^3 on an annual average basis. Table 2.15 presents the concentrations of PM_{10} modelled using OPS in 1995. A similar modelling exercise for 2010 is presented in Table 6.7.

Modelled concentrations of particulate matter decreased by 16 µg/m^3 between 1980 and 1995. About 14 µg/m^3 of this reduction is due to the decrease in sources of SIA and primary PM_{10} outside the Netherlands. A closer look at the downward trend due to the abatement of Dutch sources shows a decrease of about 1 µg/m^3 as a result of a 55 ktonne reduction in primary PM_{10} emissions (from 116 ktonnes in 1980 to 61 ktonnes in 1995). The absolute amount of inorganic secondary PM_{10} (SIA) decreased by 9 µg/m^3 due to reductions in emissions of SO_2, NO_x and NH_3. Slightly more than 1.5 µg/m^3 of the decrease in SIA is the result of reductions in emissions of acidifying species in the Netherlands, whereas the remaining 7.5 µg/m^3 is the consequence of reductions in other countries.

Table 2.15 Annually averaged primary and secondary inorganic concentrations of PM_{10} averaged over the Netherlands. Calculated for 1995, based on emissions for the Netherlands and the CEPMEIP inventory for Europe.

Dutch sources	Primary PM_{10} ($\mu g/m^3$)	NH_x ($\mu g/m^3$)	NO_y ($\mu g/m^3$)	SO_x ($\mu g/m^3$)	Summed concentration ($\mu g/m^3$)
Industry	0.4	0.0	0.1	0.1	0.6
Energy	0.0	0.0	0.1	0.1	0.2
Transport *	1.5	0.0	1.0	0.1	2.6
Agriculture	0.5	0.9	0.0	0.0	1.4
Others	0.5	0.1	0.1	0.0	0.7
Sum	**2.9**	**1.0**	**1.4**	**0.2**	**5.5**
Other countries					
Industry	0.9	0.0	0.1	0.1	1.0
Energy	0.4	0.0	0.7	1.9	3.0
Transport *	0.9	0.0	2.1	0.3	3.3
Agriculture	0.1	1.2	0.0	0.0	1.4
Others	0.7	0.0	0.5	1.1	2.3
Sum	**3.0**	**1.2**	**3.3**	**3.4**	**10.9**
All sources					
Sum	**6.0**	**2.2**	**4.6**	**3.6**	**16.5**

Note: the data presented in this table include a 0.4 $\mu g/m^3$ NH_4 correction, being a result of the structural difference between measured and modelled NH_3 concentrations. Gaseous HNO_3 (1.5 $\mu g/m^3$) is not contained in the NO_y column.
* including international shipping

In general, the relative contribution of Dutch anthropogenic sources to the PM_{10} concentration is about 35%. In the Amsterdam and Rotterdam areas, the contribution of Dutch emissions is significantly higher than in other regions. Here, we model that 50–60% of PM_{10} is of Dutch origin, of which 35–40% (i.e. about 9 $\mu g/m^3$) is primary PM_{10}. In more regional areas foreign sources contribute 65–75%, with the accent on primary and nitrate aerosol.

Figure 2.23.a shows the modelled concentrations of PM_{10} in the Netherlands. The maps in Figures 2.23.b and 2.23.c show ranges of the total PM_{10} field, which are constructed by adding the modelled 1995 field with the 'difference' maps presented in Figures 2.33 and 2.34 in subsection 2.6.1.3. In this assessment 2–9% of the surface area of the Netherlands experiences PM levels above 40 $\mu g/m^3$, whereas on 60–80% of the surface area PM levels in excess of 30 $\mu g/m^3$ are computed.

The meteorology-corrected annually averaged concentration of PM_{10} measured in 1995 was 38 $\mu g/m^3$. It is relevant to refer to the corrected concentrations here because long-term averaged meteorology was used in our calculations. Our modelled assessment of this average is 31–35 $\mu g/m^3$, of which 48–52% is explained by known sources of emissions.

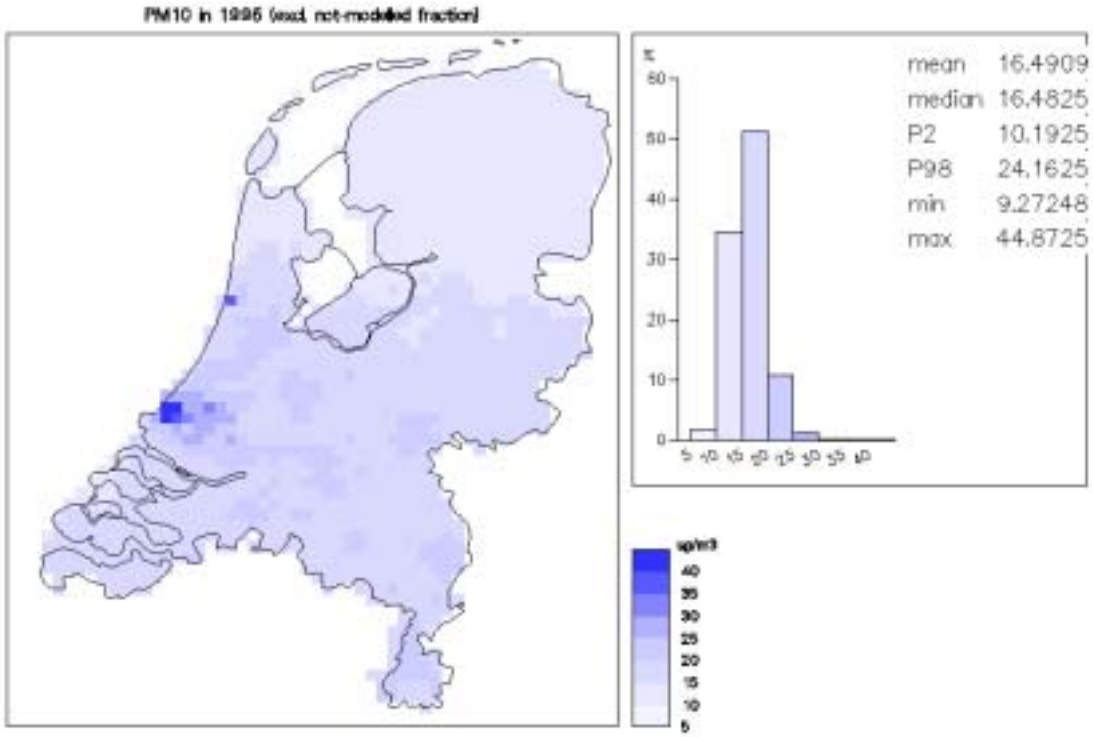


Figure 2.23.a Modelled concentration of PM_{10} in the Netherlands. The estimate of the non-modelled fraction was not added to this map.

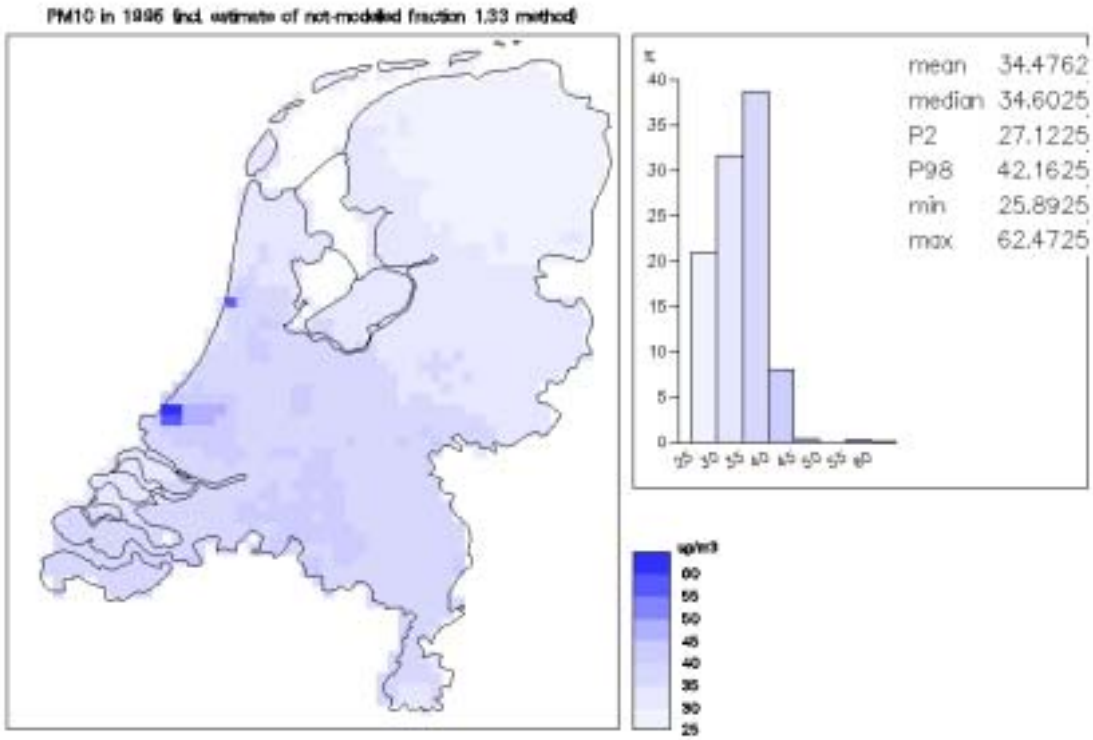


Figure 2.23.b Modelled estimate of the PM_{10} field in the Netherlands, 1995. In this figure the 'difference' map shown in Figure 2.33 with constant correction factor was added to the modelled field.

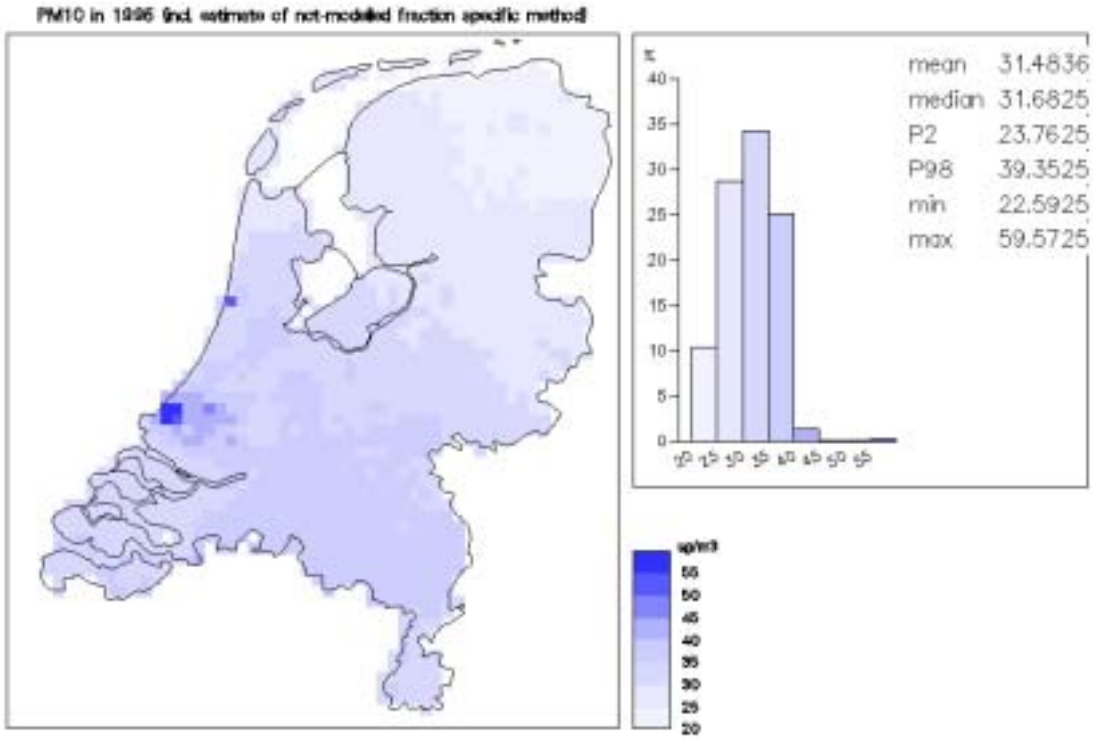


Figure 2.23.c Modelled estimate of the PM_{10} field in the Netherlands, 1995. In this figure the 'difference' map shown in Figure 2.34 with constituent-dependent correction factor was added to the modelled field.

b. Modelling results of $PM_{2.5}$

Table 2.16 shows results for $PM_{2.5}$ from calculations made with OPS. The calculated amount of SIA is allocated to $PM_{2.5}$ using 100% for NH_4, 80% for NO_y and 90% for SO_4 (Visser *et al.*, 2001).

Table 2.16 Annually averaged primary and secondary inorganic concentrations of $PM_{2.5}$ averaged over the Netherlands. Calculated for 1995 and based on the emissions as given in subsections 2.5.1. and 2.5.2. for the Netherlands and the CEPMEIP inventory for Europe.

Dutch sources	Primary $PM_{2.5}$ ($\mu g/m^3$)	NH_x ($\mu g/m^3$)	NO_y ($\mu g/m^3$)	SO_x ($\mu g/m^3$)	Summed concentration ($\mu g/m^3$)
Industry	0.2	0.0	0.1	0.0	0.4
Energy	0.0	0.0	0.1	0.1	0.2
Transport *	1.4	0.0	0.8	0.1	2.3
Agriculture	0.1	0.9	0.0	0.0	1.1
Others	0.4	0.1	0.1	0.0	0.6
Sum	**2.2**	**1.0**	**1.1**	**0.2**	**4.6**
Other countries					
Industry	0.6	0.0	0.0	0.1	0.7
Energy	0.2	0.0	0.5	1.7	2.5
Transport *	0.8	0.0	1.7	0.3	2.8
Agriculture	0.1	1.2	0.0	0.0	1.3
Others	0.6	0.0	0.4	1.0	1.9
Sum	**2.3**	**1.2**	**2.6**	**3.1**	**9.2**
All sources					
Sum	**4.5**	**2.2**	**3.7**	**3.3**	**13.8**

* including international shipping

Figure 2.24.a presents the mapped modelled results of $PM_{2.5}$. Figures 2.24.b and 2.24.c show the ranges of the modelled concentrations of $PM_{2.5}$ in the Netherlands. The latter maps were constructed by combining the modelled 1995 field with the 'difference' maps given in Figures 2.35 and 2.36 in subsection 2.6.1.3. Unfortunately, these results for $PM_{2.5}$ cannot be compared with measurements due to a lack of data.

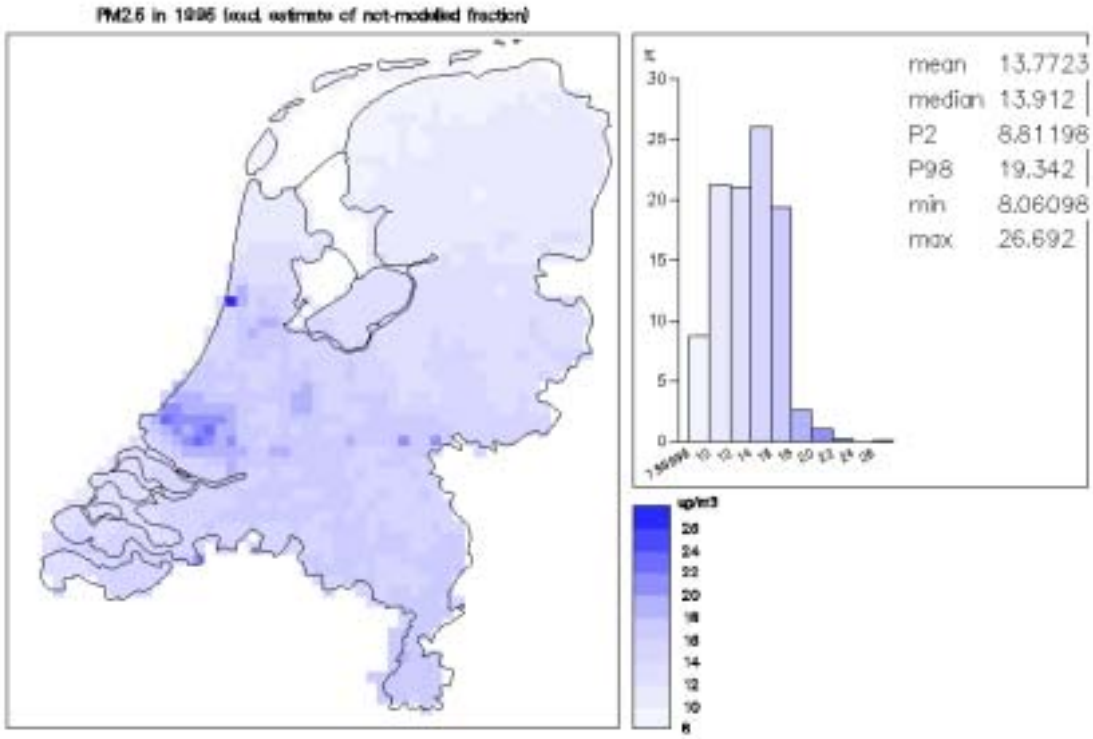


Figure 2.24.a Modelled concentration of $PM_{2.5}$ in the Netherlands. The estimate of the non-modelled fraction was not added to this map

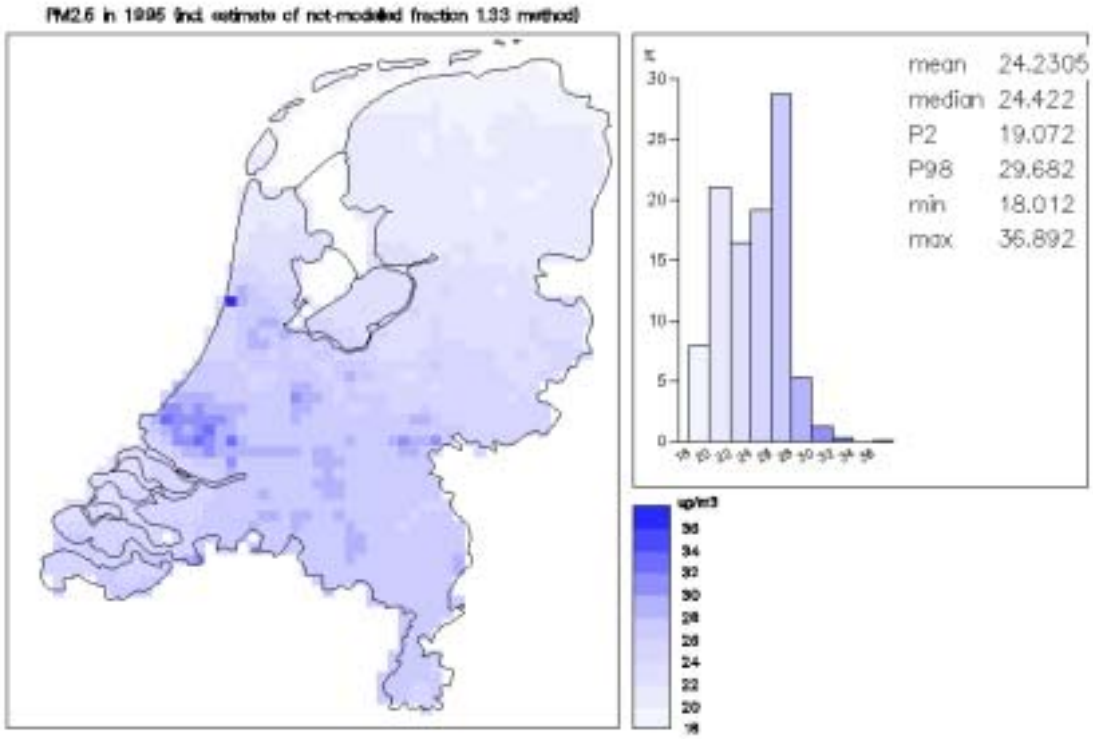


Figure 2.24.b Modelled estimate of the $PM_{2.5}$ field in the Netherlands, 1995. In this figure the 'difference' map shown in Figure 2.35 with constant correction factor was added to the modelled field.

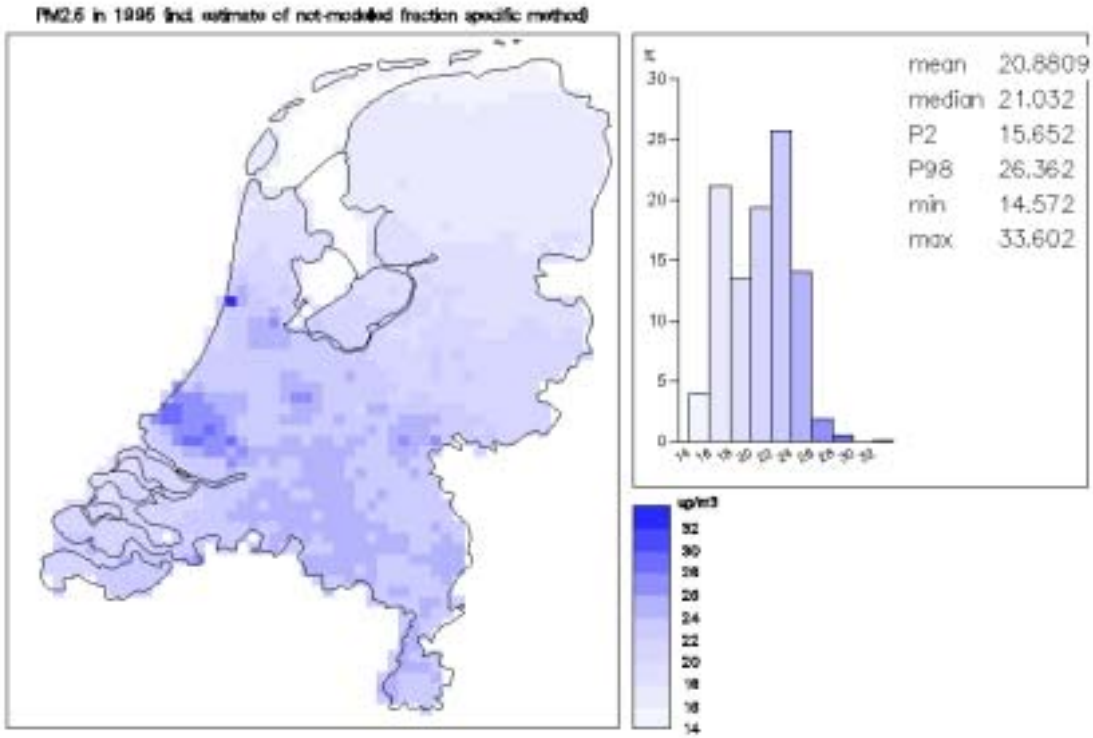


Figure 2.24.c Modelled estimate of the $PM_{2.5}$ field in the Netherlands, 1995. In this figure the 'difference' map shown in Figure 2.36 with constituent-dependent correction factor was added to the modelled field.

c. Uncertainties

Modelled results from the OPS model incorporate a number of uncertainties (Van Jaarsveld, 1995; Visser *et al.*, 2001), for which there are two main sources:

- Uncertainties in the dispersion model. Shortcomings in the description of parameterised processes (emission, dispersion, transport, atmospheric chemistry and deposition), uncertainties in meteorological data and uncertainties in specific parameters for the modelled compounds, i.e. descriptions of dry and wet deposition and chemical conversion rates.
- Uncertainties in emissions. Incomplete knowledge of particle producing processes and uncertainties in emission factors. Furthermore, there are uncertainties in process parameters such as the location of sources, emission height, heat content, initial size distribution, temporal variations.

Model uncertainties

On the basis of comparisons with measurements, the average concentrations on a regional and national scale can be calculated with the OPS model if loss parameters are properly chosen and the emissions file is correct. Errors are expected to be smaller than the values presented in Table 2.17 (Van Jaarsveld, 1989):

Table 2.17 Errors in modelled concentrations of primary PM as a function of averaging period. Data are representative of the national average.

Period	Concentration
Long-term average	*c.*10%
Annual average	*c.*15%
Seasonal average	*c.*20%

These errors can be interpreted as 1-σ or 68% confidence limits

Emission uncertainties
The spatial resolution of concentration estimates can be no better than the spatial resolution of the emission description. The accuracy of emission estimates depends on the process causing emissions and the nature of the sources. In general, the accuracy of emission estimates for diffuse sources is smaller than that for large point sources. Also, the accuracy of the emissions of several pollutants from the same source can vary considerably. Emissions of SO_2 are usually the result of sulphur in fuel or raw materials and so are relatively well-known. Emissions of NO_x, for example, depend much more on process conditions and are more uncertain. Tentative estimates of systematic errors in national totals of SO_2 and NO_x are about 20% (VROM, 1998). Emission estimates of PM_{10} have a margin of accuracy of -15% to +40% (Wesselink *et al.*, 1998). The uncertainties in natural emissions are often larger than 100%. Note that natural emissions are currently not modelled with the OPS model because they are not included in the PER.

Total uncertainty
Total uncertainties in modelled PM_{10} concentrations – based on long-term averaged meteorology using emission estimates of a specific year – are estimated at 25% (1-SD or 68% confidence limits). Contributions to the concentration of PM_{10} caused by particular source groups can be modelled with more accuracy where more knowledge is available on emissions of this source group (e.g. energy).

2.5.3.2. Modelling of daily concentrations

The modelling of daily PM concentrations is relevant because of the daily average standards in the EU in 2005. The regional dispersion models EUROS and LOTOS are used to obtain insight into daily averaged PM concentrations, sources, sinks and precursors, their dynamics and the mutual influence of all these factors on ambient PM levels.

Annually as well as daily averaged PM concentrations for Europe are calculated using two regional chemical dispersion models, EUROS (Jacobs and van Pul, 1994; van Loon 1996; Matthijsen *et al.*, 2002) and LOTOS (Builtjes, 1992). In both models, daily and annual averages are based on a summation of calculated hourly values.

a. EUROS
The EUROS model was developed at RIVM. It is a Eulerian air quality model used to simulate the dispersion and transport of components in the lower troposphere in order to evaluate possible policy measures for Long-Range Transboundary Air Pollution (LRTAP). The modelled area extends over a large part of Europe. The horizontal base

grid consists of 52 x 55 grid cells with a 0.55° x 0.55° lat/lon shifted pole projection (about 60 x 60 km^2 in the Netherlands). Local uniform grid refinement is possible up to 4 levels, resulting in a maximum latitude-longitude resolution of 0.069° x 0.069° (about 7.5 x 7.5 km^2 in the Netherlands). Transport is based on half-hourly updates from ECMWF meteorological fields for wind velocity components u and v, temperature, relative humidity and the geopotential height. Horizontal transport is described by advection and diffusion, whereas vertical transport is treated using a well-mixed boundary layer concept.

The atmospheric vertical grid structure in the EUROS version used here consists of four layers: the surface layer (SL), the mixing layer (ML), the reservoir layer (RL) and the top layer (TL). The surface layer and the mixing layer together form the atmospheric boundary layer. The depths of the four layers are modelled to be uniform over the whole domain, but vary in time during the day due to the growth of the mixing layer – except for the surface layer whose depth is fixed at 50 m. The growth of the mixing layer during daylight hours is represented through a constant climatological growth rate.

Primary emitted particles are distributed in the same four size classes as included in the CEPMEIP emission inventory (see the extensive information on the web site **http://www.mep.tno.nl/emissions/)**. Secondary sulphate and nitrate particles are removed by dry and wet deposition (at present, scavenging of primary PM by rain is not applied). Secondary PM formation is described in a condensed ozone scheme, which includes four production reactions for particulate sulphate and nitrate: two gas-phase production reactions and two first order reactions representing formation in cloud water and on existing aerosols. The SO_2 and NO_x emissions are described with monthly, weekly and daily variations. PM_{10} emissions have a daily emission profile. The results for NO_3 particles are not included here since validation showed that their parameterisation is not yet adequate.

Sources, sinks and processes that are not, or not yet, included or for which data are currently too provisional are:
1. Natural PM (e.g. sea salt, wind-blown crustal material).
2. Re-suspension of soil dust due to anthropogenic activity (e.g. traffic).
3. Explicit description of ammonia (NH_3) and ammonium (NH_4^+).
4. Formation of secondary organic aerosol (SOA).
5. Particle microphysics (interaction between particles, e.g. coagulation).
6. Explicit treatment of aqueous-phase and heterogeneous chemistry.

For nitrate aerosol, a simplified mechanism is used (no aerosol thermodynamics or ammonium variations) and there is a simple wet removal parameterisation in the current version of the EUROS model.

Future efforts will be dedicated to improving secondary inorganic aerosol (SIA) modelling: a more detailed mechanism for nitrate and the inclusion of ammonium. In addition, sea salt will be included and attention will also focus on validation of the (already implemented) improved description of vertical resolution and the vertical advection process.

b. LOTOS

The LOTOS model was developed at TNO. It is a three-dimensional transport chemistry model of intermediate complexity covering Europe. The idea behind the model is that it should contain all relevant processes (explicitly or in a parameterised form) in such a way that hour-by-hour calculations over periods of years are feasible. LOTOS was originally developed for ozone modelling. In the last few years the model has been extended to include aerosols. A general and detailed description of the model can be found in Builtjes (1992). Some relevant details on the model concept and the modelling of aerosols within LOTOS are given below.

The horizontal resolution of the model is 0.5º by 0.25º lon/lat (about 55 x 27 km^2). In the vertical, the concept of dynamical layers is applied: the depths of the three layers depend on the height of the mixing layer (varying in time and space). The first layer represents the mixing layer, the other two are equally distributed over the rest of the vertical domain. The meteorological input for LOTOS is prepared by the Free University of Berlin. The emission database of relevant components was constructed by TNO for the base year 1995. For later years, the 1995 emissions are scaled, based on the national emission totals.

Sulphate is formed in the gaseous phase as well as in the liquid phase. The oxidation of sulphur dioxide by the OH radical is represented in the gas phase reaction mechanism CBM-IV. Another important oxidation pathway, in particular in winter, is the formation of sulphate in clouds. Due to insufficient data on clouds in the meteorological input, it is difficult to explicitly represent this process in a model. Therefore, it is represented with a first order reaction constant that varies with cloud cover and relative humidity, similar to the approach followed in EUROS (Matthijsen *et al.*, 2002).

The equilibrium in the formation of ammonium nitrates, for example, is very sensitive to ambient conditions and has been calculated using a modified version of the MARS system, a module which is embedded in the MADE module (Ackermann *et al.*, 1995; Ackermann *et al.*, 1998). This module also calculates the size distribution of aerosols using a modal approach.

Sources, sinks and processes that are not, or not yet, included or for which data are currently too provisional are:

1. Natural PM sources (e.g. sea salt, wind-blown crustal material).
2. Re-suspension of soil dust due to anthropogenic activity (e.g. traffic).
3. Highly parameterised cloud chemistry (oxidation of SO_2).
4. Formation of secondary organic aerosols (SOA).
5. Surface layer (the mixing layer is taken as the lowest layer).

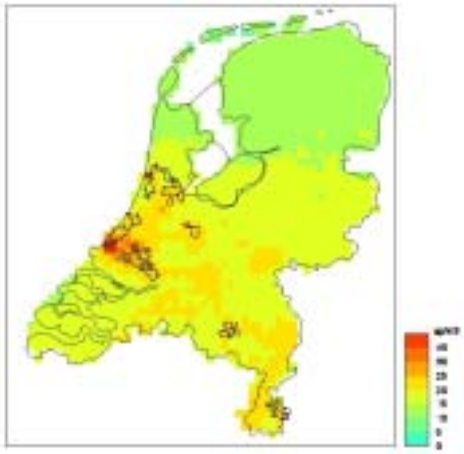
The LOTOS module contains a simple wet removal parameterisation. Also, the distribution over time of NH_3 emissions is very uncertain. Efforts will be dedicated to implementing a more detailed mechanism for SO_2 oxidation/cloud chemistry, the inclusion of sea salt, black carbon (BS or EC) and improved NH_3 emissions and depositions. Finally, attention will also focus on improving the vertical resolution (e.g. inclusion of surface layer) and coupling with a grid-refined version.

It should be noted that model results for PM are provisional, because validation of the PM_{10} concentrations and sources and sinks is still ongoing. For this reason, the data inferred from the simulations should be treated as qualitative information rather than quantitative. Results from comparative model exercises will therefore be used, since these are probably the most reliable.

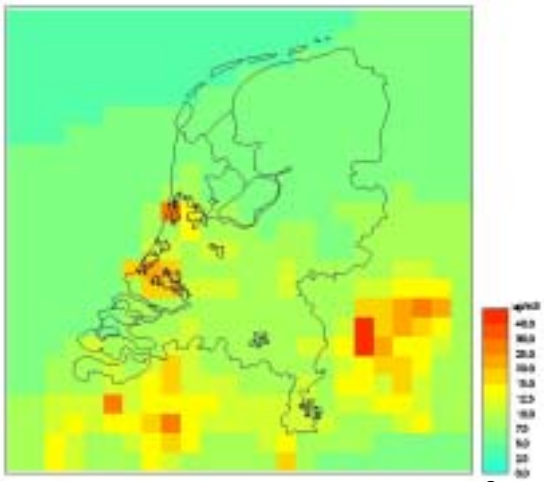
The primary PM emissions in EUROS are based on emissions from subsections 2.5.1 and 2.5.2. Emissions for the formation of secondary aerosols (SO_2, NO_x, NH_3) are taken from the Fifth National Environmental Outlook (further referred to by its Dutch abbreviation MV5; RIVM, 2000). For LOTOS, slightly higher country totals were used for primary PM as a result of a different CEPMEIP version (e.g. primary PM emissions in the Netherlands are about 10% higher).

Before discussing the daily-average results, we first give an overview of the distributions of the annually averaged total PM concentrations for 1995 in the Netherlands from the models OPS, EUROS and LOTOS; Figures 2.25.a, b and c respectively. It should be remembered that within OPS and LOTOS total PM consists of primary PM_{10} and the secondary aerosols sulphate, nitrate and ammonium, while for EUROS ammonium and nitrate are not incorporated, which complicates a quantitative comparison. In order to obtain a better comparison between the models, the EUROS results should be increased by annually averaged concentrations of ammonium and nitrate, i.e. in total approximately 8 $\mu g/m^3$ (OPS estimate). The gradients of these components over the Netherlands are relatively small when compared with PM_{10}. We corrected this difference visually by using a slightly different classification (i.e. concentrations below 20 $\mu g/m^3$ are shown at a higher resolution) to present the EUROS results in Figure 2.25.b.

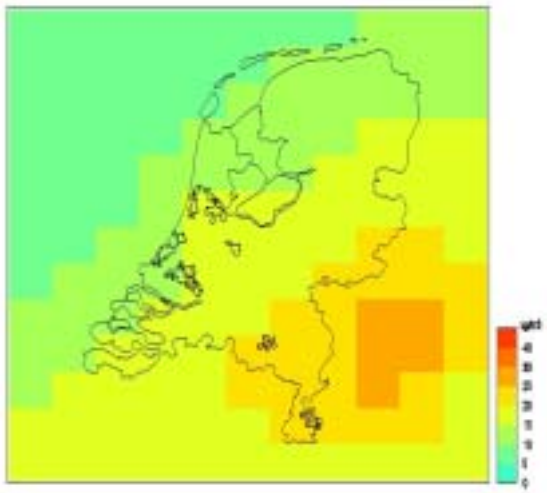
The main features of urban and/or industrialised areas in the Netherlands (and in Belgium and Germany for EUROS) are resolved equally well by OPS and EUROS modelling despite their grid size differences (5 x 5 km^2 and 15 x 15 km^2 respectively). The use of different classifications illustrates nicely the qualitative agreement between the models. The large concentration gradients found near populated areas mainly result from local sources of primary PM_{10} emissions.



2.25.a OPS (5 x 5 km^2)



2.25.b ***EUROS (15 x 15 km^2)***



2.25.c LOTOS (30 x 30 km^2)

Figure 2.25 Annually averaged concentrations of total PM_{10} in the Netherlands for 1995 in $\mu g/m^3$. The results from three models are shown, each with different grid resolutions: a) OPS (5 x 5 km^2), b) EUROS (15 x 15 km^2), c) LOTOS (30 x 30 km^2). Note that EUROS results are presented using a different classification in order to visually correct for the non-modelled secondary aerosols (i.e. NH_4 and NO_3; see also comments in the text). Within each figure the borders of the Netherlands and the cities with more than 250,000 inhabitants are shown.

Apart from the large industrialised area of Germany (the Ruhr), the LOTOS results do not distinguish urban and/or industrialised areas in the Netherlands or Belgium due to the larger grid size. Local concentration differences are spread over a larger grid area and are therefore not resolved. The small differences in emissions between LOTOS and OPS/EUROS does not result in substantial differences in the modelled concentrations. The overall background values for LOTOS are similar to those of OPS. The good overall agreement between the models implies that the differences in

model concept do not produce large differences in the average dispersion of particles and the formation processes of SIA.

In Figure 2.26 the daily averaged total PM_{10} concentrations from the LOTOS model are compared with measurements. The values are averaged results from the three regional NAQMN stations De Zilk, Vredepeel and Wieringerwerf (the location of these stations and the corresponding numbers 444, 131 and 538 can be found in Figure 2.3). The model results (red solid line, average 16.3 $\mu g/m^3$) clearly underestimate the measurements (blue solid line, average 35.5 $\mu g/m^3$), yielding a ratio of 0.46. The underestimation is a result of the uncertainties and incompleteness in the modelled processes and emission inventories. For OPS results it is shown that this so-called gap between models and measurements has been closed adequately for annual averages when all uncertainties and incompleteness are taken into account (see subsection 2.6.1).

The ratio of annually averaged concentrations of model and measurements between the individual locations varies considerably and is probably due to site-specific conditions. For the De Zilk, Vredepeel and Wieringerwerf stations the ratios are 0.39, 0.63 and 0.33 respectively (the corresponding measured annual averages are 33.8, 38.4 and 34.4 $\mu g/m^3$). To exemplify site-specific conditions we compared the coastal station De Zilk with Vredepeel, which lies far inland. De Zilk is therefore more exposed to non-modelled sea-salt aerosols than Vredepeel. The difference in sea-salt exposure between both stations is approximately 4 $\mu g/m^3$. Moreover, the grid cell that contains De Zilk covers sea as well as land surface, an inhomogeneity that leads to larger model uncertainties compared with the homogeneous rural background conditions at Vredepeel.

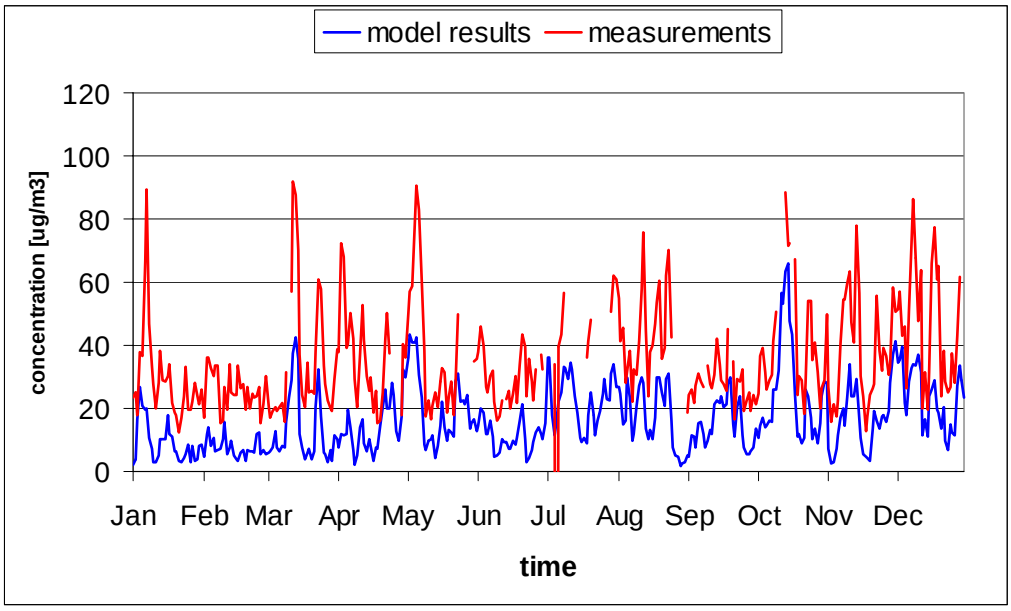


Figure 2.26 Daily averaged PM_{10} total concentrations ($\mu g/m^3$) in 1995 for measurements and model results. The blue solid line represents the average from the NAQMN stations De Zilk #444, Vredepeel #131 and Wieringerwerf #538 and the red solid line the averaged model results for the same stations.

The variations on small time scales are large, with high maxima during episodes lasting from a few days to over a week. Meteorology has a substantial influence on the behaviour of such episodes, as is shown by Visser and Römer (1999). In a qualitative sense the variations in the modelled values agree reasonably well with the measured values (their regression line is y = 0.49x - 1.60, with R^2 = 0.59, y and x being the modelled and measured values respectively), which supports the validity of the model approach regarding dispersion processes. The uncertainty for daily averages is not known for our model results, but is likely to be larger than the uncertainty given for OPS annual averages (i.e. 15%, see Table 2.17). Important for short-term modelling is the inclusion of vertical transport, which is partly based on climate. This makes it difficult and sometimes even impossible to represent episodic features induced by meteorological conditions adequately. Furthermore, as explained in subsection 2.2.1.1, the uncertainties in the measurements range from –60% to +50%, which complicates the comparison of modelled and measured concentrations on a daily basis.

c. Modelling results of SIA

The above discussion on total PM_{10} results clearly revealed the difficulties that arise in the comparison between models and measurements. In order to obtain a quantitative comparison we must focus on the secondary aerosols sulphate, nitrate and ammonium. For SIA the emissions inventories and the production processes in models are relatively well-known, so the quality of the model performance is much better addressed. Also, the contribution of the secondary PM_{10} fraction is much larger, and thus far more important than the primary fraction (e.g. for 1995 the annual average values are 12.9 µg/m^3 and 3.5 µg/m^3 respectively).

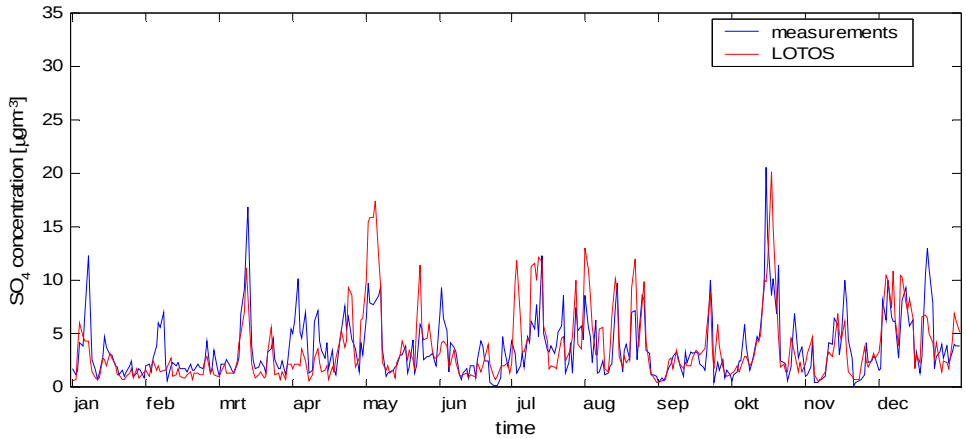


Figure 2.27 Model results (LOTOS) and measurements for daily averaged sulphate concentrations for 1995. The values are averages obtained from the NAQMN regional stations De Zilk #444, Vredepeel #131 and Wieringerwerf #538.

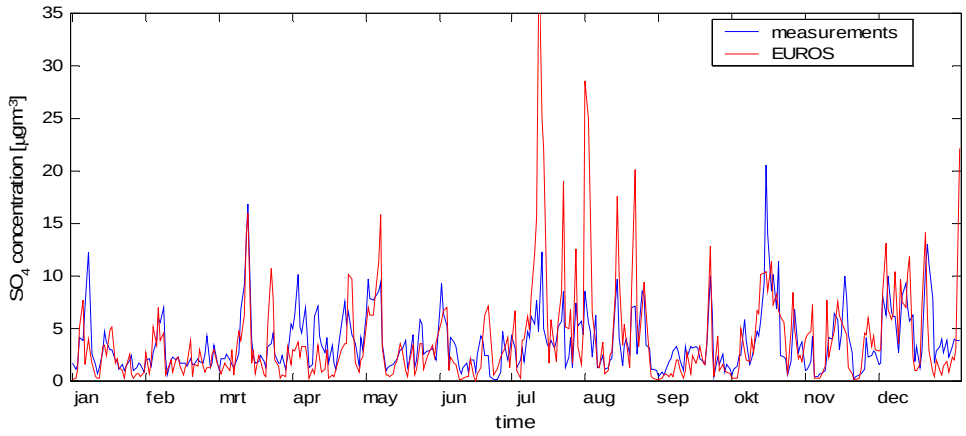


Figure 2.28 Model results (EUROS) and measurements for daily averaged sulphate concentrations for 1995. The values are averages obtained from the NAQMN regional stations De Zilk #444, Vredepeel #131 and Wieringerwerf #538.

In Figures 2.27 and 2.28 the daily averages of modelled (LOTOS and EUROS) and measured values are shown for sulphate using the average from several regional NAQMN measurement locations (see Figure caption). Although for certain episodes (e.g. periods during July for EUROS and during February and April for LOTOS) distinct differences are observed, it appears that, in contrast to total PM_{10}, the models are well capable of reproducing the measurement results. For nitrate and ammonium the results are quite similar and not presented.

Differences between models and measurements may be due to uncertainties in the modelled processes and emission inventories. It should also be realised here that point measurements are compared with grid-cell-averaged model results. This introduces uncertainties due, for example, to local conditions that are not resolved in the model, giving rise to unknown scatter. As was the case for PM_{10}, the measurements of SIA contain considerable uncertainties that may be partly responsible for the differences between models and measurements. These uncertainties are not addressed specifically in subsection 2.2.1.1., so we mention some results from Visser *et al.* (2001). Measurements performed at several locations using filter samplers showed that for the summed concentrations of nitrate, sulphate and ammonium the standard deviations between monitors at the same location varied between 3 and 4 $\mu g/m^3$. In addition, a study performed by the KEMA (Blank, 2001) showed that the absolute *minimum* (i.e. several sources of uncertainty could not be quantified) relative standard deviation for measurements of nitrate, sulphate and ammonium performed at the NAQMN stations are 11, 22 and 18% respectively.

We conclude that in the light of the aforementioned uncertainties the results from models and measurements of SIA compare satisfactorily for nitrate, sulphate and ammonium. In line with this are the results of measurements and model results (EUROS) obtained for trends in annual median and 95-percentile values for sulphate aerosols during the period 1994–1999. The agreement on 95-percentile values especially shows that models are well capable of reproducing the periodic high concentrations for secondary aerosols, which is of importance for estimating the daily exceedances of limit values.

d. Scenario study for 2010

In the EU daughter directive for PM the daily standard for 2005 has been set at 50 $\mu g/m^3$, not to be exceeded more than 35 times a year, and the maximum annual average at 40 $\mu g/m^3$. The indicative values for 2010 are 7 permitted exceedances and a maximum annual average of 20 $\mu g/m^3$. From results presented earlier in this Section it follows that it is impossible to estimate the correct daily averaged PM_{10} levels using model results only due to the incompleteness of modelled processes and emission inventories. On the other hand, for the most important secondary aerosols (nitrate, sulphate and ammonium; see Section 2.6, for example), models and measurements show fair agreement. Keeping this in mind, we tried to gain an understanding of the changes in the number of exceedances of 50 $\mu g/m^3$ for daily averaged PM_{10} concentrations using the Europe-wide emissions from 1995 and 2010.

First, we calculated the daily averages of total PM_{10} for 1995 and 2010 using the LOTOS model and 1995 meteorology. In order to estimate the changes in daily averaged PM_{10} concentrations due to emission reductions, we subtracted the difference between the concentrations calculated for 1995 and 2010 from the time-series measured at ten NAQMN stations for 1995.

Before discussing the results, we will comment on the assumptions made in the method used here. It must be realised that the most important non-modelled PM sources (sea salt and wind-blown crustal material, anthropogenic road dust) are unlikely to decrease substantially in the near future. This means they do not affect the results as they are included in the measured time-series. Secondly, the contributions from outside the model domain are small (annual average estimated to be about 0.9 $\mu g/m^3$; see Section 2.4) and changes in these in the near future therefore not very important. Finally, the contribution of the primary fraction to PM_{10} is relatively small, so errors in the PM emission inventories will not dominate the overall results. We therefore assume that the remainder of PM_{10}, i.e. SIA that can be modelled fairly well, will be the main determinant of the changes in PM_{10} concentrations in the near future.

The expected PM_{10} levels for 2010 in the Netherlands are studied in more detail in Section 6.6 using OPS. That calculation showed a nationwide decrease in PM_{10} of 5.4 $\mu g/m^3$ between 1995 and 2010. The decrease obtained here with LOTOS is 6.7 $\mu g/m^3$.

Since we are interested in annual averages and numbers of exceedances, we have presented our results in the same way as for the measured concentrations in Figure 2.15 (subsection 2.3.2.2.). The number of exceedances is plotted as a function of the annual averages, using data from all monitoring stations (regional, urban background and street) between 1993 and 2001. The average relationship for these data and their 95% confidence lines are plotted in Figure 2.29, together with the results from our scenario model runs. The upper and lower 95% confidence limits include variations in meteorology, variations in emissions, variations due to site-specific source contributions and other variations. Notice the high sensitivity of the number of exceedances to small variations in the annual average.

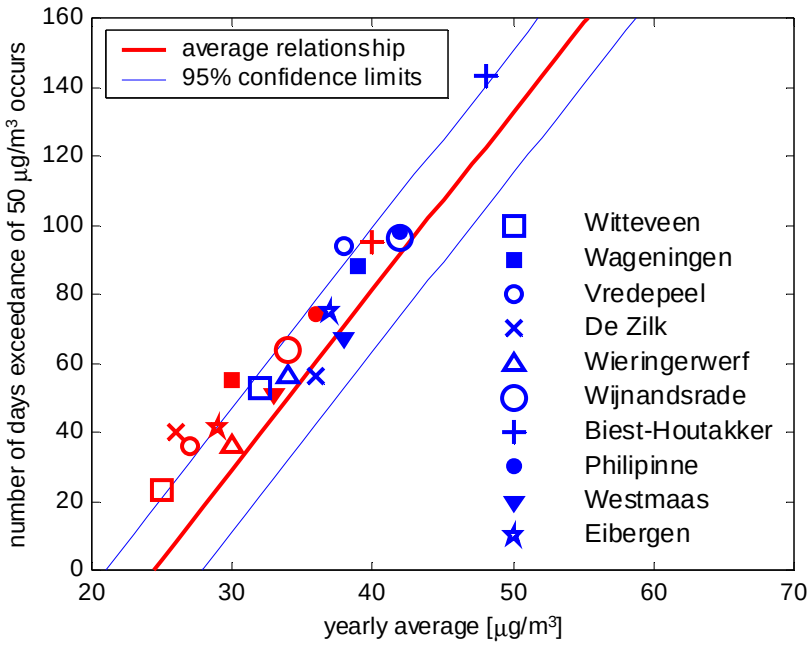


Figure 2.29 The number of days the limit value of 50 µg/m^3 is exceeded for daily average concentrations as a function of the annual average PM_{10} concentration. Average relationship for the Netherlands and the corresponding confidence lines from measurement data presented in Figure 2.15. The blue symbols correspond to the measurements from 1995 and the equivalent red symbols represent the 2010 scenario model runs extracted from the former data.

For most locations the model results fit the relationship well and the decrease in daily averages is more or less linear since the correlation between PM_{10} scenario values and measurements is very high (e.g. $R^2 = 0.93$ for Vredepeel). The model results have, on average, a somewhat higher number of exceedances than expected from the average relationship obtained from measurements. This could be explained, for example, by emission changes in the assumptions made which have not been addressed. On the other hand, the model results may also reflect the non-linearity of the relationship which, due to scatter, is not revealed from the measurements. When the average linear relationship, as obtained from the measurements, is extrapolated towards a hypothetical annual average of 0 µg/m^3, the number of exceedances becomes negative, which of course is not realistic. In reality, the relationship will bend towards zero, which is actually in line with the model results.

It appears from Figure 2.29 that the assumptions made with regard to the scenario runs have been confirmed independently by both the model and the measurement results. On the basis of this, it is not possible to draw firm conclusions for the future evolution of the number of exceedances of 50 µg/m^3 in the Netherlands. Nevertheless, the agreement between measurements and model results in Figure 2.29 is promising. It suggests that for 2010 the limit values for annual averages (20 µg/m^3) and number of exceedances (7) will not be met, which agrees with the indication from the OPS results presented in Section 6.6.

2.6. Agreement of models and measurements

2.6.1. Non-modelled part of PM

A difference of approximately 20 µg/m^3 was noticed between the annual average PM_{10} measurements of approximately 40 µg/m^3 and the annual average concentrations modelled using a deterministic dispersion model in the Netherlands report on a study to close this gap (Bloemen *et al.*, 1998; Buringh *et al.*, 1998; Visser *et al.*, 2001). For this present study, measurements were performed over a period of one year at the six sites presented in Figure 2.30.
The following non-modelled sources were found:

- background PM of the northern hemisphere (2.4.1.);
- natural crustal contribution (2.4.2.) and the anthropogenic part of crustal material;
- sea salt (2.4.3.).

For both sea salt and crustal material estimates were presented in Visser *et al.* (2001). These estimates will be borrowed for this exercise of gap closure; both the high and the low estimates for sea salt will be used. For background PM and sulphate, use has been made of the report prepared by Weijers *et al.* (2000) (see also Section 2.4), as it concerns PM originating from outside the 2,000 x 2,000 km^2 area of the OPS/SIGMA model. Weijers *et al.* arrived at a contribution of 0.9 µg/m^3.

A further gap-closure exercise was performed for two different 'scenarios' involving the correction factor for losses of semi-volatiles in the FAG instrument: a uniform correction factor (1.3) and a site-specific correction factor (of 1.15 and 1.45 respectively).

Table 2.18 Measured and modelled annual average PM_{10} concentrations. Measurements taken using various instruments at six sampling sites. Error bounds represent 68% confidence limits. The high sea-salt estimate (based on -Na^+ estimates) is presented.

Annual average PM_{10} concentrations	1 Nijmegen (µg/m^3)	2 Rotterdam Overschie (µg/m^3)	3 Amsterdam Overtoom (µg/m^3)	4 Amsterdam Stadh. Kade (µg/m^3)	5 Vredepeel (µg/m^3)	6 De Zilk (µg/m^3)
Modelled OPS/SIGMA	20.8 ± 5.2	23.8 ± 6.0*	23.5 ± 5.9	23.5 ± 5.9*	20.8 ± 5.2	16.0 ± 4.0
Sea salt high (Dichote Na^+)	5.2 ± 0.3	7.5 ± 0.6	7.1 ± 0.6	7.1 ± 0.6	6.2 ± 1.0	7.8 ± 0.6
N. hem. backgr.	0.9	0.9	0.9	0.9	0.9	0.9
Crustal estimate	5.7 ± 0.6	4.3 ± 0.5	2.8 ± 0.2	6.3 ± 0.5	4.4 ± 0.5	1.7 ± 0.3
Summed total high	32.4 ± 5.3	36.5 ± 6.1*	34.3 ± 6.0	37.8 ± 6.0*	32.2 ± 5.3	26.4 ± 4.0
Corrected FAG	31.1 ± 1.9	38.7 ± 2.0	34.3 ± 1.9	33.9 ± 1.9	34.5 ± 1.8	31.1 ± 1.6
Fraction model/measured	1.04 ± 0.23	0.94 ± 0.21	1.00 ± 0.23	1.12 ± 0.25	0.94 ± 0.20	0.85 ± 0.17

2.6.1.1. Uniform correction factors

Table 2.18 presents measured and modelled annual average PM_{10} concentrations that may result in 'gap' closure; the FAG correction factor used is 1.3. The Table shows that the average ratio [modelled / measured], more popularly known as the 'gap', is closed adequately. Generally speaking, averaged over all six different sites, the average gap now is **0.98 ± 0.20** with the high estimate for sea salt and **0.90 ± 0.20** with the low estimate (not presented; see Visser *et al.*, 2001).

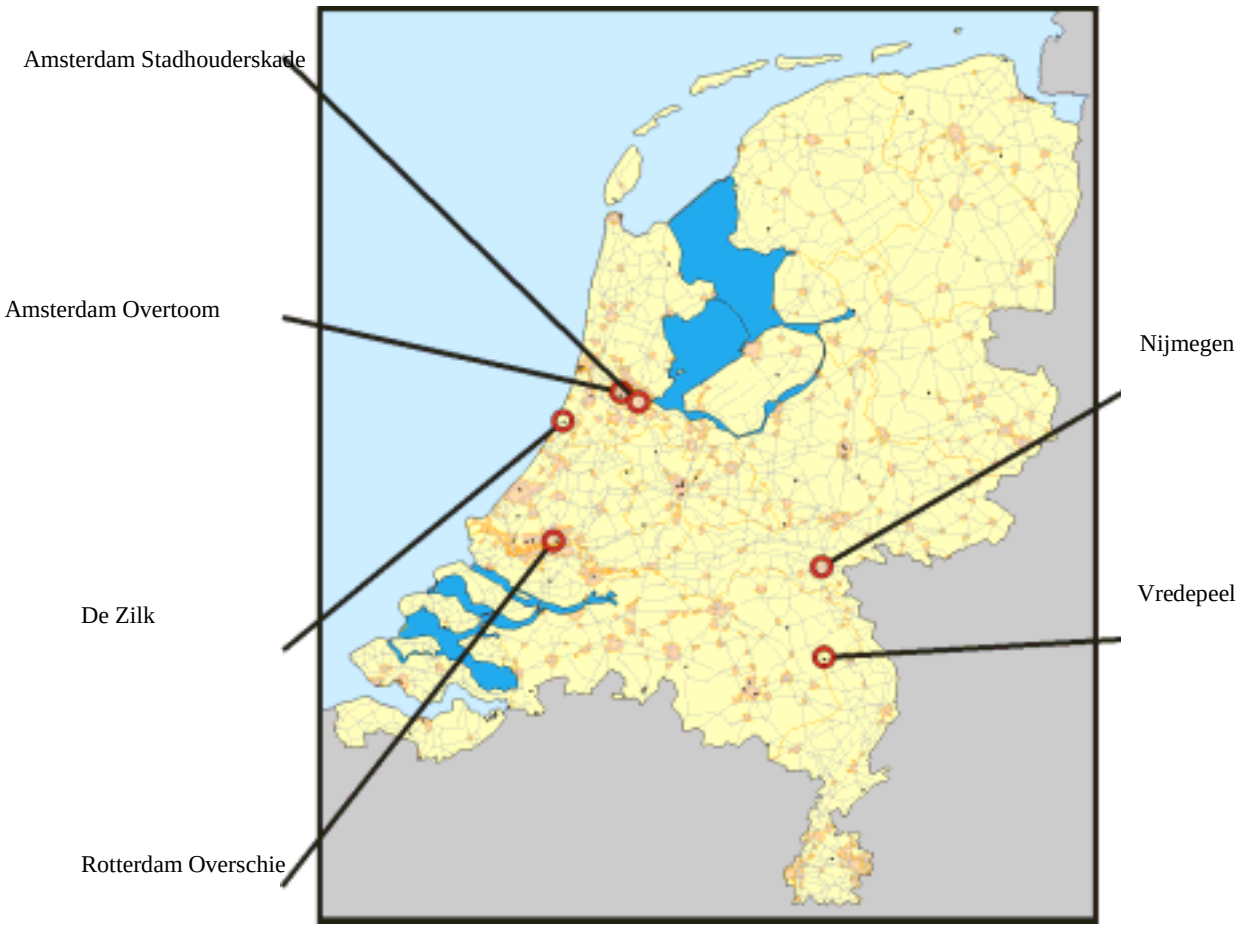


Figure 2.30 The six measurement sites from Visser et al., 2001.

When the uncertainty ranges are taken into account, the previously observed 'gap' can be considered to have been closed adequately. This holds for both high and low estimates. A test for the ratio of modelled to measured to be equal to 1.00 would require 2-σ bounds around the estimated ratio (assuming $\alpha = 0.05$). If such bounds are applied to the ratios in Table 2.18, we accept the null hypothesis of ratios being equal to 1.00 in all cases.

This result can be called an adequate gap closure when the fact that a number of elements (e.g. phosphorus) and components (e.g. carbonate) were not measured

during the project is also taken into account. The influence of carbonaceous material with diameters larger than 2.5 µm was not included, either. Water was left out of this analysis, as were secondary organic aerosols (SOA). The corrected model values do not differ from the measurements when tested with an $\alpha = 0.05$.

a. Ammonium nitrate measurements at a regional station

The value of the correction factor at a regional station was investigated by studying measurements of PM_{10} and ammonium nitrate at the RIVM's NAQMN station Wieringerwerf. The results of the absolute correction, 24-hour PM_{10} measurements multiplied by 0.3, were compared with the daily ammonium nitrate measurements performed by denuder/filter pack sampling. The results for 1999 and 2000 are presented in Figure 2.31.

*Figure 2.31 Daily ammonium nitrate measurements (denuder/filter pack) and daily absolute correction of PM_{10} (PM10corr = 24-h PM_{10} * 0.3) for 1999 and 2000 in Wieringerwerf (NAQMN #538).*

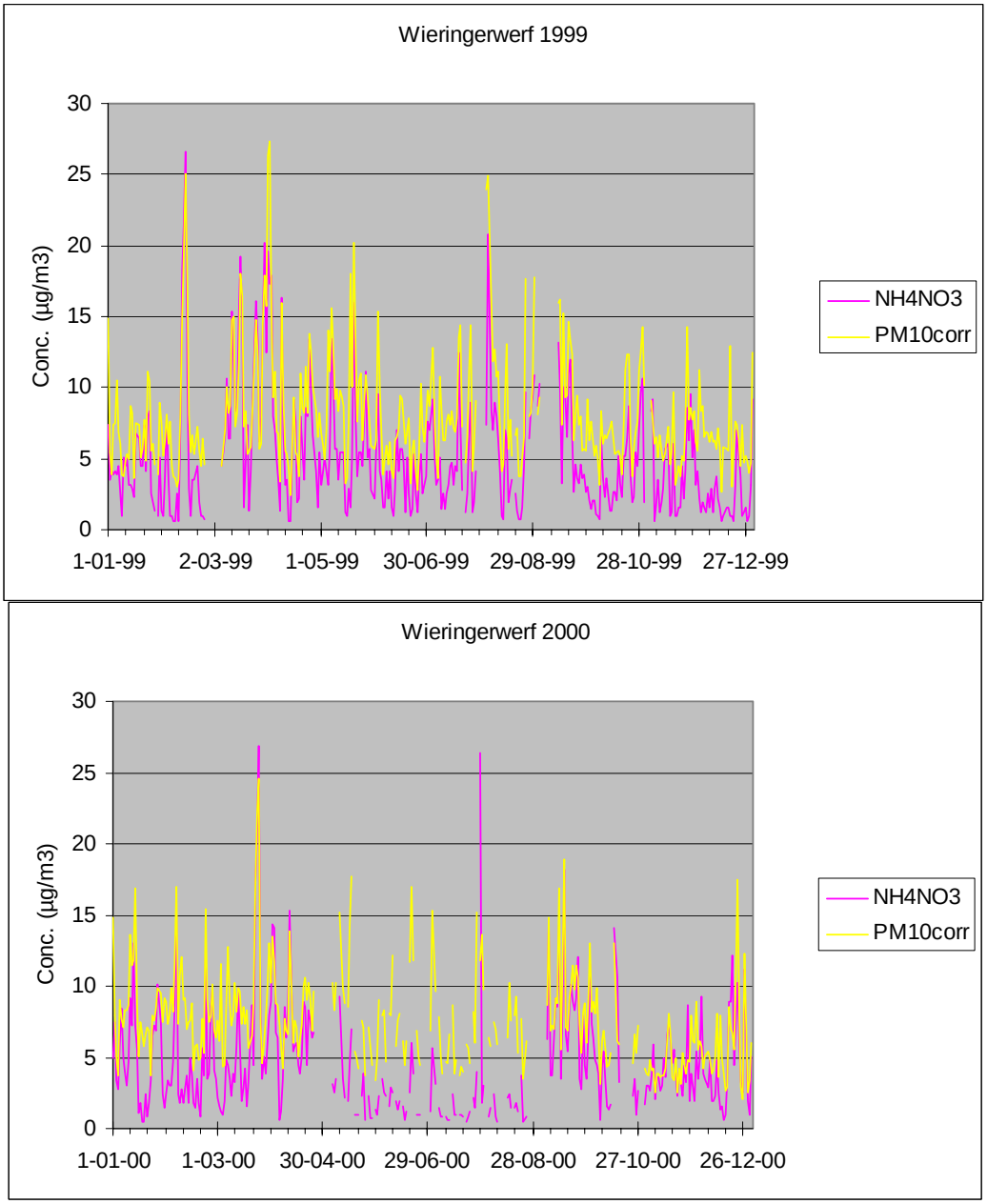
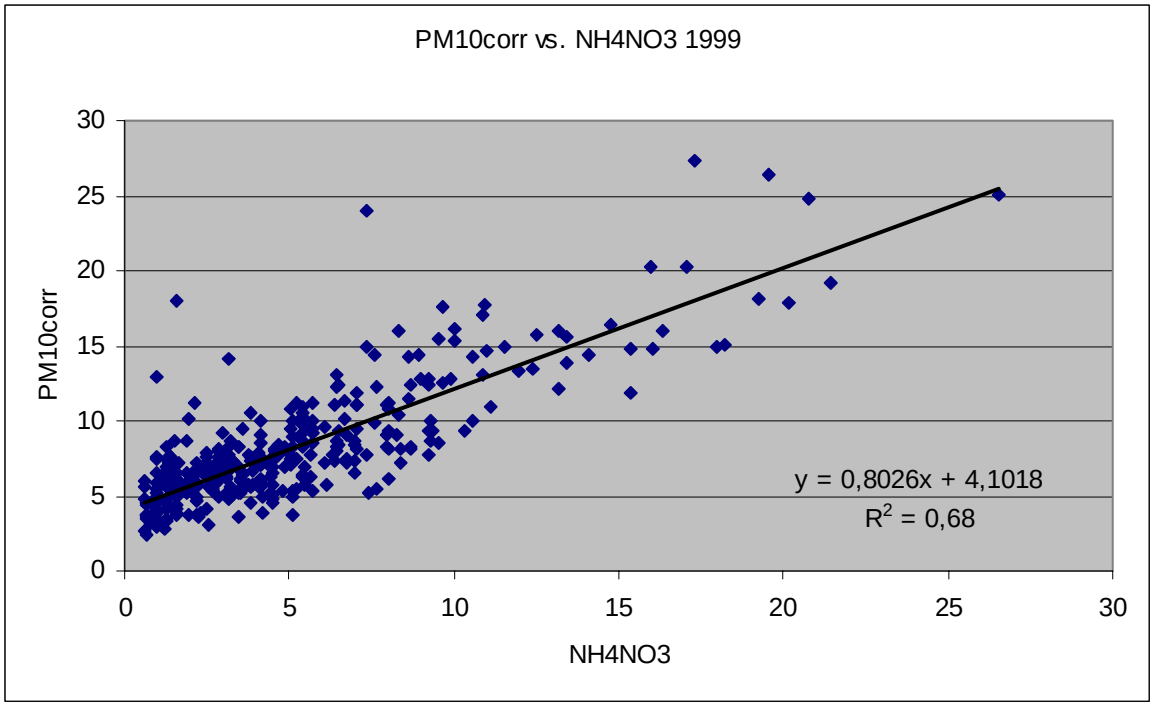
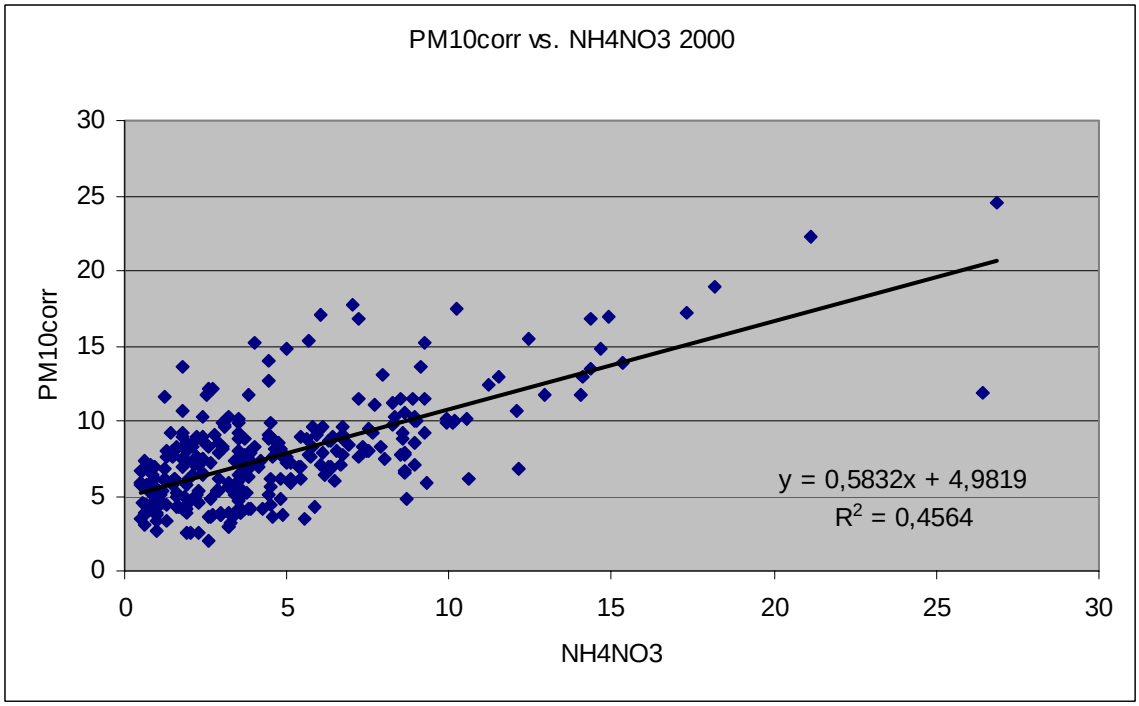


Figure 2.31 indicates good correlation between the ammonium nitrate concentrations and the factor of 1.3 used to correct PM_{10} for losses in the continuous PM_{10} monitors. This is further illustrated by showing the scatter plots of the same data in Figure 2.32.

*Figure 2.32 Scatter plot of daily ammonium nitrate measurements (denuder/filter pack) and daily absolute correction of PM_{10} ($PM10corr$ = 24-h PM_{10} * 0.3) for 1999 and 2000 in Wieringerwerf (NAQMN # 538).*





The scatter plots indicate that the correction of 1.3 at Wieringerwerf is approximately 4–5 µg/m3 too low in 1999 and 2000 to correct adequately for losses other than those of ammonium nitrate. As most likely components for these other losses organic compounds are suggested, but more detailed and chemically specified monitoring is advised before this question can be answered fully.

In order to investigate whether the correction factor of 1.3 on a daily basis might result in exceedance of the daily European PM_{10} standard, the sum of the *uncorrected* PM_{10} and ammonium nitrate measurements were computed. These values were then compared with PM_{10} measurements corrected by a factor of 1.3. The results are presented in Table 2.19.

Table 2.19 Number of days with exceedance of the 24-hour EU standard of 50 µg/m³ for PM_{10} with different corrections for missing semi-volatiles.

	1999	2000
Number of daily results	335	306
Ammonium nitrate in µg/m³	5.1	4.6
PM_{10} corrected by 1.3 in µg/m³	33	31
Uncorrected PM_{10} + ammonium nitrate in µg/m³	30	28
Number of days PM_{10} > 50 µg/m³	Days	Days
PM_{10} corrected by 1.3	46	25
Uncorrected PM_{10} + ammonium nitrate	39*	19*

*not corrected for semi-volatiles other than ammonium nitrate

Table 2.19 shows that annual average ammonium nitrate concentrations measured by denuder/filter pack sampling in 1999 and 2000 were 5.1 and 4.6 µg/m³ respectively. (The standard measurements taken by LVS in the NAQMN produced 5.2 and 2.8 µg/m³ respectively as an annual average for 1999 and 2000 if all measured nitrate had been ammonium nitrate.) These values are in line with the model calculations shown in Figure 2.1.

In Wieringerwerf (#538) the annual average of PM_{10} did not exceed the European annual standard of 40 µg/m³ in 1999 or 2000. Table 2.19 also illustrates that the EU daily PM_{10} standard (35 exceedances of 50 µg/m³) was exceeded in 1999 but not in 2000. Use of the factor of 1.3 results in a higher number of exceedances in Wieringerwerf than accounted for merely by correcting for losses of ammonium nitrate. Figure 2.31 shows that without any ammonium nitrate, there still is an unexplained loss of 4–5 µg/m³ on an annual average basis.
Hence, the previous assertion that the factor of 1.3 is not adequate for daily correction has *not* been confirmed by the experimental data collected at Wieringerwerf. How this works out for an urban background or street station still has to be established, however. Daily concentrations of ammonium nitrate may vary considerably over the Netherlands due to its dynamic equilibrium with nitric acid and ammonium. Further

research into the applicability and value of the correction factor for losses of semi-volatiles is therefore recommended.

2.6.1.2. Site-specific correction factors

To correct the FAG, site-specific correction factors were established for the Netherlands. These factors are 1.15 for the regional sites and 1.45 for urban sites. When these site-specific correction factors are used, a slightly different picture emerges from the previous one in Table 2.18.

The average ratio of modelled to measured is to a large extent closed. The average gap is **0.95 ± 0.20** with the high sea-salt estimate and **0.87 ± 0.20** with the low estimate. This also results in 'gap' closure, specifically when the uncertainty limits are taken into account.

There is a slight preference for the site-specific corrections, as the average differences between the modelled and measured values are not quite so extreme.

2.6.1.3. Handling the non-modelled fraction of PM

a. Introduction

Not all constituents of PM are in the emission database in the Netherlands, so it is impossible to model this missing fraction. The SIA contributed by the precursors of nitrate, sulphate and ammonium and the primary anthropogenic PM are currently calculated and summed. To this sum of modelled PM a 'difference' map of non-modelled PM is added for 5 x 5 km^2 grid cells.

This section addresses the quantification of the non-modelled fraction of $PM_{2.5}$ and $PM_{10.}$ Two approaches were used to produce a 'difference map', resulting in a range for the non-modelled fraction:

- a correction factor of 1.33 for the measurements;
- a correction factor that depends on the amount of ammonium nitrate and organic compounds present.

In both approaches it is assumed that the contribution of the non-modelled part is constant in time, though in some respects a number of factors may be time-dependent, e.g.:

- The reaction of chloride from sea salt with acidic components in the air (HNO_3, H_2SO_4), with the result that chloride can vaporise as HCl and lower the contribution from sea salt. Measurements in the Netherlands indicate that on an annual average basis, chloride depletion currently occurs at a rate of approximately 1.5 to 2 $\mu g/m^3$. Additional measures to reduce NO_x and SO_2 emissions could result in a lower concentration of acidic gases in the air, resulting in a lower chloride deficit and, as a consequence, a possible increase in the PM concentration.
- Carbonate-containing soil constituents (e.g. $CaCO_3$) will react with acidic components in the air (HNO_3, H_2SO_4), releasing the carbonate part of the particulate into the air as CO_2 and H_2O. The significance of this reaction has not been estimated.

- A reduction in anthropogenic VOC emission is foreseen in Europe, resulting in a reduction in the contribution of secondary organic aerosols (SOA) to PM_{10} and $PM_{2.5}$.
- Volatile organic carbon (VOC) is adsorbed by the filters and as a consequence increases the mass. Changes in VOC emission will change the measured mass (see also Turpin *et al.*, 2000).
- Part of the soil/crustal material is re-suspended by factors such as wind erosion or emitted by volcanic eruptions, another part is the result of agricultural activities. Recent literature suggests the contribution of agriculture to be important. Changes in agricultural activities (e.g. reform of agricultural practices) or the amount of land in use by farmers might lead to a systematic change of the assumed contribution.

b. PM_{10}

Constant correction factor

The calculated differences between measurements and model calculations in the period 1994–1999, with a systematic correction of the measurement (using a factor of 1.33), is presented in Figure 2.33.

On the coast of the Netherlands and near the cities of Rotterdam, Utrecht and Amsterdam, the non-modelled part appears to be somewhat lower than expected. On the other hand, the contribution in the south of the Netherlands appears to be unbalanced; somewhat on the high side in the province of North Brabant and somewhat on the low side in Limburg. This is a sign that this first approach does not produce a completely satisfactory 'difference map'.

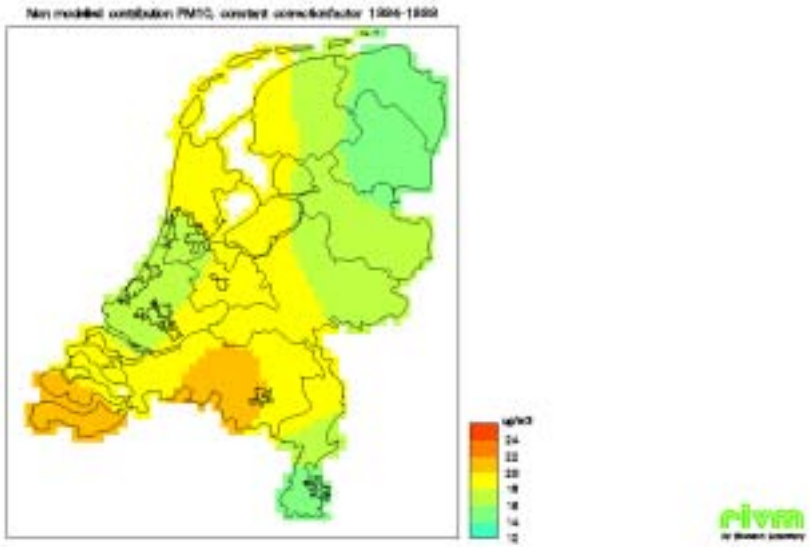


Figure 2.33 'Difference' map of non-modelled part, constant correction factor (1.33)

Constituent-dependent correction factor

In the second approach, a constituent-dependent amount is added to the measurements of the β-attenuation monitor (FAG). The hypothesis behind this approach assumes that the correction factor is a net outcome of gains and losses of material in the FAG (Putten *et al.*, 2001a,b,c).

The hypothesis is that some or all ammonium nitrate and semi-volatiles are lost and that some water may be absorbed. Values for ammonium and nitrate are available at some of the PM_{10} monitoring sites. For sites where ammonium and nitrate or BS measurements were not available, interpolated values were used. The following hypotheses were used to calculate the non-modelled part:

- An average loss of 75% of ammonium nitrate.
- To account for the loss of semi-volatile organics, 35% of the BS concentrations
- Absorption of water: a constant, negative correction factor of 1 $\mu g/m^3$ of water absorbed and water absorbed to non-volatised ammonium nitrate.

This results in an averaged correction factor of:

- 1.15–1.25 for the regional stations,
- 1.21–1.35 for the urban background and;
- 1.32–1.53 for the street stations.

The correction factor based on the considerations above showed a trend in time. In the period 1994–2001 the correction factor became smaller. For regional stations it decreased from 1.25 to 1.15, for urban background stations from 1.35 to 1.21 and for street stations from 1.53 to 1.32 (see Table 2.20).
The thus corrected measurements were compared with the values modelled. The average differences between the modelled and the corrected measured concentration was between 14 and 18 $\mu g/m^3$ (average 15 $\mu g/m^3$) on the regional scale. Compared with the approach using a constant correction factor (1.33), the calculated contribution of the non-modelled part decreased by an average of 3 $\mu g/m^3$ for the regional stations, 1 $\mu g/m^3$ for the urban background stations and 0.8 $\mu g/m^3$ for the street stations; see Figure 2.34.

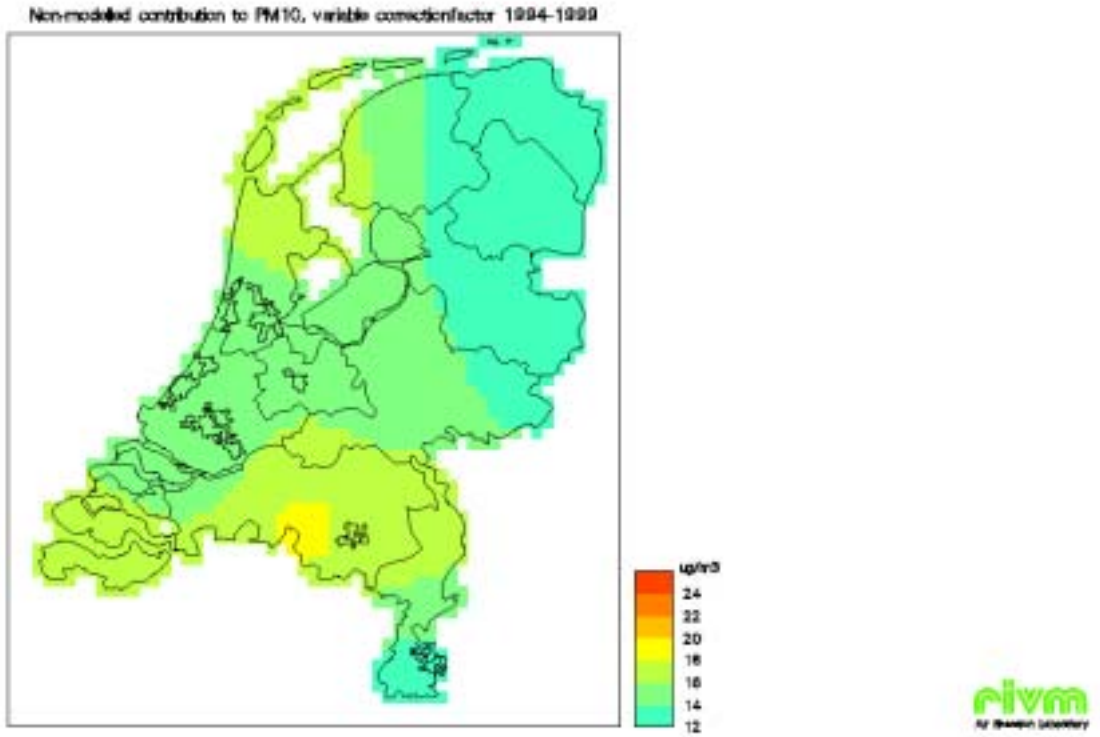


Figure 2.34 'Difference' map of the non-modelled part, variable correction factor

Table 2.20 Comparison between measurements from the Dutch National Air Quality Network and OPS calculations using a constituent-dependent correction factor.

	NAQN, constituent-dependent correction factor								OPS calculations, actual meteo						
year	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**Average**
rural, average correction factor:	*1,25*	*1,23*	*1,24*	*1,21*	*1,20*	*1,20*	*1,16*	*1,15*							**Difference**
Vredepeel	48	36	40	38	34	32	30	30	21	22	24	24		16	**18**
Wijnandsrade	37	39	37	36	32	28	27	27	21	23	26	25		19	**13**
Houtakker	45	44	44	42	37	34	30	31	20	22	25	25		16	**20**
Braakman	38	38	43	40	37	30	30	28	19	20	24	25		15	**17**
Westmaas	34	36	40	37	33	31	30	28	19	21	26	26		16	**14**
De Zilk	30	34	39	34	31	26	27	25	17	18	22	22		15	**14**
Wieringenwerf	32	31	40	30	28	29	26	21	14	15	18	18		11	**17**
Eibergen	34	34	39	35	29	26	24	24	20	21	24	23		15	**13**
Wageningen	39	36	40	40	33	31	30	28	21	22	25	25		16	**15**
Witteveen	30	28	32	28	24	23	27		16	16	20	19		11	**12**
city, average correction factor:	*1,35*	*1,32*	*1,30*	*1,28*	*1,27*	*1,26*	*1,22*	*1,21*							
The Hague	41	41	46	39	41	39	32	32	21	23	27	27		19	**18**
Rotterdam	41	41	46	41	39	36	33	33	26	28	33	33		22	**13**
Amsterdam	38,1	41	48	37	37	34	30	27	24	25	29	29		20	**14**
street, av. correc.factor (av 5 stat.):	*1,53*	*1,45*	*1,43*	*1,41*	*1,41*	*1,42*	*1,37*	*1,32*							
Vlaardingen	40	42	48	43	39	35	31	30	26	27	32	32		22	**14**
Breukelen - highway	42	40	45	42	37	36	34	29	21	22	26	25		17	**19**
total	**37**	**36**	**39**	**36**	**32**	**29**	**28**	**27**	**19**	**20**	**23**	**23**		**15**	**15**
regional	***37***	***36***	***39***	***36***	***32***	***29***	***28***	***27***	***19***	***20***	***23***	***23***		***15***	***15***
city	***40***	***41***	***47***	***39***	***39***	***36***	***32***	***31***	***24***	***25***	***29***	***30***		***20***	***15***
street	***41***	***41***	***47***	***42***	***38***	***36***	***33***	***29***	***23***	***24***	***29***	***29***		***19***	***16***

b. $PM_{2.5}$

Adequate measurements of $PM_{2.5}$ concentrations in the Netherlands are not yet available, so it is not possible to derive a similar 'difference' map for non-modelled contributions of $PM_{2.5}$. However, it is possible to project what differences can be expected, assuming there is no crustal material or sea salt in $PM_{2.5}$.

Due to the predominant influence that 'correction' of PM levels has on compliance with EU standards, a research programme that addresses the issues of losses of certain fractions of PM and the influence of water on the resulting figures is essential. This research should also delve into the issues of seasonal and spatial influences, characterising and quantifying these factors in order to produce more reliable PM measurements in the future. The constructed 'difference' maps for $PM_{2.5}$ with a constant and a constituent-dependant correction factor are presented in Figures 2.35 and 2.36.

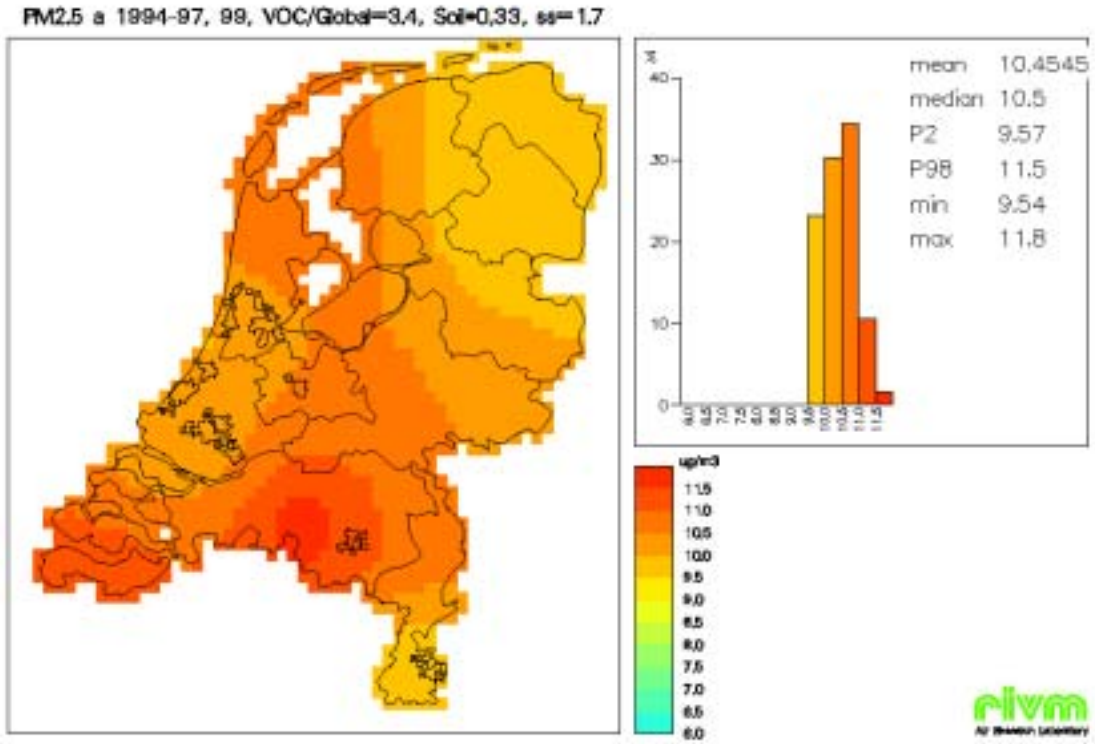


Figure 2.35 'Difference' map of non-modelled contribution to $PM_{2.5}$ with constant correction factor for PM_{10}

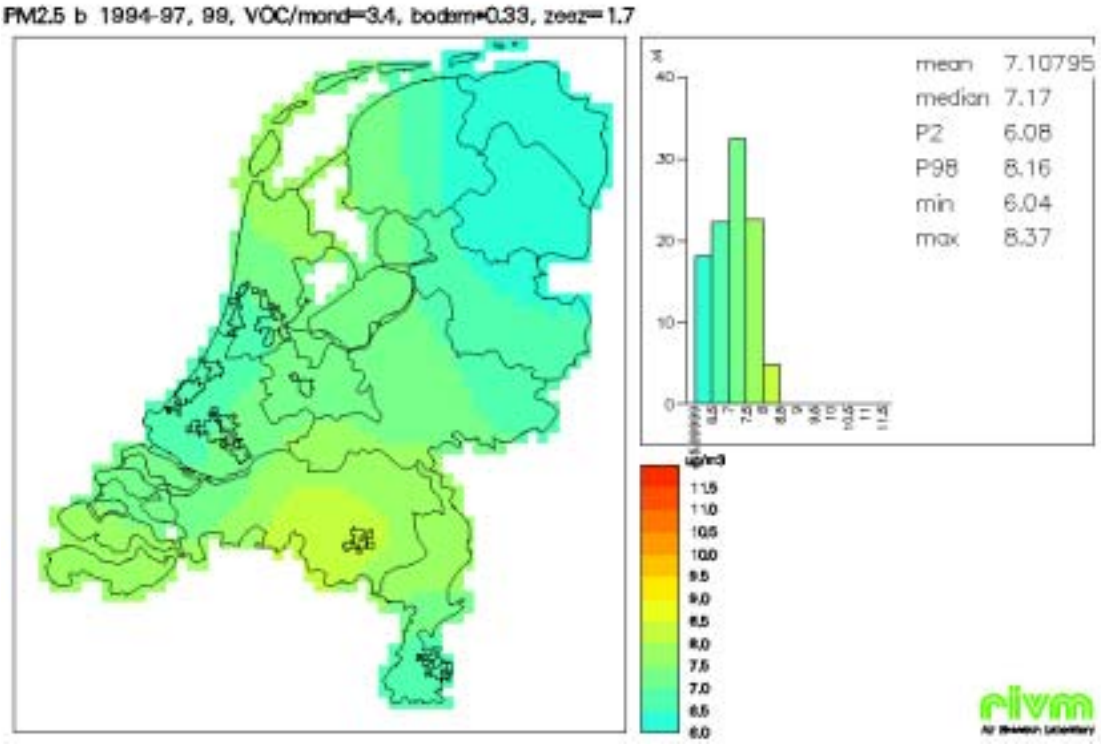


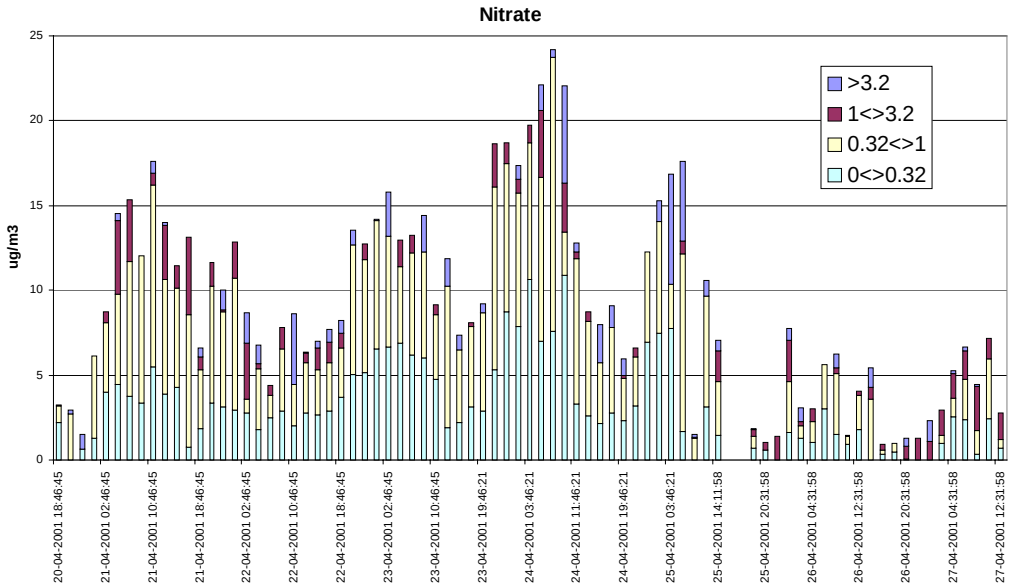
Figure 2.36 'Difference' map of non-modelled contribution to $PM_{2.5}$ with a constituent-dependent correction factor

2.6.2. Validation of short-term models

One goal within the NAP programme is the collection of experimental data (concentrations, size distribution) on background levels of SIA in order to validate short-term dispersion models (e.g. LOTOS and EUROS). For this reason, the summarised results of two measurement campaigns are reported here (Weijers *et al.*, 2002). These campaigns were organised at the site of the KNMI meteorological tower at Cabauw in the centre of the Netherlands. The instrument used was the Steam Jet Aerosol Collector (SJAC) (Slanina *et al.*, 2001); collected data were compared with the simulations of the LOTOS model.

2.6.2.1. Measurements of SIA

Time-series of the secondary aerosol measured in the first campaign are presented in Figure 2.37. The series indicate relatively high nitrate concentrations, i.e. 3–5 times higher than the sulphate concentrations. These enhanced ratios were measured at 200 m (during a relatively short period of one week), indicating their relevance for larger areas.



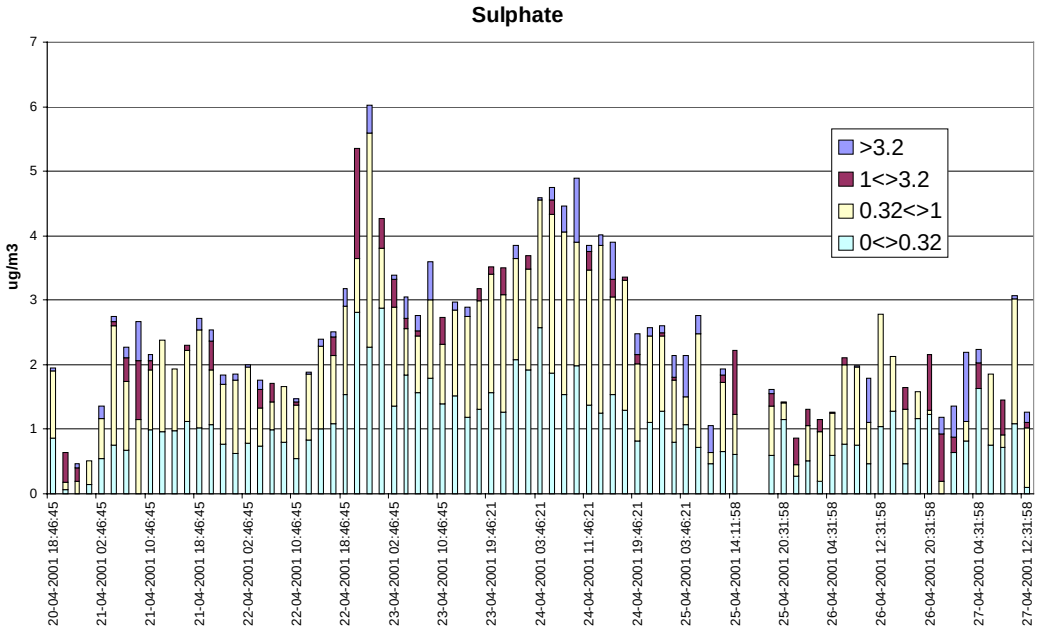


Figure 2.37 Temporal evolution of the nitrate and sulphate mass concentration during the April 2001 campaign; distribution over four size classes (in μm).

The differing synoptic situations prevailing during the first campaign render the classification of the trajectories into three 'classes' plausible (see Figures 2.38, 2.39 and 2.40):

1. arriving from the northeast on 21 and 22 April.
2. arriving from the southeast on 23 and 24 April.
3. originating in the Atlantic Ocean and later entering the Dutch zone from the southwest between 25 and 27April.

Figure 2.38 Trajectories for 22 April.

Figure 2.39 Trajectories for 23 April.

Figure 2.40 Trajectories for 26 and 27April.

The average concentrations of nitrate and sulphate belonging to these trajectories are shown in Figure 2.41 as a function of particle size. These figures confirm the earlier observation (see Figure 2.2) that concentrations of nitrate and sulphate are higher in air that has traversed areas east of the Netherlands (classes 1 and 2), compared with those from the southwesterly direction (class 3). The results in Figure 2.2 were measured at ground level at Petten and not at a height of 200 m.

Looking in more detail at Figure 2.41, the shape of the size distributions appears to be fairly similar. Sulphate is high and of comparable magnitude in the two smallest size fractions (i.e. below 1.0 µm), the highest nitrate concentration is systematically found in the size range between 0.32 and 1.0 µm.

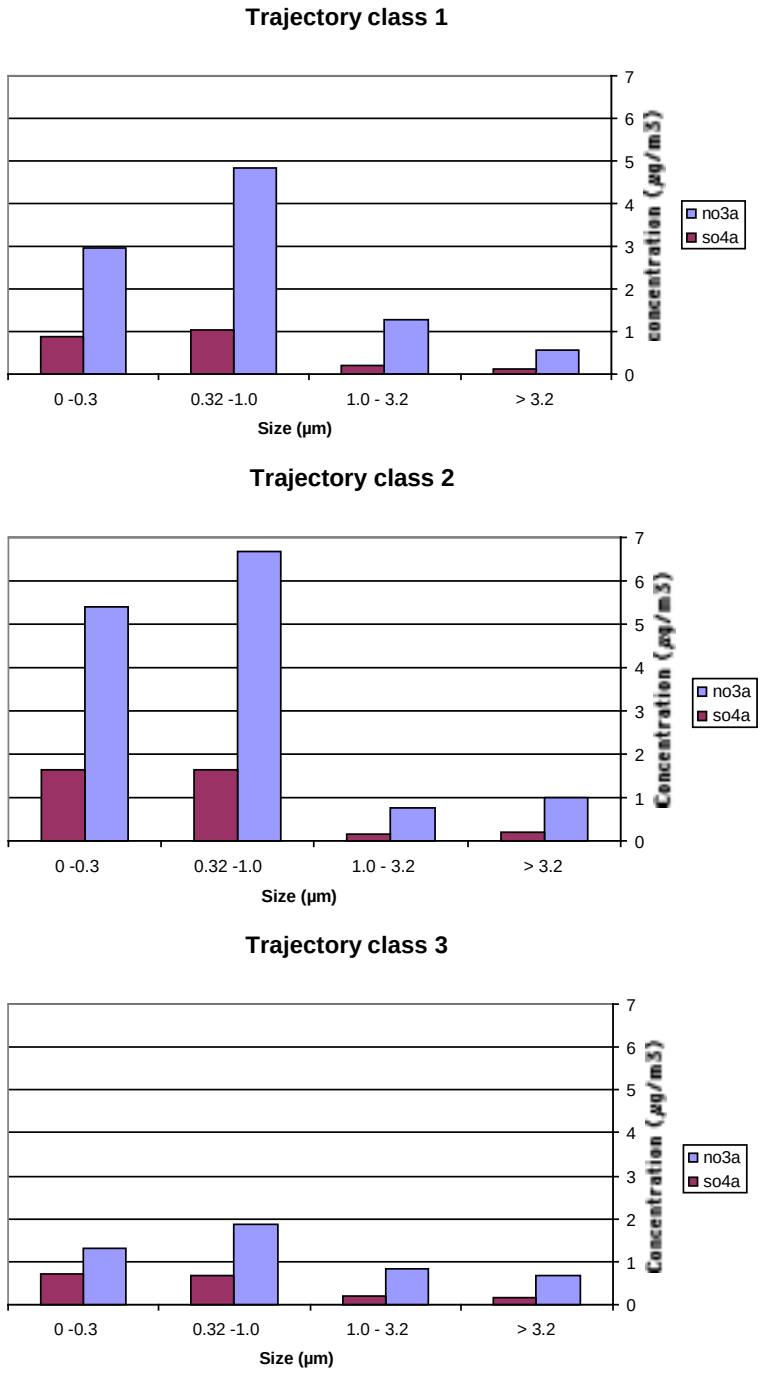
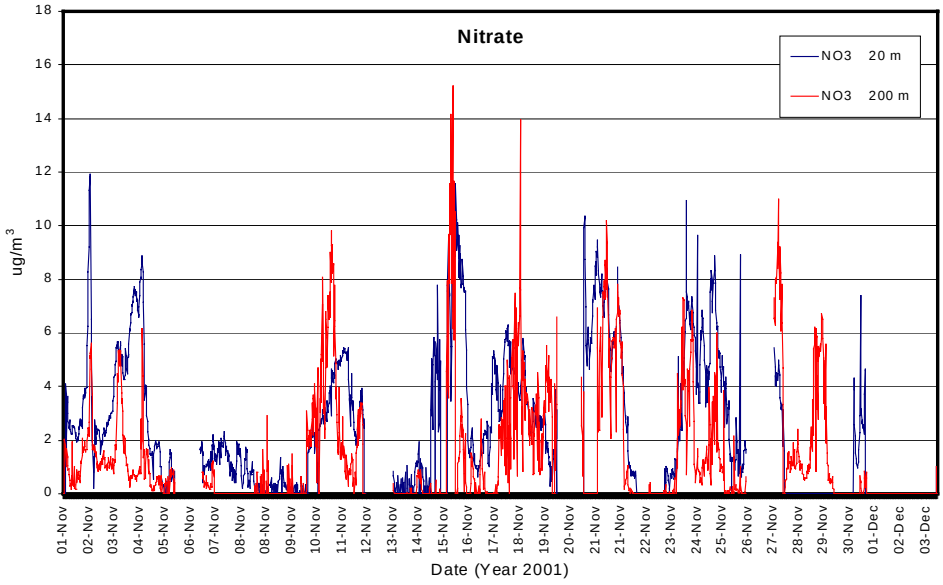


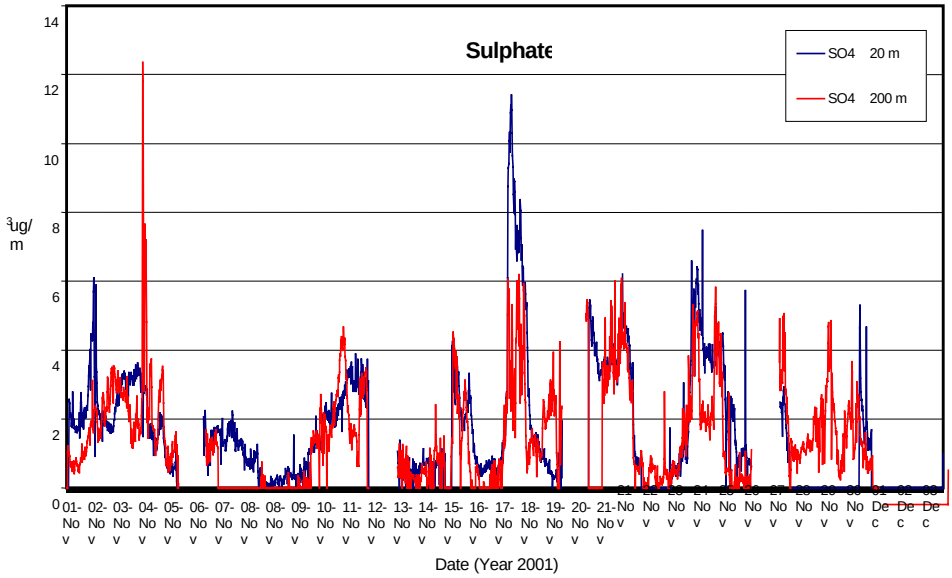
Figure 2.41 Size distribution of nitrate and sulphate as function of trajectory class.

The conclusion is that the size distribution of sulphate and nitrate as a function of transport direction does not vary substantially (except for some scaling factor). This is important for modelling practices, as an adequate generation of particle-size distributions in models still remains very difficult. These data suggest that, in the future, models like LOTOS may tabulate the size distribution simply as a function of the wind sector and simplify calculations.

Data collected during the November campaign are presented in Figure 2.42. In this case, the SIA were measured at two heights: 200 m and 20 m.



2.42.a



2.42.b

Figure 2.42 Concentrations of nitrate (a) and sulphate (b) as measured in November 2001 by the SJAC system; measurements were taken at two levels (200 m and 20 m)

In the November campaign, SIA was again dominated by nitrate, an observation that is valid at both 20 m and 200 m. However, the nitrate-to-sulphate ratios are now considerably smaller compared with the previous (April) campaign: 1.3 (at 200 m) and 1.6 (20 m). Trajectory analysis reveals that no continuous transport from the continent occurred during the November measurement period as did in the April campaign. Even at times when relatively high concentrations were measured, the sampled air had been considerable influenced by the Atlantic Ocean or the North Sea (see Figure 2.43). It merely confirms that high concentrations of nitrate in the

Netherlands are found in air that has traversed large distances over the European continent.

Correlation of the SIA measurements taken by an SJAC at Cabauw at 20 m and 200 m with similar NAQMN measurements of SIA in Bilthoven by LVS show high correlation coefficients for a spatial distance of approximately 25 km. Daily sulphate concentrations measured at 20 m at Cabauw and at ground level at Bilthoven correlate quite well: R=0.88. The correlation coefficient for sulphate at Cabauw at 20 m and at 200 m is only slightly less (0.80); these coefficients express the large-scale transport characteristic for sulphate. In the case of nitrate, the statistical relationship between Bilthoven and Cabauw at 20 m (R=0.86) is clearly stronger than the correlation between 20 m and 200 m (R=0.56). The difference between these coefficients indicates the presence of local and low-level emissions. It should also be noted that the correlation coefficients of nitrate are based on 16 days. A further investigation of the various influences on nitrate levels in the Netherlands appears warranted.

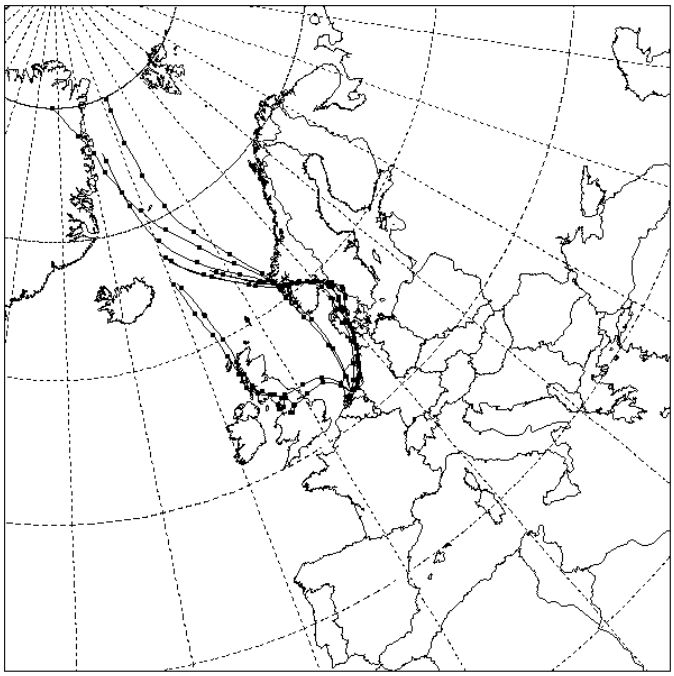
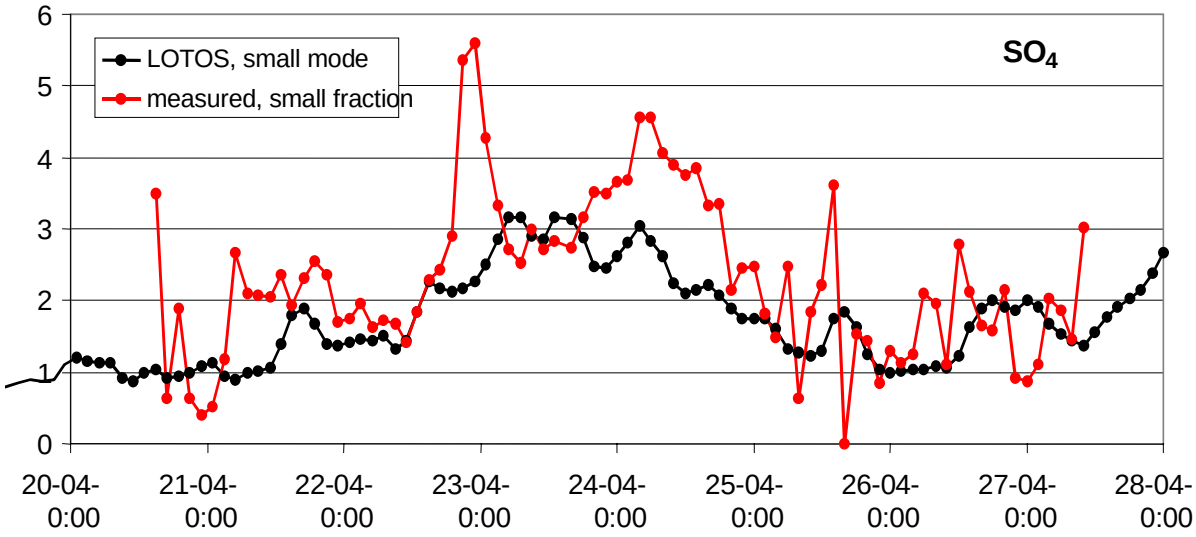


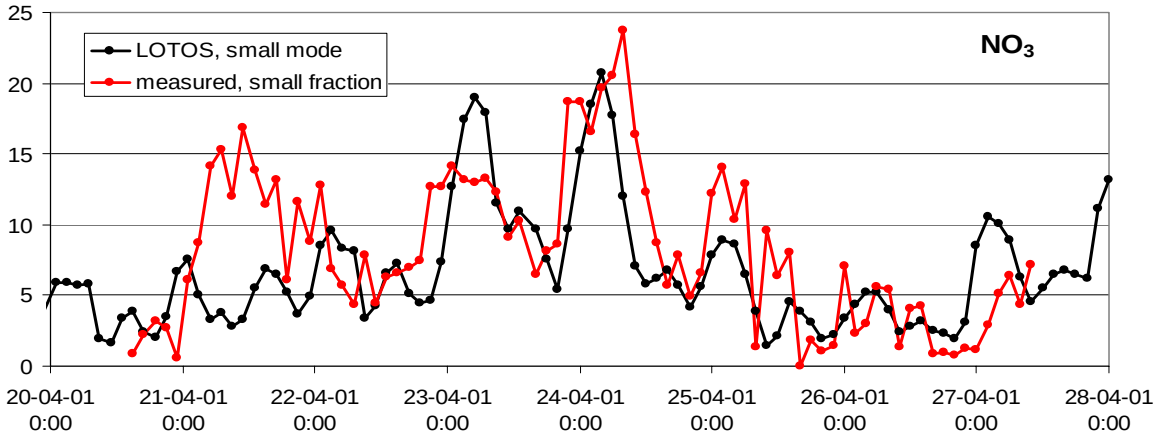
Figure 2.43 Trajectories for 14–16 November.

2.6.4.2. Modelling of SIA

Simulations of the April campaign show that LOTOS is capable of representing the measured SIA components at a high temporal resolution. With some exceptions, the elevated levels in the observations are also predicted by the model. The result of the modelling exercise is given in Figure 2.44.a and 2.44.b and compared with the measurements. In order to compare the modelled data with the measurements properly, only the fine mode data (<3.2 μm) are presented, as the formation of coarse mode sulphate and nitrate is not incorporated in the model.



2.44.a



2.44.b

Figure 2.44 Modelled and measured concentrations (µg/m^3) of sulphate (a) and nitrate (b) fraction <3.2 µm at 200 m at Cabauw, 20–28 April 2001

The high values for nitrate during the nights of 23 and 24 April are simulated quite well. The maximum of sulphate during the night of 24 April is also fairly well reproduced. The peak during 23 April, however, appears to be modelled at a different time in the model. The drop in concentrations during the morning of 24 April, associated with a change in meteorological conditions (a low-pressure system is moving in) is captured quite well.

LOTOS was also run for the November 2001 data. The simulation was started on 28 October, which is three days before the actual start of the measurements, in order to properly spin up the model. Concentrations of nitrate and sulphate measured by the SJAC were compared with the mass simulated by LOTOS. The measured data relate to the fine fraction ($PM_{2.5}$), which is comparable to the size of the two modes in LOTOS: the nucleation mode and the accumulation mode. It is noted that the SIA concentrations in the coarse fraction are usually low, as was already illustrated by the April measurements. Figure 2.45 gives a comparison for sulphate at a height of 20 m.

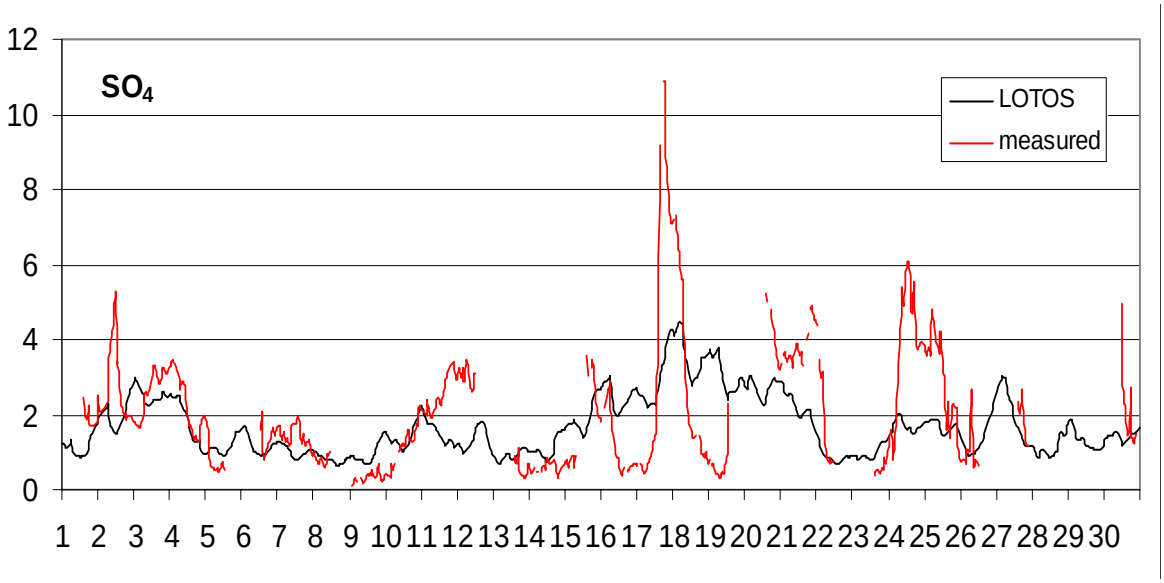


Figure 2.45 Measured and modelled concentrations of sulphate in µg/m^3 at Cabauw (20 m) for November 2001.

The measured and modelled concentrations for sulphate agree to some extent, in particular in the first half of the month. However, some improvement is needed. For example, it is apparent that around 18 November and, to a lesser extent, around 12 and 24 November observed peaks in the sulphate concentrations are not reproduced by the model. The peak values around 18 November are also present in the observations at 200 m, but with a much lower peak value of around 5 µg/m^3, which is about equal to the values at 20 m simulated by LOTOS. This suggests the existence of a large vertical gradient in the sulphate concentrations. In these cases LOTOS cannot reproduce the observed concentrations, because it gives an average over the mixing layer and hence will underestimate high concentrations relatively close to the ground.

The relative variance of the measured time series is largest for nitrate, implying that it will be more difficult for LOTOS to model the nitrate concentrations correctly. Nevertheless, the modelled and measured concentrations are in the same order of magnitude, as is shown in Figure 2.46.

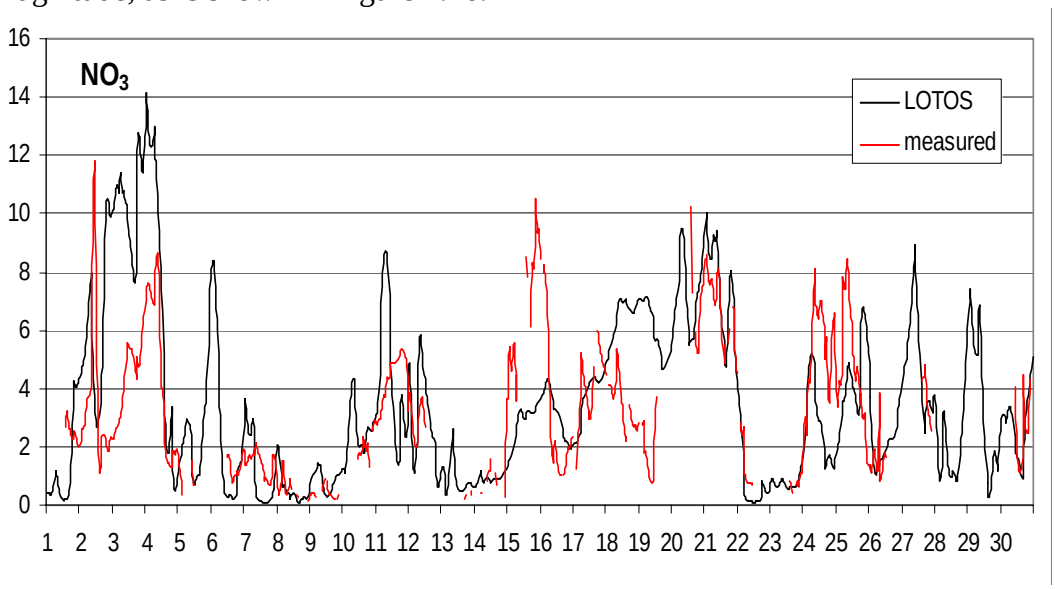


Figure 2.46 Measured and modelled concentrations of nitrate in μg/m^3 at Cabauw (20 m) for November 2001.

Moreover, the elevated levels in the calculated concentrations and measurements occur at the same time. In particular, the elevated nitrate levels on 20 and 21 November and the subsequent drop in the early morning of 22 November are quite nicely reproduced by the model. On two occasions there is a striking mismatch: on 3 and 4 and on 15 November. In both cases, the modelled sulphate levels agree quite well, so that differences must be caused by ammonium nitrate. Since agreement between the modelled ammonium levels (not shown) on 3 and 4 November is quite good, the differences in measured and simulated nitrate concentrations cannot be fully explained by the available data.

In conclusion, it can be stated that at present the modelled concentrations of nitrate and sulphate agree reasonably well with the measurements, as evidenced by the representation of the inorganic aerosol components at a high temporal resolution. However, further improvements are still needed.

2.6.3. Validation of long-term models

2.6.3.1. Contribution of traffic in urban areas

a. Introduction

Road traffic is a major source of PM in urban areas. Road traffic emissions are the result of car exhaust, wear on tyres and brake linings and, indirectly, the re-suspension of road dust, crustal and organic. In general, these PM emissions are characterised by their mass and size (PM_{10} and $PM_{2.5}$), by the number of particles or by chemical composition. Most research on PM emitted by road traffic has been related to PM_{10} and $PM_{2.5}$. Recently, more research has also focused on the number of particles, the largest numbers of which are smaller than 0.1 μm, the so-called 'ultrafine' particles (UF). Research has also been directed at the chemical composition of PM, with the emphasis on heavy metals, elemental carbon (EC) – which causes the 'blackness' of PM or black smoke (BS) – and primary organic compounds (OC) such as polycyclic aromatic hydrocarbons (PAH). This interest in various characteristics of PM is linked to the search for toxicological evidence to explain the epidemiologically found associations of PM in ambient air with health effects.

As part of the NAP, TNO has performed specific research on the contribution of traffic emissions from roads and inner-urban highways to PM air quality in urban areas, taking into account PM_{10}, $PM_{2.5}$, the number of particles, EC and OC.

b. Modelled traffic contribution

Figures 2.47 and 2.48 present the modelled ***contribution*** of traffic emissions to air quality in urban areas adjacent to a highway for the annual concentrations of PM_{10}, $PM_{2.5}$, and EC as a function of the distance to the axis of the highway.

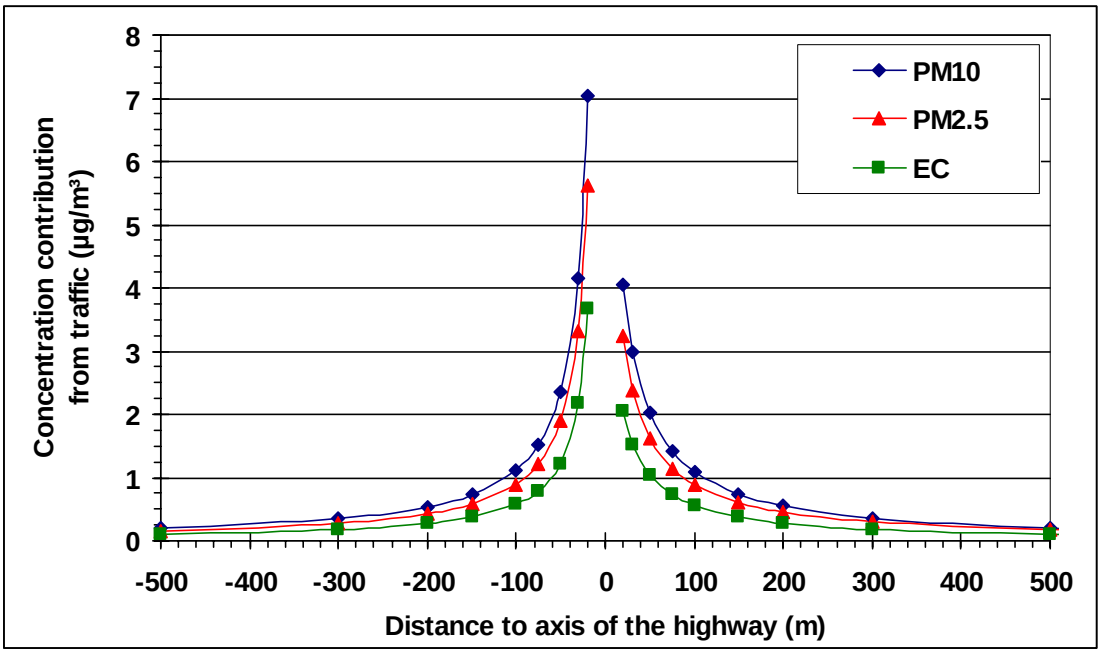


Figure 2.47 The contribution of the A13 highway traffic to annual concentrations of PM_{10}, $PM_{2.5}$ and EC ($\mu g/m^3$) in Rotterdam, Netherlands as a function of the distance to the axis of the highway.

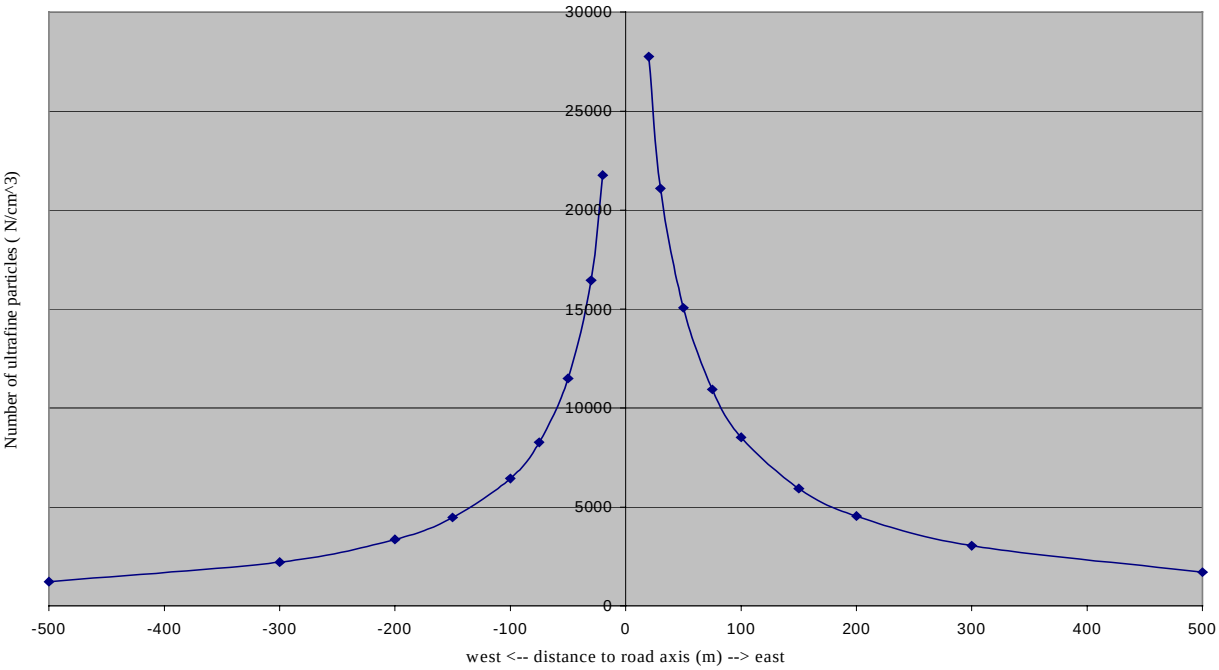


Figure 2.48 The contribution of the A13 highway traffic to annual concentrations of ultrafine particles (UF) as a function of the distance to the axis of the highway.

Figure 2.47 shows that the contribution of traffic to PM and EC levels depends to a great extent on the distance from the highway. At 50 m from the highway the

contribution is about 3 μg/m^3 for PM_{10} and $PM_{2.5}$ and about 1 μg/m^3 for EC. Figure 2.48 shows an analogous contribution of highway traffic to the air quality of ultrafine particles. At 50 m the UF contribution is about 20,000 particles per cm^3.

The modelled annual contribution from traffic at the location along Highway A1 is shown in Table 2.21.

Table 2.21 The modelled annual contribution of traffic to annual PM_{10}, $PM_{2.5}$, UF and EC in Rotterdam, Netherlands at 40 m from the axis of a highway.

	PM_{10} μg/m^3	$PM_{2.5}$ μg/m^3	UF N/cm^3	EC μg/m^3
Highway A1 (40 m)	3	2	20,000	1

c. *Measured traffic contribution*

Table 2.22 gives the annual concentrations of PM_{10}, $PM_{2.5}$, UF and EC at three locations in Rotterdam.

Table 2.22 Annual average measured concentrations for PM at locations A1 and A3 near a highway and in a street canyon (Pleinweg) in Rotterdam, Netherlands (2001).

	PM_{10} μg/m^3	$PM_{2.5}$ μg/m^3	UF N/cm^3	EC μg/m^3
Highway A1 (40 m)	31 ± 9	22 ± 6	50,000	3.1 ± 0.7
Highway A3	30 ± 9	24 ± 9	20,000	1.4 ± 0.3
Pleinweg	38 ± 4	27 ± 15	40,000	3.2 ± 0.7

The annual ***contribution*** from traffic emissions is given in Table 2.23.

Table 2.23 Annual measured contribution from traffic emissions to ambient air concentrations for PM at location A1 near a highway and in a street canyon (Pleinweg) in Rotterdam, Netherlands (2001).

	PM_{10} μg/m^3	$PM_{2.5}$ μg/m^3	UF N/cm^3	EC μg/m^3
Highway A1 (40 m)	1 ± 13	-2 ± 11	30 000	1.7 ± 1
Pleinweg	6 ± 10	3 ± 17	20 000	1.8 ± 1

The annual contribution is determined by taking location A3 as the urban background location and by subtracting the A3 concentrations from the values at locations A1 and Pleinweg. Location A3 is situated relatively close to highways, so these 'background' values may be relatively high. Hence, actual contributions can in practice be higher than estimated.

From the results in Table 2.23 we learn that the contribution of *local* highway traffic to PM_{10} and $PM_{2.5}$ air quality is not significant at A1. This is related to the high PM_{10} background and favourable atmospheric dispersion around highways. Consequently, it is difficult to assess the impact of *local* traffic emissions. By contrast, the contribution of inner-urban traffic to PM_{10} – especially the coarse fraction of $PM_{2.5-10}$ – in the Pleinweg street canyon is relatively high when compared with location A1 near the highway. It is noted that the PM levels in the street canyon are not statistically significant different from those of the highway. Therefore this finding has to be seen as an indication only, which needs a better experimental substantiation. However, these findings confirm research in the UK (DETR,1999), which concluded that the enclosure of air pollution in a street canyon results in relatively high concentrations of local coarse $PM_{2.5-10}$, mainly as a result of the re-suspension of road dust. Re-suspended particles are of a coarser size fraction than direct exhaust emissions.

From Table 2.23 it is concluded that for UF particles and EC, the contribution made by highway and inner-urban traffic to local air pollution is comparable. For the highway, the model computes a six times higher value (Table 2.21) than assessed in Table 2.23. The CAR model appears to overestimate the EC contribution, while the contribution for highways appears to be underestimated. Despite three times fewer vehicles per 24 hours, the enclosure of air pollution in a street canyon results in comparable pollution levels. It is concluded that EC and UF are better indicators for measuring the impact of *local* traffic on *local* levels of PM than mass-based indicators like PM_{10} and $PM_{2.5}$. More experimental data for EC and UF are required to further substantiate this conclusion.

The data in Table 2.22 and 2.23 illustrate that street canyons may have concentration levels of PM similar to urban areas near highways, even when traffic density is three times as low as on a highway. However, in general, the *composition* of PM may be quite different: *street canyons* have a higher ratio of coarse to fine particulates (e.g. re-suspended road dust). This initial finding (which requires verification by more experimental data) may contribute towards explaining different health effects near highways and inner-urban roads.

2.6.3.2. Contribution of a regional highway

The contribution of a regional highway to PM_{10} levels was estimated in a manner similar to the traffic contribution in urban areas by comparing the PM measurements at three NAQMN sites: Breukelen (#641), a street station situated right next to the A2 highway from Utrecht to Amsterdam, and the regional background stations in Westmaas (#437) and Wageningen (#724).
The daily variations are now considered in more detail for the street station Breukelen; see Figure 2.49. The average PM_{10} concentration for weekdays (Monday–Friday) is 37.7 $\mu g/m^3$ (± 1.7); for Saturdays and Sundays the average level is 35.5 $\mu g/m^3$ (± 1.5). Regional background levels in Westmaas and Wageningen for

Monday–Friday are 33.7 µg/m^3 (± 1.4); for Saturdays and Sundays the average regional background concentration of PM_{10} is 32.7 µg/m^3 (± 1.4).

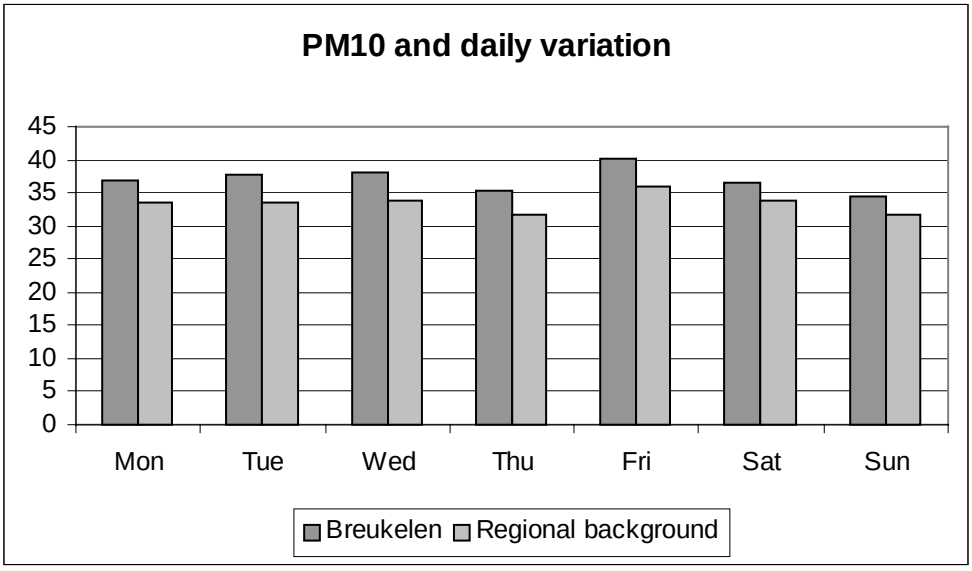


Figure 2.49 Average weekday PM_{10} concentration in µg/m^3 in 2000 for the street station Breukelen (Highway A2) and average of the regional stations Westmaas and Wageningen.

The CAR model was used to find out whether the observed PM_{10} contribution made by the highway could be explained by modelling. It should be noted that due to the uncertainties of both the measurements and the model, only a very general comparison is possible. Input for the model was the numbers of passing vehicles per 24 hours, the percentage of trucks and buses, distances between road and monitoring station, type of landscape, average speed of passing cars, and regional background concentration levels. Estimated traffic densities in the year 2000 on the A2 near Breukelen are presented in Table 2.24.

Table 2.24 Traffic densities in the year 2000 as measured on the A2 near Breukelen (Source: Directorate-General of Public Works and Water Management, AVV Transport Research Center); absolute number of vehicles averaged per 24 hours, and the relative contributions of the vehicle classes Cars, Light Duty Vehicles (LDV) and Trucks.

	Mon	Tues	Wed	Thurs	Fri	Sat	Sun
Total vehicles	71,486	74,215	74,467	75,837	76,643	55,532	56,053
Cars %	86	86	86	87	87	93	95
LDV %	8	8	8	8	8	6	4
Trucks %	6	6	6	5	5	1	1

The background concentration is estimated as the average from two regional PM_{10} and NO_x monitoring stations geographically nearest to the highway: Westmaas and Wageningen. The traffic contribution of PM_{10} and of NO_x at Breukelen in 2000 is estimated as the local concentration levels at the highway minus the background concentration. The results of the simulation using CAR and the measured concentrations are shown in Table 2.25. The contribution of NO_x concentrations at

Breukelen was calculated as a validation of the model assumptions, as the emission factors of NO_x are reasonably well-known, unlike the PM emissions.

Table 2.25 Measured traffic contribution (concentration at Breukelen minus background concentration) and contribution of NO_x and PM_{10} in 2000 calculated by the model CAR, by day of the week at Breukelen.

	NO_x ($\mu g/m^3$) Measured	NO_x ($\mu g/m^3$) Modelled	PM_{10} ($\mu g/m^3$) Measured	PM_{10} ($\mu g/m^3$) Modelled
Monday	109	120	3.3	1.5
Tuesday	105	124	4.3	1.5
Wednesday	102	125	4.2	1.6
Thursday	95	120	3.7	1.5
Friday	118	121	4.4	1.5
Saturday	78	65	2.9	0.9
Sunday	67	66	2.7	1.0
Mon–Fri avg.	106	122	4.0	1.5
Sat–Sun avg.	72	66	2.8	1.0

Table 2.25 shows the contribution of NO_x to have been estimated reasonably well. For weekdays, the CAR model calculated 116% of the measured contribution; on weekend days 92%. The ratio of NO_x concentrations in Breukelen on weekdays to that on weekends was also modelled adequately by CAR: the ratio is 0.69 (measured) versus 0.54 (modelled). This result also indicates that the quality of the parameterisation of dispersion in the CAR model is satisfactory and that it could be used in principle to calculate the particulate contribution made by highway traffic as well.

For PM_{10}, the ratio of weekday traffic contribution to weekend contribution is 0.70 for the measurements and 0.67 for the CAR model. This is a very fair result and shows that the relative results for PM_{10} are satisfactory. However, the absolute values differ. In absolute terms, the CAR model presents only about 40% of the measured PM_{10} contribution in Breukelen. This rather poor result can be attributed to a number of factors. A plausible cause of the difference between measured and modelled PM_{10} in Table 2.25 is re-suspended crustal material. The CAR model uses only primary PM_{10} emissions. The definition of traffic emissions does not take into account the contribution of re-suspended crustal material caused by vehicle-induced turbulence. According to Visser *et al.* (2001), this re-suspension is considerable. The annual average contribution of road dust re-suspended by passing traffic at a major road in Amsterdam amounted to 2–3.5 $\mu g/m^3$,while the local tailpipe contribution of EC + OC was 2.1 $\mu g/m^3$. This indicates that the indirect PM_{10} contribution made by re-suspension is probably of the same order as that of the direct traffic contribution. If this re-suspension of crustal material by traffic at the highway by Breukelen had been of the same order as that in Amsterdam, it would account for nearly all of the mass in the CAR model currently still missing. This considerable contribution of re-suspended material was also found by the studies conducted in Rotterdam by Spoelstra and Keuken (2002).

2.6.3.3. Modelled and measured SIA

The previously presented OPS model combined with national and European emission inventories results in annually averaged PM_{10} concentrations that seem only partially to match measured results from the National Air Quality Monitoring Network. Recent work by Visser *et al.* (2001) indicated that the previous differences between models and measurements could be adequately explained when the non-included natural sources, anthropogenic crustal material and the contribution of the northern hemisphere, were taken into account. This is elaborated in subsection 2.6.1.

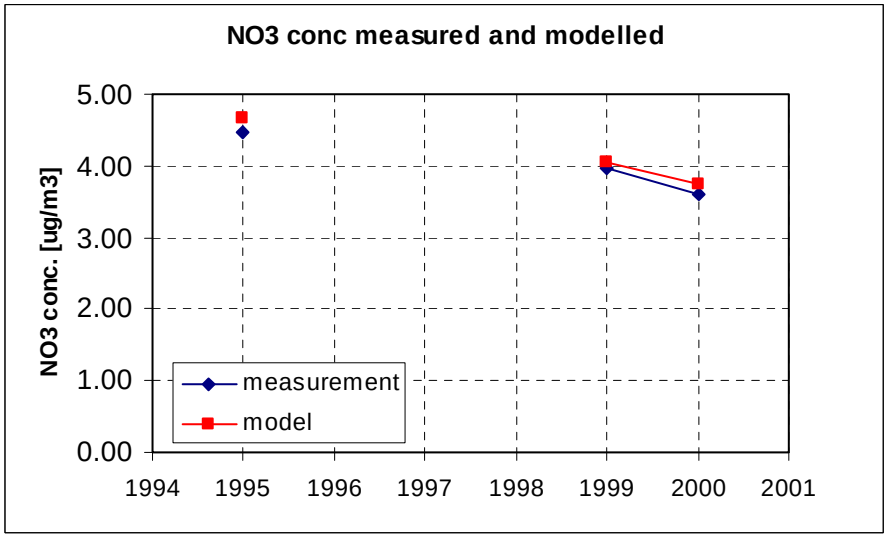


Figure 2.50 Measured and modelled concentrations of NO_3 aerosol. Modelled NO_3 does not include gaseous HNO_3.

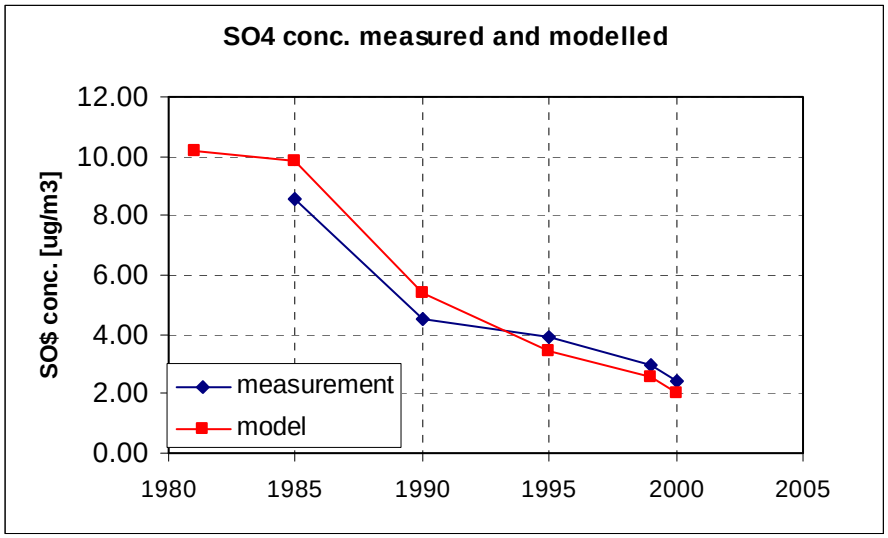


Figure 2.51 Measured and modelled concentrations of SO_4 aerosol

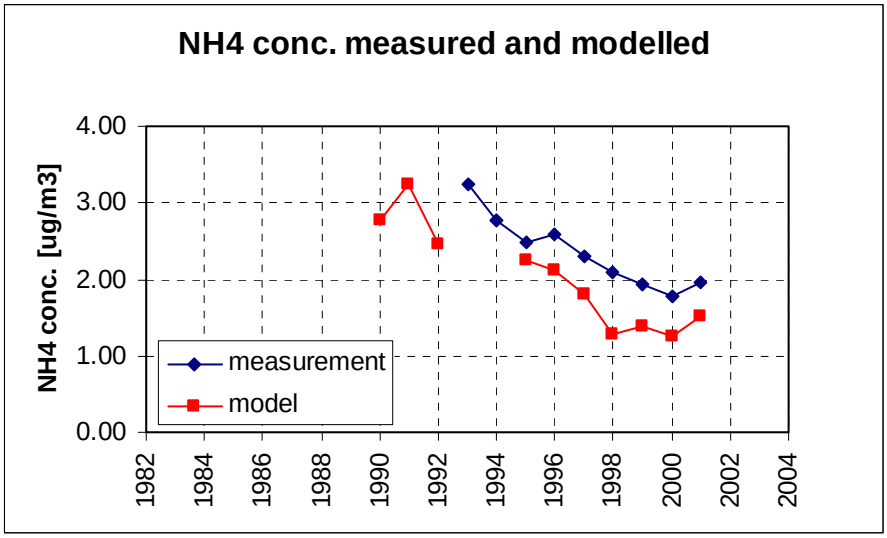


Figure 2.52 Measured and modelled concentrations of NH_4 aerosol.

The minor differences between the modelled SIA levels in Figures 2.50, 2.51 and 2.52 and the measurements in the NAQMN indicate that the model is capable of reproducing concentration trends and also the absolute level of SIA. Ammonium aerosol is somewhat underestimated due to the previously reported discrepancy between NH_3 measurements and emissions (Van Jaarsveld *et al.*, 2000). The difference is estimated at 0.4 $\mu g/m^3$ ammonium and this is used as a correction in the 1995 and 2010 results. The volatilisation of ammonium nitrate from filters probably also plays a role. The filter method used underestimates the 'true' concentrations of ambient ammonium nitrate, as has been explained in 2.2.1.1.

The performance of state-of-the-art modelling of ammonium nitrate in aerosol is treated in Metzger's PhD thesis (2001). Using thermodynamic descriptions of the dissociation of ammonium nitrate and the behaviour of aerosol/water mixtures, acceptable agreement is obtained with measurements at ECN, for example, in winter. In summer, though, the model predicts nitrate concentrations which are on average only 25% of the measured values. The obvious conclusion is that these models are able to predict wintertime nitrate levels adequately, but for summers produce results that are too low, especially at high temperatures (under Dutch circumstances).

Validations of modelled and measured levels of SIA indicate that the models do a good job as they produce similar results, and that the trends in time (over periods of decades) can also be modelled adequately.

2.7. Relevance of a stationary background monitor

In the past, the use of air quality data from a fixed background monitoring site as a representative measure for the population exposure has been criticised in the scientific literature. Some researchers asserted that ambient concentrations, particle and other, were only weakly correlated with personal exposure. This, however, is the point when the correlation between personal exposure and ambient concentrations is assessed at a single point in time over a population. On the other hand, when the personal concentrations over time are correlated with a time-series of ambient measurements,

this correlation becomes much larger. For a time-series study this is the kind of correlation that is relevant, and not the other, cross-sectional correlation.

It has been shown in studies in the Netherlands that the correlation in the variations of personal and outdoor mass concentrations of PM_{10}, and especially of $PM_{2.5}$, is reasonably high (Janssen *et al.*, 1998; Janssen *et al.*, 2000). This result is confirmed in other current time-series studies. For 10- to 12-year-old children and 50- to 70-year-old adults, the median correlation coefficient between outdoor and personal PM_{10} was R=0.63 and R=0.50 respectively. For particles with diameters smaller than 3 μm in a group of 10- to 12-year-old children, the median outdoor personal correlation was R=0.86 (Janssen *et al.*, 1997). For the elderly in the Netherlands these results indicate that variations in ambient PM outdoors are associated with approximately a quarter of their variations in personal PM. For children, these correlation coefficients indicate that variations in a central site monitor are associated with 40 to 75% of the variation in personal PM. Janssen *et al.* (2000) have shown that for $PM_{2.5}$ the median of the correlations between personal samples and the outdoor concentrations was R=0.79 in Amsterdam and R=0.76 in Helsinki. For absorption (as a marker for EC) the correlation coefficients were even higher: R=0.93 in Amsterdam and R=0.81 in Helsinki. Additionally, Sarnat has indicated that ambient gaseous pollution concentrations correlate well with personal $PM_{2.5}$ concentrations but not with personal gaseous pollutant concentrations (Sarnat *et al.*, 2001), which provides strong evidence that ambient gaseous concentrations act as surrogates for personal $PM_{2.5}$. Similarly, this might also be the case for other PM fractions, e.g. PM_{10}, but no such data were measured in the Sarnat study. Evidence is also beginning to emerge that ambient $PM_{2.5}$ measurements can be used to estimate population personal long-term exposures to $PM_{2.5}$ as well (Oglesby *et al.* 2000).

One has to realise that in a population the day-to-day correlation between the personal cloud (dusty habits, smoking) and ambient air pollution (PM) or meteorology is assumed to be very low to zero. This means that the health effects caused by such habits do not tend to have any association with the daily levels of ambient air pollution (PM). Consequently, in a time-series study these personal cloud effects are averaged out. Whatever the objections to the use of a stationary site monitor, a correlation coefficient of R=0.5 and higher indicates that data from a fixed site is representative for at least part of the personal exposure to ambient PM in a population. In the next chapter, on the epidemiology of PM, a connection will be presented between PM air quality measurements at such a stationary background site monitor and statistically observed health effects.

3. Epidemiology of PM

3.1. Nature of PM-associated health effects

Over the last decade a large number of epidemiological studies have been published on the association between ambient PM exposure and possible health effects. Reported health outcomes are pulmonary function decrements, respiratory symptoms, hospital and emergency department admissions, and mortality. More recently, studies have been published that focus on birth defects and general practitioner visits in relation to PM exposure. In general, a distinction can be drawn between two types of health outcome: *mortality* and *morbidity (including lung function decrements and symptoms).*

3.1.1. Mortality studies

3.1.1.1. Mortality due to acute exposure to PM

More than a hundred published studies (Samet, 2002) have assessed the relationship between excess *daily mortality* (acute mortality) and exposure to PM (expressed as PM_{10}, $PM_{2.5}$, $PM_{10-2.5}$, TSP, BS, CoH) in human populations. A paper published recently by Dab *et al.* (2001) gives a good overview of 15 review papers on studies concerning health effects of PM (in terms of mortality and morbidity). In these studies a time-series analysis is performed on the relationship between daily mortality counts in a population (city, county, country) and daily variations in air pollution levels. Although the first publications in the early 1990s showed differences in their statistical analysis techniques, most of the recently published studies show general correspondence in their analytical approach.

Most interesting in the recent publications are multi-city studies, in which uniform statistical analyses are performed on a data set consisting of daily data from several cities (Katsouyanni *et al.*, 1997; Samet *et al.*, 2000; Schwartz *et al.*, 2000). The advantage of these analyses is the uniform statistical approach and the exclusion of publication bias, because all city-specific data are presented. The analysis most used is the Generalised Additive Model (GAM) Poisson regression with adjustments for seasonal cycles, long-term trend, temperature, day of the week. In a few more recently published studies adjustment was also made for dew point and barometric pressure in addition to the aforementioned co-variates. The outcomes of the models are Relative Risk (RR) estimates, expressing the excess daily mortality (total mortality or cause-specific mortality) per magnitude increase in PM (per 100 $\mu g/m^3$ or per interquartile range, etc.). In the published literature, with only few exceptions, generally consistent mortality RR have been reported, although heterogeneity is reported between different study locations (Samet *et al.*, 2000; Katsouyanni *et al.*, 2001). However, the exact value of the RR presented in the NMMAPS by Samet *et al.* (2000) has recently been questioned because of a discussion about the validity of the software default values. Because no new results for the NMMAPS were available when this report was conceived, the original values are presented here. This question of convergence criterion is treated in more detail in 3.4.1.1., where its influence on the Dutch time-series is estimated. The most recent Dutch update of the mortality analyses is presented in 3.4.1.

The results of these studies confirm previously reported associations, and also add to our knowledge about the relevant PM size fraction. In general, cause-specific analyses show

higher RR for respiratory and cardiovascular mortality than for total mortality or other causes. Although these RR are small compared to RR for lifestyle-associated risk factors (e.g. smoking) and although it has been argued by some that the observed effects estimates for ambient air pollution are not sufficiently constant across epidemiological studies and that epidemiological studies are trustworthy only if they show relatively large effects estimates (e.g. large relative risks), these arguments have only limited weight in relation to ambient air pollution studies. In any large population exposed to ambient air pollution, even a small relative risk for a widely prevalent health disorder could result in a substantial public health burden attributable to air pollution exposure. We look at these issues in greater detail in the following sections.

In most of the studies, the simultaneous inclusion of gaseous pollutants in the regression models did not meaningfully change the PM effect sizes. Gaseous pollutants also remained statistically significant in the two-pollutant models, suggesting that there is an independent association of short-term PM and gaseous pollutant exposures with mortality. Sarnat has shown recently that ambient gaseous pollution concentrations correlate well with personal $PM_{2.5}$ concentrations but not with personal gaseous pollutant concentrations (Sarnat *et al.* 2001), which provides strong evidence that ambient gaseous concentrations act as surrogates for personal $PM_{2.5}$. This might also be the case for other PM fractions like PM_{10}, but no such data were measured in the Sarnat study. However, it is worthwhile interpreting the gaseous associations, found in numerous studies, in the light of the $PM_{2.5}$ findings.

Some new studies have focused on the role of crustal material in PM and mortality effects (Laden *et al.*, 2000; Mar *et al.*, 2000). Summarising, the results of these studies suggest that crustal particles (coarse or fine) per se are not likely to be associated with daily mortality. However, the evidence is not yet conclusive, because in Phoenix (Arizona) data suggest that coarse PM ($PM_{10-2.5}$) may be associated with mortality, and part of this fraction consists of crustal material (but not exclusively crustal). Several new studies have conducted source-oriented evaluations of PM components using factor analysis to estimate daily concentrations due to underlying source types (e.g. mobile emissions, soil, coal) (Laden *et al.*, 2000; Mar *et al.*, 2000; Tsai *et al.*, 2000).

In summary, these studies suggest that a number of source types are associated with mortality, among them vehicle emissions, coal burning and vegetative burning. The crustal factor from fine particles was, however, not associated with mortality data. Therefore, not every fraction of PM seems to be even ‘causal’ for the observed health effects, in other words to be even toxic. Some fractions (and their sources) are probably more health-relevant than others.

Mortality effects from ultrafine particles (UF) are of particular interest, because UF, like all particles with diameters smaller than 4 µm, are able to penetrate deep into the respiratory system. Of these particles, UF may be even more special, as a fraction of it seems to be able to pass the ‘body barrier’ owing to its ability to be translocated from the lung into the interstitium and beyond. This association with UF has been investigated in the city of Erfurt in former East Germany (Wichmann *et al.*, 2000). Both mass concentrations and number concentrations of UF were associated with excess mortality, but when a gaseous co-pollutant was added to the model, drastic reductions in the estimated RR for UF were sometimes found. Since mass and number concentrations were rather well correlated with gaseous co-pollutants, it is very difficult to separate the PM- or gaseous-specific health effects.

3.1.1.2. Mortality due to chronic exposure to PM

There are far fewer published studies on the mortality effects of *long-term* exposures to ambient PM than on the mortality effects of acute exposure. The current number of published cohort studies is five. The total number of publications in scientific journals about these five cohort studies is, of course, much larger, but essentially all conclusions are based on the information contained in just five cohort studies.

1. The first one is the Harvard Six Cities (HSC) study (Dockery *et al.*, 1993). In the HSC study, the associations between mortality and long-term exposure to sulphate and several PM size fractions were investigated in six American cities. The HSC comprised 8111 white subjects, who were followed for 111 076 person years. This study suggests that the fine mass component ($PM_{2.5}$) is more strongly correlated with mortality than is the coarse component ($PM_{15 \text{ or } 10 \text{ minus } 2.5}$). Overall, the results showed statistically significant relationships between long-term exposures to PM and/or sulphates and excess mortality. A further follow-up of this study has recently been completed, but not yet published.
2. The American Cancer Society (ACS) study (Pope *et al.*, 1995, with a follow-up in 2002) is the second study of long-term exposure. In the ACS, the risk factor data of 500 000 adults were linked with air pollution data and combined with vital status and cause of death between 1982 and 1998. This is the largest cohort available to date. The results of the ACS indicate that fine particle and sulphur oxide-related pollution were associated with all-cause, cardiopulmonary and lung cancer mortality. Each 10 $\mu g/m^3$ elevation in $PM_{2.5}$ was associated with an approximately 4%, 6% and 8% increased risk in the respective causes of death. Measures of coarse PM, TSP and gaseous pollutants such as NO_2, CO and O_3 were not consistently associated with mortality.
3. The Adventist Health Study on Smog (AHSMOG) (Abbey *et al.*, 1999) represents the third major U.S. prospective cohort study of chronic PM exposure-mortality effects. In 1977 the study enrolled 6338 non-smoking non-Hispanic white Seventh-Day Adventist residents of California between the ages of 27 and 95. The authors found long-term ambient concentrations of PM_{10} to be associated with increased risks of all natural cause mortality in males, mortality with any mention of non-malignant respiratory causes in models that included both sexes, and lung cancer mortality in males. Lung cancer analyses were based on small numbers (18 deaths for females and 12 for males) requiring further exploration in follow-up studies.
4. In a fourth cohort study, Lipfert *et al.* (2000a) reported results from a large-scale mortality analysis using a prospective cohort of up to 70 000 men assembled by the U.S. Veterans Administration (VA) who were diagnosed as hypertensive in the mid-1970s. The study cohort was male, middle-aged (51 ± 12 years) and included a larger proportion of African-Americans (35%) than the U.S. population as a whole and a large percentage of current or former smokers (81%). Contrary to three of the previous cohort studies, no associations with particulate matter were found in relation to mortality.
5. In the Netherlands, the association between mortality and distances to major roads of the home addresses of respondents in combination with a regional and urban background is being studied in an ongoing cohort study – The Netherlands Cohort Study on Diet and Cancer (NLCS). A cohort of 120 000 55- to 69-year-old subjects was enrolled in a study on diet and cancer in 1986. Subjects were recruited from a large number of communities throughout the Netherlands. Hoek *et al.* (2002c)

investigated a random sample of 4492 people from the full cohort. Long-term exposure to traffic-related air pollutants (BS and NO_2) was estimated for the 1986 home address. During the follow-up period, 489 people died. Cardiopulmonary mortality was significantly associated with living near a major road (RR 1.95). For total deaths the RR for living near a major road was 1.41, though statistically not significant. Non-cardiopulmonary and non-lung cancer deaths were unrelated to air pollution.

A grant has been obtained from HEI to include all 15 000 deaths that have occurred so far in the cohort for the analysis of a relationship with traffic-related air pollution. Results are expected in 2005. With support from NAP, a small extension of the analysis will be conducted in the course of 2002.

In qualitative terms it can be argued that the US results (ACS and HSC) pointing to health effects associated with chronic exposure to PM have been corroborated by this Dutch study, which focuses on the effects of traffic (Hoek *et al.*, 2002c).

Weighing all the evidence of these cohort studies and assigning specific significance to the fact that the first two studies comprised a sample of the general population and have been thoroughly re-analysed, some of the authors are convinced that chronic exposure to ambient levels of PM in the US have led to the reported health effects. Both studies (HSC and ACS) have been supplemented by a follow-up, which generally points in the same direction as well, though only one of the follow-ups (ACS) has as yet been published. The AHSMOG population, however, consisting of Seventh-Day Adventists, is very different in a number of lifestyle factors from the general population, so that findings in this particular population cannot easily be extrapolated. The HSC and NLCS studies, and to a somewhat lesser extent the ACS study, are truly population-based. This is not so for the veterans study (VA), for which only a preliminary report is available. Clearly, the weight of the evidence is with the HSC, ACS and NLCS studies at the moment.

However, some of the authors conclude that the overall picture from these five studies is mixed in regard to the evidence concerning chronic exposure to PM and associated mortality, as some of the studies point to PM effects and some do not. These authors think it is too early to come to a conclusion. Not finding a significant PM effect in two of the studies does not, of course, prove that there are no health effects from chronic exposure to PM. The information from the other studies indicates that the associations can be called qualitative at the least and warrants the serious attention of those involved with environmental health effects. The question of whether these studies can also be used for the quantitative risk estimation of chronic PM effects in the Netherlands is of a different order and will be explored later on.

A legitimate question to ask when interpreting the results of ‘ecologic’ studies is whether the individual concentrations are represented well enough by the ambient concentrations of the central site monitors. Exposure in the US cohort studies of chronic exposure to PM is described at a group level instead of at an individual level. This may lead to the so-called ecological fallacy. The group-level information on all the people living within a radius of a number of kilometres is represented by the concentration at a central site monitor or a number of monitors considered to be representative for the sought-after personal exposure to ambient PM. Reviewing the epidemiological literature on the relation between ambient air pollution exposure and cancer, which included the first two cohort studies (HSC and ACS), Katsouyanni and Pershagen (1996) concluded: ‘A major drawback in the studies has been the inadequate characterisation of air pollution exposure. First, measurements of air pollution in

the study areas should span the time period relevant for the disease etiology and preferably should include concentrations of suspected carcinogens. Second, an estimation of individual exposure should be based on exposure studies, studies of the time activity patterns, and the geographic distribution of pollutants in a micro-scale. Exposure studies should provide data on how individual exposure is related to the levels measured at fixed monitors, considering different activities and transportation used. Further methods for retrospective exposure assessment covering periods of several decades should be developed. The results of such studies could be used as input in large analytic epidemiological investigations to address the problems of measurement error and reduce uncertainties in the RR estimates.'

The currently used exposure measures (in HSC, ACS, AHSMOG) are those during the final number of years immediately preceding death. Unfortunately, we do not really know the time window for relevant exposure. Referring to cessation of smoking, a 50% reduction for lung cancer risk takes approximately 10 years, and after 20 years it has decreased to a background level. For cardiopulmonary diseases this period appears to be much shorter and may be in the order of years. If different individual measures of exposure comprising such a window of exposure, which are of course much harder to come by, had been used, the magnitude of the assigned exposures would probably have been larger than the currently assigned PM exposures in the studies. Quantitatively, this might mean that the currently reported relative risks in the chronic studies, assuming that they would remain statistically significant with the new exposure measures, would have been lowered by a similar amount to that by which the exposures have been augmented. These arguments about exposure validity indicate that the time domain of extrapolation needs to be explored very carefully.

The effect of the reduction in PM over the last few decades should show up in the follow-up period in lower numbers of health effects associated with PM compared with the previous period in the cohort studies for chronic exposure. Unfortunately, the currently reported follow-up of the ACS study does not order its information in a way to present such a picture. The associated extra risk of cardiopulmonary mortality in the US per 10 $\mu g/m^3$ decreases from 13% to 4% when the first period is compared with the total period, which includes the first period and the follow-up. A more or less similar RR would seem the logical result to expect when the decreases in PM concentrations over the last few decades are taken into account and chronic exposure to PM is causal for the health effects. A decrease in RR could be due to a change in the ambient PM mix. However, for lung cancer in the ACS the picture is reversed, and lung cancer mortality per 10 $\mu g/m^3$ increases from 1.4% to 8% when the first period is compared with the total period. Finding both results in the same study seems contradictory.

Krewski *et al.* (2000) indicated that in the ACS study the relationship between PM exposure and mortality disappeared for high SES. A finding like this mostly points to some sort of effect modification, though it could also point to an insufficient control for occupational exposures of those with less than high school education. In the ACS follow-up (Pope *et al.*, 2002) the health effects of PM, which are significant for those with an education level less than high school, disappear in Figure 4 for those with more than high school education. This may indicate that something in the lifestyle or diet of the group in question can mitigate or prevent the PM-associated health effects. Further research in this direction is warranted, as it may lead us to lifestyle or other personal factors that could be influenced to reduce PM risks to the population.

Lipfert *et al.* (2000a) concluded from the VA as a new insight: '... the general decline of mortality responses to air pollution with increasing follow-up time. This trend could suggest depletion of the cohort of its most susceptible subjects, a concentration-response threshold, increasing uncertainty about the exposures and the characteristics of the cohort, or all of these. A bona fide chronic effect would be expected to be manifested throughout the period of follow-up, especially as the cohort ages, not just at the beginning. It thus follows that other such cohort studies should also examine the ramifications of the timing of air pollution exposure.' A substantiation against an extrapolation in time can be found in a very recent cross-sectional study for five specific periods from 1960 to 1997, which examined temporal and spatial relationships between air pollution and age-specific mortality rates for US counties (Lipfert and Morris, 2002). On the basis of attributable risks computed for overall average concentrations, the strongest associations were found in the earlier periods, with attributable risks usually less than 5%. Stronger relationships were seen when mortality and air quality were measured in the same period and when locations were limited to those of the HSC. Responses to PM, CO and SO_2 declined over time, suggesting that the results of previous studies may no longer be applicable.

Although most of the mortality studies focused on the total population or the elderly, new studies on mortality in the very young suggest PM effects in a susceptible sub-population of children (Bobak and Leon, 1999; Loomis *et al.*, 1999; Lipfert *et al.*, 2000b). Post-neonatal mortality has been associated with PM levels. In the current NAP project on PM-related health effects we have tried to collect data on post-natal and post-neonatal deaths in the Netherlands to study this phenomenon and to find out if it occurs in the Netherlands too. Unfortunately, we were unable to obtain these data because they were not available for privacy reasons.

a. Transferability to the Netherlands of health effects of chronic exposure

Two of the previously presented long-term studies indicate (relatively) high mortality risks for rather low levels of PM_{10}, $PM_{2.5}$ or sulphates in the US. A quite recent re-analysis of the data of the HSC and the ACS study (Krewski *et al.*, 2000) sponsored by the HEI indicated that the original calculations and data handling satisfied all the necessary scientific and technological standards.
An excess of 20 µg/m^3 $PM_{2.5}$ in the HSC is associated with 28% excess mortality, leading to a relative risk of RR = 1.28 per 20 µg/m^3 $PM_{2.5}$ (US-EPA, 2001). If the results of these US studies are also applicable to the Netherlands (with an annual average concentration of approximately 20–25 µg/m^3 $PM_{2.5}$), the magnitude of the RR signifies that the health effects of chronic exposure to PM constitute a huge problem with a large health impact. If true for the Netherlands, the American HSC figures imply on average an excess mortality of 30%, resulting in a serious shortening of our life span. However, the most recent RR from the follow-up of the ACS is somewhat lower than that from the HSC. Pope *et al.* (2002) indicate a 4% rise in all-cause mortality and a rise of 8% in lung cancer mortality for a 10 µg/m^3 elevation in $PM_{2.5}$. Translated into the current situation in the Netherlands, with approximately 20–25 µg/m^3 $PM_{2.5}$, this would mean an additional all-cause mortality of approximately 10% and an additional lung cancer mortality of 20%. These health implications remain considerable; careful exploration is necessary to see whether such an extrapolation from the US to the Netherlands is permitted. At the average concentrations, the health effects estimated by Pope *et al.* (2002) are even higher.

Some of the authors argue that there is no fundamental difference between sources and air pollution concentrations in the US and the Netherlands. Differences within the US are probably larger than between relevant areas in the US and the Netherlands. Long-term effects have been found both in the west and in the east in the US, which is why some of the authors conclude that the long-term effects found in the US can be extrapolated to the Netherlands. The health impact of PM in the Netherlands can be quantified on the basis of the HSC and ACS results.

However, some of the authors point more to the differences than to the existing similarities in PM air pollution in the US and the Netherlands. A formal argument could be used to declare such an extrapolation non-permissible. For instance, strictly scientifically speaking, extrapolating a regression (and these epidemiological results are essentially regressions) outside of its geographical and time domain is never permitted. The air pollution mix in the Netherlands and that in the US are quite different in a number of aspects, and so are the health effects probably, too. Concentrating on one aspect only, namely sulphate, the latest exposures in the ACS presented in Pope *et al.* (2002) show US sulphate levels to be 6.2 $\mu g/m^3$ with an SE of $\pm$ 2.0. This leads to the conclusion that the current annual average levels of sulphate in the Netherlands, which are some 2 $\mu g/m^3$, are presently outside the 95% confidence interval of the US sulphate data in the ACS. Of course, sulphate levels in the Netherlands were also higher in the past, but for an up-to-date risk estimate we need to use our current levels.

Using the HSC data at face value for the Netherlands may lead to inconsistencies in the risk estimates when different indicators for the air pollution mix are used, e.g. when the current increase in all-cause mortality in the Netherlands is estimated using two different indicators from the HSC. Using the Dutch $PM_{2.5}$ data and HSC RR leads to an increase in all-cause mortality of approximately 30% in the Netherlands, as has been indicated above. When, on the other hand, the HSC RR data are used and sulphate is taken as an indicator instead of $PM_{2.5}$, then the previously found increase in all-cause mortality in the Netherlands completely disappears, because the ambient levels of sulphate in the Netherlands are considerably lower than those in even the least polluted of the six American cities in the HSC, and the RR in the HSC were calculated in comparison with the location with the lowest pollution levels. Ayres (2002) cautioned in an editorial how unwise it was to extrapolate effect size coefficients from one area to another with the questions currently still open.

In reality, it is necessary to determine very carefully, conclude some of the authors, which constraints need to be met concerning the air pollution mix and geographical location (spatial) or time period (temporal) in order to validate an extrapolation and use the results obtained in the US for a quantitative risk assessment elsewhere.

These conclusions or differing views concerning the quantitative extrapolation of foreign data are no reason for complacency, however. All these arguments and the qualitative information from the four US studies plus the preliminary results of the Dutch cohort study constitute a strong plea for a well-designed prospective Europe-wide cohort study, to provide us with some of the much desired answers to the question of the health effects of chronic exposure to PM.

Last but not least, even without the effects attributable to chronic exposure to PM, the magnitude and seriousness of the health effects of acute exposures implied by the time-series studies (which have been replicated numerous times, Samet (2002)) are such that a high

position on the research agenda is justified. Once the causal factors of these PM-associated health effects have been elucidated, it may become possible in the future to develop cost-effective mitigation or abatement strategies, or both.

3.1.2. Morbidity studies

3.1.2.1. Hospital admissions and ER visits for respiratory diseases

Several morbidity outcomes have been studied in relation to PM exposure. The most investigated association is unarguably the association between daily hospital admissions for respiratory diseases and hospital emergency room visits. A recently published paper (Dab *et al.*, 2001) gives a good overview of seven review papers on studies concerning hospital admissions associated with particulate matter. As in the acute mortality studies, most of the recently published studies show general correspondence in their analytical approach, compared with the first studies published. The Generalised Additive Model (GAM) Poisson regression analysis with adjustments for seasonal cycles, long-term trend, temperature, day of the week is most used. Again, results from multi-city studies are of special importance because they are larger in magnitude than individual studies, have standard statistical analyses and do not suffer from potential publication bias when 'negative' results are found (Katsouyanni *et al.*, 1996; Spix *et al.*, 1998; Samet *et al.*, 2000; Schwartz *et al.*, 2000).

Evaluation of the large number of studies reveals a generally consistent positive association between hospital admissions for respiratory diseases (COPD, asthma, pneumonia). The estimated excess risks fall in the range 5–25% per 50 $\mu g/m^3$ PM_{10} increments, with those for asthma visits and hospital admissions tending to be somewhat higher than for COPD and pneumonia hospital admissions. In the latest Dutch study on the association between PM_{10} exposure and hospital admissions, the RR for respiratory admissions was 1.03 (lag 3), for COPD 1.06 (lag 3) and for asthma admissions non-significant (RR per 63 $\mu g/m^3$ PM_{10} increment). The largest RR therefore was for COPD and not for asthma admissions (Vonk and Schouten, 2002). Further details of this latest study can be found in 3.4.2. New, limited, foreign evidence points not only to PM_{10} and $PM_{2.5}$ as relevant size fractions, but also towards associations of such admissions with ambient coarse particles ($PM_{10-2.5}$) (Sheppard *et al.*, 1999; Lippmann *et al.*, 2000). In addition, like in the acute mortality studies, in most of the studies where gaseous pollutants were simultaneously included in the regression models the PM effect sizes were not meaningfully affected, although there is a tendency for a decrease in the PM estimates through the addition of a gaseous pollutant. Gaseous pollutants remained statistically significant as well in the two-pollutant models, suggesting that there is an independent association of short-term PM and gaseous pollutant exposures with acute respiratory morbidity. However, Sarnat has shown recently that ambient gaseous pollution concentrations correlate well with personal $PM_{2.5}$ concentrations but not with personal gaseous pollutant concentrations (Sarnat *et al.*, 2001) which provides strong evidence that ambient gaseous concentrations act as surrogates for personal $PM_{2.5}$. This might also be the case for other PM fractions like PM_{10}, but no such data were measured in the Sarnat study. It is, however, worthwhile interpreting the gaseous associations, found in numerous studies, in the light of the $PM_{2.5}$ findings.

3.1.2.2. Hospital admissions for cardiovascular diseases

In recent years new insight has evolved regarding the relation between daily variations in ambient PM and daily variations in hospital admissions for cardiovascular diseases. Overall, the results of these studies suggest that elevated concentrations of PM are associated with hospital admissions for cardiovascular disease. In studies where gaseous pollutants were simultaneously included in the regression models, the PM effect sizes were not meaningfully affected. Gaseous pollutants also remained statistically significant in the two-pollutant models, suggesting an independent effect of short-term PM and gaseous pollutant exposures on acute cardiovascular morbidity. For US cities, an overall estimate for the RR ranges from 3 to 10% per 50 $\mu g/m^3$ increment in ambient PM_{10}. In the Netherlands, an excess of 3% (CI: 2–4%) in daily hospital admissions has been associated with a 63 $\mu g/m^3$ increase in PM_{10} at lag 0 (Vonk and Schouten, 2002).

3.1.2.3. Studies on individual cardiovascular outcomes

Although ecologic and semi-ecologic epidemiological studies have linked particulate air pollution with cardiopulmonary mortality and morbidity, the underlying biological mechanisms remain largely unknown. New studies published recently have investigated in small, susceptible populations (elderly people with pre-existing cardiopulmonary diseases) the associations between daily variability in PM ambient concentrations and daily variability in physiological measures of cardiovascular function at an individual level, i.e. by assessing information on potential confounders and effect modifiers at an individual level (Liao *et al.*, 1999; Pope *et al.*, 1999a; Pope *et al.*, 1999b; Dockery *et al.*, 1999; Gold *et al.*, 2000). These studies showed that decreased heart rate variability (HRV) and increased heart rate (HR) were associated with increased levels of ambient PM air pollution. However, the few studies performed so far are not consistent: negative associations between PM and heart rate have also been found. In a study conducted in Germany, both PM and gaseous pollutants were associated with an increase in heart rate (Peters *et al.*, 1999). New, European results on this issue will evolve from the EU-funded ‘Ultra’ project, which is a multi-centre study in German, Finnish and Dutch cities focusing on the health risks of UF.

An interesting new approach in air pollution cardiovascular epidemiology is use of the registration of implanted cardioverter defillibrators in relation to ambient air pollution levels (Peters *et al.*, 2000). Although some initial results from these studies suggest that ambient air pollution (not only PM but gaseous components as well) was associated with arrhythmia events, these results should be interpreted with caution because of the small number of participants with a cardiac event in the study. Interesting work has also been done in the field of plasma viscosity. Significant elevations in blood viscosity were found during an air pollution episode in Germany in the 1980s (Peters *et al.*, 1997). Other blood indices like blood counts, fibrinogen and IL6 have also been found to change in relation to ambient air pollution. These findings provide further support for some intriguing hypotheses regarding possible mechanisms by which PM exposure could be linked to adverse cardiac health outcomes.

3.1.2.4. Lung function and respiratory symptoms

In contrast to the consistency between the different studies of mortality and hospital admission and ambient PM air pollution, there is surprisingly less consistency between the results of

studies that assessed the relationship between ambient PM and lung function and/or respiratory symptoms. Most of these studies followed a panel of subjects (asthmatics or non-asthmatics or children with respiratory symptoms) over a certain period of time, with measurements of pulmonary function (daily or otherwise) and daily registrations of respiratory symptoms. The lung function measures used were Forced Vital Capacity (FVC), Forced Expiratory Volume (FEV) and Peak Expiratory Flow rate (PEF). Respiratory symptoms mostly investigated were upper respiratory symptoms, lower respiratory symptoms and cough.

The largest study to date was the PEACE study, a multi-centre study in Europe with participation from all over the continent – from Finnish to Greek research groups and others in between (Roemer *et al.*, 1998). The study's overall conclusion was that no consistent association could be found between ambient PM_{10} and respiratory health outcomes in the panels of children with chronic respiratory symptoms. A later review argued that the design of panel studies made them much more susceptible to uncontrolled confounding than the long time-series studies on mortality and hospital admissions (Roemer *et al.*, 2000). They showed, for instance, that influenza-like illness, as registered in a GP sentinel system, was clearly associated with lung function and symptoms in concurrent panel studies (Van der Zee *et al.*, 2000a) In fact, in the Netherlands the PEACE study was part of a multi-year series of panel studies which actually did show fairly consistent effects of PM on lung function, medication use and symptoms (Boezen *et al.*, 1998; Van der Zee *et al.*, 1998; Boezen *et al.*, 1999; Van der Zee *et al.*, 1999; Van der Zee *et al.*, 2000a). This adds to earlier studies in the Netherlands which have documented acute effects of PM on lung function and symptoms in children and adults (Dassen *et al.*, 1986; Brunekreef *et al.*, 1989; Hoek *et al.*, 1990; Brunekreef and Hoek, 1993; Hoek and Brunekreef, 1993; Roemer *et al.*, 1993; Hoek and Brunekreef, 1994; Dusseldorp *et al.*, 1995; Gielen *et al.*, 1997; Hoek *et al.*, 1998).

In the context of the NAP, a study is being conducted by IRAS and other groups of the long-term effects of traffic-related air pollution. A cohort of about 4000 new-born children was enrolled in a birth cohort study on the development of allergy and asthma in 1996/1997. Subjects were recruited from Rotterdam and surroundings, from the central-eastern part of the country, and from the three northern provinces. With support from the EU, a monitoring programme was conducted to estimate concentrations of $PM_{2.5}$, BS and NO_2 at the home address. Measurements were conducted at 40 sites to develop a stochastic model based on variables obtained from a Geographic Information System (GIS). Using this model, $PM_{2.5}$, BS and NO_2 concentrations were then estimated for all 4000 addresses in the birth cohort. A similar study was conducted in Stockholm and Munich. The study showed that in all three study countries, there was significant spatial variation in long-term concentrations of $PM_{2.5}$, BS and NO_2. A large fraction of the variance of these concentrations could be readily explained by variables available in Geographic Information Systems (GIS). This was especially true for BS (Hoek *et al.*, 2001c; Hoek *et al.*, 2002b). An initial analysis has been made of the relationship between traffic-related air pollution exposure and respiratory symptoms observed in the first two years of life in Germany and the Netherlands. In Germany, cough without infection and dry cough at night were associated with all three measures: $PM_{2.5}$, BS and NO_2 (Gehring *et al.*, 2002). In the Netherlands, influenza and upper respiratory tract infections were associated with traffic-related air pollution, and there was an indication of an association with wheeze and asthma as well (Brauer *et al.*, 2002). A grant has been obtained from the EU to conduct further follow-up of the three cohorts until the age of four, and to include various measures of indoor air pollution. With support from NAP, the exposure database which was developed in the TRAPCA project will be linked to the medical data

obtained at the age of four in the Dutch cohort. Data management is almost complete and results are expected in the course of 2002.

3.1.2.5. Long-term PM exposure and respiratory symptoms

A large number of studies investigating the relationship between chronic exposure to ambient PM and respiratory symptoms have been published in recent years. In some well-conducted large studies of both adults and children (most importantly the 24-cities study in the US (Dockery *et al.*, 1996; Raizenne *et al.*, 1996; Spengler *et al.*,1996, and SCARPOL and SAPALDIA in Switzerland (Ackermann-Liebrich *et al.*, 1997; Braun-Fahrlander *et al.*, 1997; Zemp *et al.*, 1999; Kunzli *et al.*, 2000)) lung function and bronchitis symptoms (but not asthma) are associated with PM and PSA (US). Recent studies, some of which were conducted in the Netherlands (Oosterlee *et al.*, 1996; Brunekreef *et al.*, 1997; Van Vliet *et al.*, 1997; Roorda-Knape *et al.*, 1998) have shown that exposure to traffic-related air pollution was related to respiratory health endpoints. Recent exposure work has demonstrated that living near busy roads and freeways has clearly demonstrable effects on ambient, indoor and personal exposure to traffic-related air pollution, including PM (Janssen *et al.*, 2001; Rijnders *et al.*, 2001).

3.1.3. Conclusions on morbidity and mortality studies

There is growing evidence in the scientific literature indicating that daily changes in ambient PM air pollution are associated with changes in mortality and morbidity. Respiratory and cardiovascular mortality show the largest associations with ambient PM and hospital admissions for both these diseases are significantly associated with it. However, there is also evidence that other components of the air pollution mixture can have independent associations with health outcomes. In most of the studies gaseous components were also significantly associated with mortality and hospital admissions. New studies focusing on the underlying mechanism that may lead to the observed health effects provide further support for some intriguing hypotheses regarding possible mechanisms by which PM exposure could be linked to adverse cardiac health outcomes. Several studies show associations between lung function and/or respiratory symptoms and ambient PM air pollution or traffic-related air pollution. The most recent Dutch updates are summarised in 3.4.1. (on mortality) and 3.4.2. (on morbidity).

3.2. Form of concentration-response curve and threshold concept

There is general agreement in the scientific literature that there is a concentration-response relationship with no indication of a threshold value. The exact form of this relationship, however, remains unclear. Some studies show linear relationships, while others show non-linear relationships. These non-linear relationships are quite often convex. There is some suggestion that the form of the concentration-response curve depends on the outcome variable being studied (Krewski *et al.*, 2000). In the Dutch mortality time-series analyses (Hoek *et al.*, 1997), non-linear concentration-response curves were found for all pollutants. More recent analyses (see 3.4.1.1. and Figures 3.1 and 3.2) showed similar results. There is also general agreement in the scientific literature that it is impossible to identify a threshold below which health effects do not occur. This was also the case in the previous Dutch mortality time-series analyses (Hoek *et al.*, 1997). Recent re-analyses and an update of time-series studies

confirmed the absence of a threshold value in the concentration-response curves between PM and total and/or cardiopulmonary mortality (Fischer *et al.*, 2002).

The conclusion of this section is that it is not possible to determine a threshold for PM effects and that a convex (non-linear) concentration-response curve indicates that health effects seem to level off at higher concentrations.

3.3. Specific populations at risk

Health effects related to PM have been shown to occur in all groups of the population: neonates, infants, children, young and old adults. However, there seem to be specific groups in which the associations with health effects and PM are stronger than in the total population. (Boezen *et al.*, 1999).

3.3.1.The elderly

The group probably identified best is the group of the elderly, presumably those with pre-existing cardiopulmonary diseases. Relative risks for mortality and morbidity tend to be larger in the elderly group (> 65). In the Netherlands this was the case for mortality and hospital admissions, with a general tendency for higher RR estimates in the age group 65 and above (Hoek *et al.*, 1997; Vonk and Schouten, 1998 and 2002). Age-specific analyses showed that statistically significant associations between air pollution and mortality were mostly found in the elderly (> 65) and that the RR for deaths below the age of 65 were generally smaller than in the older age groups. Exceptions to these overall conclusions were the associations between air pollution and pneumonia deaths, which showed a high RR in the age group 45–64, suggesting that we cannot rule out the fact that for some causes of death (pneumonia) younger age groups also appear to be at risk. Higher RR of particulate matter for deaths in the over-65 age group have been reported before in Philadelphia (Schwartz and Dockery, 1992), Cincinatti (Schwartz, 1994) and Amsterdam (Verhoeff *et al.*, 1996) and more recently in Montreal (Goldberg *et al.*, 2000). For Rotterdam, with TSP as a PM metric, RR were substantially larger for deaths over the age of 78, the median age at death (Hoek *et al.*, 1997).

3.3.2. Pre-existing diseases

There is substantial evidence in the epidemiological literature that cause-specific relative risks for cardiopulmonary mortality are larger than for total mortality, indicating that people with pre-existing cardiopulmonary diseases are more susceptible to the effects of air pollution. This is also the case in the Dutch time-series analyses of PM and cause-specific mortality (Hoek *et al.*, 2001a). Deaths due to heart failure, arrhythmia, cerebrovascular and thrombocytic causes were more strongly associated with air pollution than cardiovascular deaths in general. Excess relative risks were 2.5–4 times larger for these categories than for total cardiovascular disease mortality. Heart failure deaths, which made up 10% of all cardiovascular deaths, were found to be responsible for about 30% of the cardiovascular deaths related to air pollution.

Also, hospital admissions for respiratory diseases are consistently associated with ambient PM levels. New studies of hospital admissions indicate that hospital admissions for cardiovascular diseases are significantly associated with PM ambient air pollution. So, not only for mortality,

but for morbidity as well there appears to be a group of people (the elderly) with pre-existing cardiopulmonary diseases who are at greater risk than the (younger) disease-free population. An extremely interesting and enlightening study from Montreal was very recently published (Goldberg *et al.*, 2000; Goldberg *et al.*, 2001). The investigators were able to identify the underlying clinical conditions of the deceased based on medical records from insurance databases. Increases in PM levels were linked to higher death rates in subjects with acute lower respiratory disease, congestive heart failure and chronic coronary artery disease. These results strongly suggest that persons with certain pre-existing cardiac or respiratory conditions are at short-term increased risk of ambient-PM-associated mortality.

3.3.3. Children

On the other side of the age distribution, in the very young, the group of neonates and young infants seems to be at higher risk of mortality and morbidity. Some studies have suggested that neonatal mortality and birth defects are related to PM ambient air pollution (Bobak and Leon, 1999; Loomis *et al.*, 1999; Lipfert *et al.*, 2000c). This is an important finding, if true, because childhood death or illness may cost a very large number of productive (and disease-free) life years. However, the number of studies that have thus far investigated this issue is still small, so conclusions must be drawn with caution. Based on dosimetry, the children group is at risk of PM ambient air pollution health effects because they spend a greater amount of time outdoors, they have greater physical activity levels and higher breathing rates, and a higher dose per lung surface area (see Table 4.1).

Although a number of studies have shown respiratory health effects in children related to PM ambient air pollution, other studies have failed to show this association. Therefore, the evidence that children as a group are more vulnerable to ambient air pollution than other age groups is less conclusive than that for age and pre-existing cardiopulmonary diseases as a risk factor for susceptibility to PM ambient air pollution.

3.3.4. Conclusion

Pre-existing cardiopulmonary diseases and old age seem to be the most important risk factors for susceptibility to PM ambient air pollution. Based on a small number of epidemiological studies, the very young (neonates and post-neonates) may be susceptible as well.

3.4. Recent Dutch results with acute exposures

3.4.1 Netherlands update of time-series analyses based on mortality

3.4.1.1. Study design and convergence criterion

In the Netherlands we used the GAM approach to study the association between daily mortality and hospital admissions and air pollution over two periods: 1986–1994 and 1992–1998. One of the purposes of the first study was to evaluate whether the magnitude of the air pollution effects as reported in studies from other countries was similar in the Netherlands, and to answer the question of transferability experimentally. The results of the study were used to estimate the actual number of premature deaths in the Netherlands associated with current levels of air pollution. The results of the first study period (1986–1994) have been

presented (Hoek *et al.*, 1997a; Hoek *et al.*, 1999; Hoek *et al.*, 2000; Hoek *et al.*, 2001b). In the first time-series analyses, data for PM_{10} were only available for the three-year period 1992–1994, because regular Dutch measurements of PM_{10} only started in 1992.

In 1999 this series was extended with another four years of ambient air pollution data, and it was decided to re-run the former analyses on a larger PM_{10} data set (1992–1998) to obtain more stable and more precise estimates than was possible with the first analyses. The associations for other particulate matter air pollution (BS, SIA, nitrate and sulphate) and for the gaseous components (SO_2, NO, NO_2, CO and NH_3) were also to be analysed for the same period. The results of this analysis are reported in more detail by Fischer *et al.* (2002) and are summarised in this section.

All statistical analyses were performed using the default convergence criteria as used in the S+ software. While finalising the analyses, it became apparent that in the US there was some discussion about the validity of these default criteria for correlated data sets like the mortality-air pollution data. This was demonstrated for the first time in the US-NMMAPS study (National Morbidity, Mortality and Air Pollution Study). The researchers involved in this study suggested that the S+ default convergence criteria used in GAM might be 'inadequate to assure that the convergence of its iterative estimation procedure actually reaches the maximum likelihood' (fax HEI, 30 May 2002). Within a couple of days Dr. T. Hastie distributed a new set of convergence criteria, which are stricter than the initial criteria (developed for computer use in 1990, when computers were much slower than today), through the S+ mailing list. More can be found on the following website:
(http://www.insightful.com/support/faqdetail.asp?FAQID=139&IsArchive=0)

For this reason, we also re-ran our models using the new convergence criteria for S+ that had recently become available. However, due to time constraints we decided to run the re-analysis on a core-set of the whole data set. Since the focus of our project was on PM_{10}, we re-calculated the association between PM_{10} and five outcome measures, based on the existing confounder models and with the new convergence criteria. The results of these re-calculations are shown in Table 3.1. The table shows that the impact of the change in convergence criteria on the point estimates is minimal (less than 10%). Standard errors shifted slightly towards 1, but all the significant associations remained. We therefore concluded it was unlikely that results for other air pollutants would change if the new convergence criteria were used. Since the issue is still being debated while the report is being finalised, we decided not to re-analyse the other air pollution associations. All the reported RR are for the 'old' default S+ convergence criteria.

Table 3.1 Relative Risk (RR) for mortality at lag 1 for PM_{10} and various causes of death in the Netherlands per 100 $\mu g/m^3$ increase in PM_{10} (1992–1998) with the strictest new convergence criteria and with default S+ criteria.

	Strictest new criteria	Default S+ criteria	% difference in risk estimate
Total mortality	1.034	1.036	6
Respiratory mortality	1.115	1.120	4
COPD mortality	1.105	1.111	5
Pneumonia mortality	1.115	1.121	5
Cardiovascular mortality	1.024	1.025	4

3.4.1.2. Methods

The mortality data for the Netherlands were obtained from Statistics Netherlands (CBS) for the years 1986–1998. Dutch citizens who died outside the Netherlands and non-residents who died inside the Netherlands were excluded, as well as deaths for which the exact date of death was unknown. Also, individuals who died as a result of accidents (ICD-9 code 800 and higher) were excluded. Stratification according to address density and according to region was performed to assess whether the association between air pollution and mortality was different for different degrees of urbanisation and for different regions within the country with different levels of air pollution. Statistics Netherlands uses as a measure for urbanisation the mean number of address densities within a municipality. Address densities are classified into 5 classes. Class 1 has a very high degree of urbanisation (≥ 2500 addresses per km^2) and Class 5 is non-urban (< 500 addresses per km^2), with the three intermediate classes being 1500–2500, 1000–1500 and 500–1000 addresses per km^2. Because address densities changed over the research period 1986–1998, we chose 1992 (middle of the period) as the reference year by which municipalities were classified. The regions with the highest address densities in the west of the Netherlands are referred to as 'traffic high' in the remainder of this report and the region with the lowest address densities in the north, south and east of the Netherlands as 'traffic low'. On the basis of OPS modelling, the difference between 'traffic high' and 'traffic low' will be a factor of approximately two for traffic-related primary PM.

Air quality data were obtained from the National Institute of Public Health and the Environment (RIVM) in Bilthoven, which operates the national air quality monitoring network (NAQMN). See Figure 2.3 and subsection 2.3.1. for a description of the network and monitoring sites. The air pollution data used for the analysis are presented in Tables A1 and A2 of Annex A, and in the report by Fischer *et al.* (2002), of course. Data relating to temperature, relative humidity and air pressure, influenza and pollen were collected in a similar manner to the analyses of Hoek *et al.* (1997a) and Hoek and Brunekreef (1999). The relationship between daily changes in air pollution and mortality was modelled using Poisson regression. In Poisson regression models the logarithm of mortality is modelled with a linear function of predictor variables. All pollution-mortality associations were adjusted for long-term and seasonal trends, influenza morbidity counts, ambient temperature, relative humidity and barometric pressure and indicators for day of the week and holidays. The method described is based upon an earlier study conducted in the Netherlands (Hoek *et al.*, 1997a), the APHEA-2 study (Katsouyanni *et al.*, 2001) and the NMMAPS study (Samet *et al.*, 2000). The analysis consisted of a confounder model identification phase and a phase in which air pollution was added to the identified confounder model. In the identification phase a systematic procedure was followed to select a confounder model without taking into account air pollution.

For presentation, regression coefficients were transformed into relative risks associated with concentrations changes guided by the difference between approximately the 1-th and 99-th percentile of the concentration distribution for the Netherlands as a whole. The exact concentration changes per pollutant are presented in the second column of Table 3.9.

3.4.1.3. Results of time-series update on mortality

Table A3 in Annex A shows the associations between air pollution and daily total mortality (lag 1 and weekly average) for different regions in the Netherlands. A summarising table for a lag of one day is presented in Table 3.2 below.

*Table 3.2 Association between air pollution and **total mortality** in different regions of the Netherlands. Relative risk (RR*1000) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders.*

	NL			Cities			Non-urban			Traffic high			Traffic low		
Pollutant	RR	95%	CI	RR	95%	CI	RR	95%	CI	RR	95%	CI	RR	95%	CI
PM_{10}lag 1	1,036	1,025	1,047	1,018	0,995	1,043	1,037	1,025	1,049	1,020	1,001	1,039	1,040	1,019	1,061
BS lag 1	1,040	1,028	1,053				1,040	1,026	1,053				1,039	1,015	1,063
NH_3 lag 1	1,023	1,010	1,037				1,023	1,008	1,037				1,019	0,997	1,041
CO lag 1	1,048	1,031	1,065	1,035	1,003	1,069	1,061	1,037	1,084	1,028	1,003	1,053	1,062	1,022	1,103
NO lag 1	1,018	1,006	1,030	1,012	0,994	1,030	1,020	1,003	1,036	1,011	0,997	1,025	1,003	0,973	1,034
NO_2 lag 1	1,035	1,024	1,046	1,042	1,020	1,064	1,030	1,018	1,043	1,028	1,011	1,045	1,046	1,023	1,069
SO_2 lag 1	1,042	1,020	1,065	1,031	0,997	1,067	1,037	1,012	1,062	1,036	1,009	1,063	1,039	0,996	1,083
O_3 lag 1	1,041	1,024	1,059	1,024	0,985	1,065	1,041	1,023	1,059	1,038	1,007	1,070	1,039	1,008	1,072
SIA lag 1	1,039	1,027	1,052				1,041	1,027	1,055				1,047	1,023	1,071
NO_3 lag 1	1,030	1,020	1,040				1,030	1,019	1,041				1,040	1,021	1,060
SO_4 lag 1	1,028	1,018	1,038				1,030	1,019	1,041				1,041	1,022	1,060

The shaded RR are statistically significant with a 95 CI.

All statistically significant RR were above unity. Significant associations were found for all air pollutants; for PM_{10}, BS, CO, NO_2, SO_2 and secondary aerosol all lags had a statistically significant association with total mortality. NH_3, NO and O_3 were less consistently associated with total mortality, although statistically significant associations were found for different lags as well. Contrary to what is expected, the relative risks in the four major cities or in the 'high traffic' region were not substantially higher than in the other parts of the Netherlands.

A season-specific analysis of total mortality shows that for all pollutants except NO_3 relative risks were higher in the summer than in the winter season (Table A4), and statistically significant RR were above unity in the winter period, with the exception of O_3. For ozone, significant associations were only found for the summer period. In two-pollutant models for total mortality (Table A5) the independent effect of ozone remained, irrespective of what co-pollutant was added to the model. Except when BS was added as co-pollutant, the association between PM_{10} and total mortality remained statistically significant in two-pollutant models. NO_2 and SO_2 also remained statistically significant in two-pollutant models. Most of the models with CO, NH_3 or NO as the co-pollutant resulted in significant negative associations for CO, NH_3 or NO.

Tables A6 to A8 show similar associations between air pollution and daily respiratory mortality in the Netherlands for the different regions, seasons and the two-pollutant models. All statistically significant RR were above unity. Again, significant associations were found for all air pollutants. All relative risks, except one (O_3 lag 1), were larger than the RR for total mortality. Relative risks for the four major cities and the rest of the Netherlands were comparable; substantially higher risks were found for the 'low traffic' region compared with the 'high traffic' region. For all pollutants, higher RR were found in the summer period, with the exception of NO and NO_3. In two-pollutant models all statistically significant ozone associations disappeared; the association between PM_{10}, NO_2, SO_2 and BS (except when PM_{10} was the co-pollutant) and respiratory mortality remained statistically significant, irrespective

of the second pollutant added to the model. CO, NO, SIA, NO_3 and SO_4 were less stable when a co-pollutant was entered into the model.

Table A9 shows the associations between air pollution and daily chronic obstructive pulmonary mortality for different regions of the Netherlands. Except for NH_3, significant associations were found for all pollutants, and all statistically significant RR were above unity, except for two ozone associations in the urbanised region. With the exception of a few associations, larger RR were found overall compared with RR for total mortality; RR comparable with respiratory mortality were also found. In the four major cities and the 'high traffic' regions substantially less significant RR were found than in the non-urban and 'low traffic' areas. All RR were higher for the summer period (Table A10). In two-pollutant models (Table A11) SO_2 remained statistically significant in all models, while PM_{10}, BS and NO_2 remained statistically significant in all but one model. Ozone associations reduced when other pollutants were added to the model. Relative risks for CO, NO, NH_3 and secondary aerosol were less stable when a co-pollutant was entered, while for NH_3 and NO, negative associations were found when a co-pollutant was added.

Results for pneumonia mortality are shown in Table A12. All pollutants were statistically associated with pneumonia mortality, while overall the highest RR were found for this association compared with total and other cause-specific mortality. Again, all statistically significant associations were above unity. RR for the four major cities and 'high traffic' regions were comparable with RR in non-urbanised and 'low traffic' regions. For both seasons (Table A13) statistically significant associations were found. PM_{10} and BS were associated with pneumonia mortality in both seasons. For CO, NO and NO_2, SO_2 and NO_3 most of the winter RR were statistically significant, while for NH_3, secondary aerosol and SO_4 this was the case for most of the summer RR. Ozone and SO_4 were only associated with pneumonia mortality during the summer season. In the two-pollutant model PM_{10}, SO_2 and BS were best associated with pneumonia mortality (Table A14). NO was negatively associated with pneumonia mortality when a co-pollutant was added to the model, while ozone associations reduced to non-significance with the addition of co-pollutants.

The associations between daily cardiovascular mortality and air pollution for different regions in the Netherlands is shown in Table A15. For all pollutants except NH_3, statistically significant associations were found with cardiovascular mortality. The magnitudes of the RR are comparable with the RR for total mortality. In cities and 'high traffic' regions, substantially less significant associations were found compared with rural and 'low traffic' regions. In both seasons statistically significant associations were found for several pollutants, but the associations for the summer period were more consistent. NO, SO_2, O_3 and secondary aerosols were not associated with cardiovascular mortality in the winter season. NO_3 and SO_4 associations disappeared when the analyses were stratified by season (Table A16). In two-pollutant models NO_2, SO_2 and O_3 were most consistently associated with cardiovascular mortality, while NO and secondary aerosols were least stable in the two-pollutant models (Table A17).

3.4.1.4. Discussion and conclusion

One of the aims of the current study was to compare the results of the 1992–1998 period with those of the previous period studied, 1986–1994, with interest focusing particularly on the comparison of the results for PM_{10}, because in the previous analyses it had only been possible

to analyse a relatively short period of three years. Although we have data from 1986 for pollutants other than PM_{10}, we decided that for comparability our new analyses would be restricted to the 'core' PM_{10} period, so comparison was only possible between the 'new' period (1992–1998) and the 'old' period (1992–1994). However, apart from the 'external' comparison we made internal comparisons as well, to see whether dose-response associations in our current analyses differed between the periods 1992–1994 and 1995–1998. This was analysed by incorporating an interaction term for period in the one-pollutant model.

Our current results confirm the conclusions from the previous time-series analyses. All outcome measures revealed significant associations for all components. In addition to the previous analyses we also used NH_3, NO and secondary aerosol (SIA) in our analyses. In two-pollutant models NH_3 and NO remained less stable in the models, indicating that these pollutants are less causally related to the effects than other pollutants and that associations for these pollutants found in one-pollutant models are largely driven by co-linearity with other components. This may also be the case with other pollutants that showed significant associations in one-pollutant models. However, PM_{10}, BS, NO_2 and SO_2 especially remained stable in two-pollutant models. Sarnat showed recently that ambient gaseous pollution concentrations correlate well with personal $PM_{2.5}$ concentrations but not with personal gaseous pollutant concentrations (Sarnat *et al.*, 2001), which provides strong evidence that ambient gaseous concentrations act as surrogates for personal $PM_{2.5}$. This might also be the case for other PM fractions like PM_{10}, but no such data were measured in the Sarnat study. It is, however, worthwhile interpreting the gaseous associations, found in numerous studies, in the light of the $PM_{2.5}$ findings. Buringh *et al.* (2000) have made a plausible case for SO_2 not being the causal factor for mortality, though it is a good surrogate. An exceptional pattern in the two-pollutant models was discernible for the cardiovascular mortality for which NO_2, SO_2 and O_3 were most stable in two-pollutant models. This is an interesting finding, suggesting that respiratory mortality is most associated with PM-related indicators for air pollution and cardiovascular mortality more with gaseous and summer-type-related air pollutants. This finding needs to be more extensively elaborated before firm conclusions can be drawn, however.

In the current study we restricted the analyses to five mortality outcomes: total mortality, respiratory mortality, chronic obstructive lung disease mortality, pneumonia and cardiovascular mortality. Lung-disease-related outcomes (respiratory, COPD and pneumonia) showed higher RR than total mortality and cardiovascular mortality. Relative risks for total mortality and cardiovascular mortality were comparable. The highest risks were found for pneumonia mortality.

The current analyses show overall higher RR for all pollutants than the previous analysis for 1986–1994. Instead of a RR of 1.020 (95% CI: 1.004–1.037) per 100 $\mu g/m^3$ PM_{10} lag 1 (1992–1994), we found a RR of 1.036 (95% CI: 1.025–1.047) in the current analysis (1992–1998), which is almost a doubling of the previous result. Higher RR were not only found for PM_{10}, but for all other pollutants as well. In order to make the most relevant comparisons of the influence of the time period versus model specification, the results for both periods using the same model are presented. For the first period (1992–1994) using the new model, the RR for PM_{10}-associated total mortality became 1.027 (95% CI: 1.012–1.041) per 100 $\mu g/m^3$ and for the second period (1995–1998) it was 1.047 (95% CI: 1.031–1.062) per 100 $\mu g/m^3$. The average concentrations of PM_{10} in the Netherlands were quite similar during both periods, 41.3 $\mu g/m^3$ and 40.0 $\mu g/m^3$ respectively.

The main conclusion is that model differences have resulted in a slight increase in RR for the first period (1992–94), from 1.020 per 100 µg/m^3 of PM_{10} in the old model to 1.027 in the new. The substantial increase in RR found in the second part of the period is, therefore, probably the result of some phenomenon that happened during one of the periods. A decrease in air pollution levels might be one of the reasons. This increase in RR for a period of three years in a time-series analysis might also have some consequences for interpretation of the heterogeneity between studies, as it indicates that even for a relatively large population of 16 million people in a time-series study covering a considerable time period, a difference of a few years in the timing of the analysis can lead to quantitatively different RR estimates. The influence a decrease in air pollution levels may have is illustrated in Figure 3.1 for PM_{10} and in Figure 3.2 for BS, where the smoothed concentration-response curves are presented .

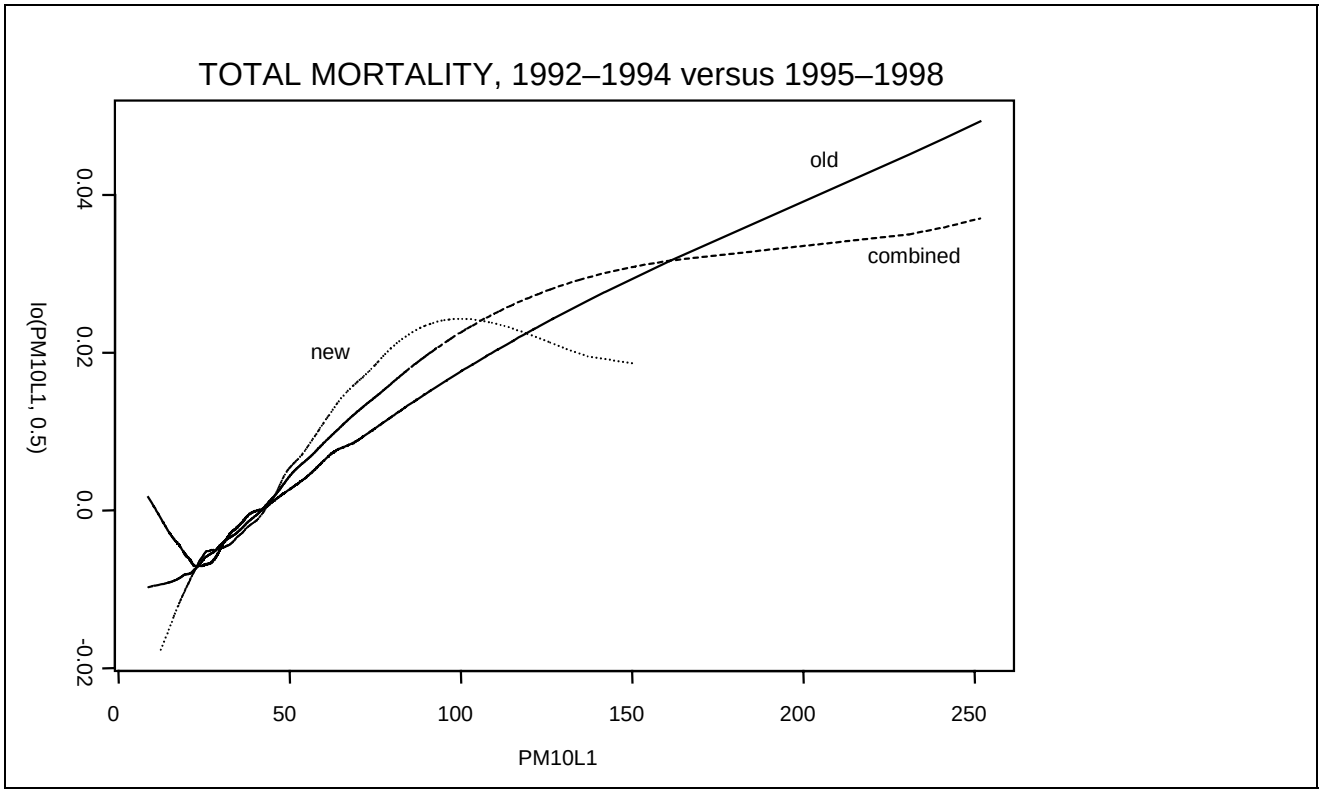


Figure 3.1 Total mortality in 1992–1994 versus 1995–1998, lag 1 for PM_{10}.

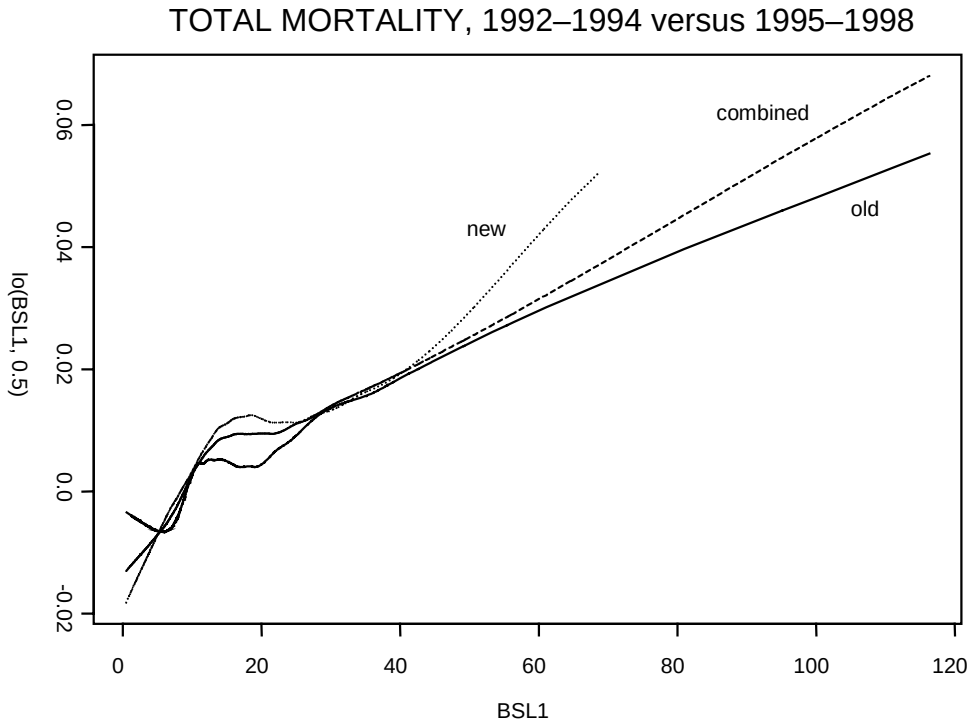


Figure 3.2 Total mortality in 1992–1994 versus 1995–1998, lag 1 for BS.

The figures show the modelled association between mortality and air pollution levels (lag 1). Consistent with the previous study, concentration–response curves were non-linear, with an overall pattern of steeper mortality-concentration associations being seen in the lower concentration range. Since the concentration-response curves are not linear and level off at the higher concentration levels, in the current analyses the steeper slope at the lower concentrations for those pollutants that have decreased in concentration during the study period will have a relatively greater influence on the RR estimate. As the RR is estimated as a linear association, the resulting slope of the line will be steeper. This will also result in a larger RR in the current analyses compared with the RR found in the first time-series study for the pollutants that show a decreasing trend in their ambient levels. However, this is probably not the case for PM_{10}, as the average concentration remained nearly constant during that period, but PM_{10} had rather more high outliers in the first period.

The estimated increase of 3.6% in the daily number of deaths for a 100 $\mu g/m^3$ increase in daily PM_{10} (lag 1) is slightly lower than the estimate from the APHEA2 project, based on the average of the results from 29 European cities, where an increase of 6% (95% CI 4%–8%) was estimated (Katsouyanni, 2001). The APHEA2 estimate is based on a 2-day average PM_{10} concentration (average of lags 0 and 1), whereas our estimate is based on a 1-day lag. It has been found that exposures based on more than 1-day averages are associated with larger effect estimates (Anderson *et al.*, 1997; Zmirou *et al.*, 1998; Spix *et al.*, 1998; Schwartz, J., 2000). In addition, regional differences in the magnitude of the RR have been reported within the European region, with lower RR in eastern European cities. In APHEA2, important effect modifications for several factors were found (Katsouyanni, 2001): in cities with low average NO_2 levels the estimated increase in daily mortality per 100 $\mu g/m^3$ PM_{10} was lower than in cities with high average NO_2 levels; in relatively cold cities lower increases in mortality per increment of PM_{10} were found than in warmer cities; and in cities with low standardised

mortality rates higher increases per increment of PM_{10} were found than in cities with high mortality rates. The NMMAPS study (Samet *et al.*, 2000) in the USA found an overall increase of 5% in the total daily number of deaths per 100 $\mu g/m^3$ increase in PM_{10}, based on a 1-day lag and an average of 90 cities. This estimate is comparable with ours. Regional differences were also found in NMMAPS, with higher estimates in the northeast region. The NMMAPS estimates for PM_{10} at lag 1 in Figure 22 (Samet *et al.*, 2000) varied from 2.5% in the southwest to 7.5% per 100 $\mu g/m^3$ in the northeast, with the other regions lying in between these two extremes. As has been mentioned previously, the exact value of these NMMAPS results is the subject of new calculations.

A recent Dutch time-series study in the city of Amsterdam (Roemer *et al.*, 2001a, b and 2002) showed that people living very close to major roads (with approximately twice the primary traffic contribution compared to an urban background) had a higher RR for all-cause mortality than those living in the average urban background in Amsterdam. Roemer et al (2001b) report that the RR for background PM_{10} at lag 1 was 1.027 per 100 $\mu g/m^3$ for the total population of Amsterdam and 1.049 for the 10% of the population living close to major roads in the city. In a later supplementing letter, Roemer *et al.* (2002) showed that for the population living near major roads the RR becomes nearly unity and non-significant at lag 1 when locally measured BS is used as a tracer instead of background PM_{10}. Nevertheless, BS allows a better exposure characterisation because it has been measured at the traffic sites and at the background sites, unlike PM_{10} which is only measured at urban background sites in Amsterdam. In general, BS is a better proxy for traffic than a mass measure like PM_{10}. A similar decrease in RR with a better exposure characterisation for the traffic group also occurs with NO_2. NO_2 is also more of a proxy for traffic than PM_{10}, and the RR for the traffic population at lag 1 halves when the traffic sites in Amsterdam are used for exposure characterisation instead of the background sites.

However, on a larger geographical scale – at a national level instead of close to major roads – the recent nationwide update of the time-series analyses indicated that the RR for all-cause mortality in the four major cities of the Netherlands is slightly lower (but not statistically significant) than in the more regional part of the rest of the Netherlands. Average PM_{10} levels are almost equal for both situations, whilst the primary PM contributed by traffic is almost doubled in the urban background compared with the regional situation in the Netherlands. If traffic had been the sole source of PM-related health effects, this would have been revealed by a higher RR in the major cities. We found larger and more statistically significant RR for the rural regions and ‘traffic low’ regions compared with the four major cities and the ‘traffic high’ region respectively. A similar trend was found in the previous report, where RR were not substantially larger for the four major Dutch cities. It seems, however, that in the current analyses differences between rural and urban, and ‘traffic low’ versus ‘traffic high’ become more pronounced. In our previous report we indicated that exposure misclassification might be more of a problem in cities with a larger number of local sources of air pollution compared with rural areas. It has been shown that within the city of Amsterdam there is spatial variation in air pollution levels outdoors and indoors, depending on the air pollutant measured. The traffic-related air pollutants especially showed high spatial variability. Another possibility is that the fraction of people dying in hospitals and nursing homes is larger in cities, and both these groups spend less time outdoors than the general population. In Figures 3.3 and 3.4 the concentration-response curves for PM_{10}-associated all-cause mortality are presented for the ‘four large cities’ and ‘regional’ areas in the Netherlands.

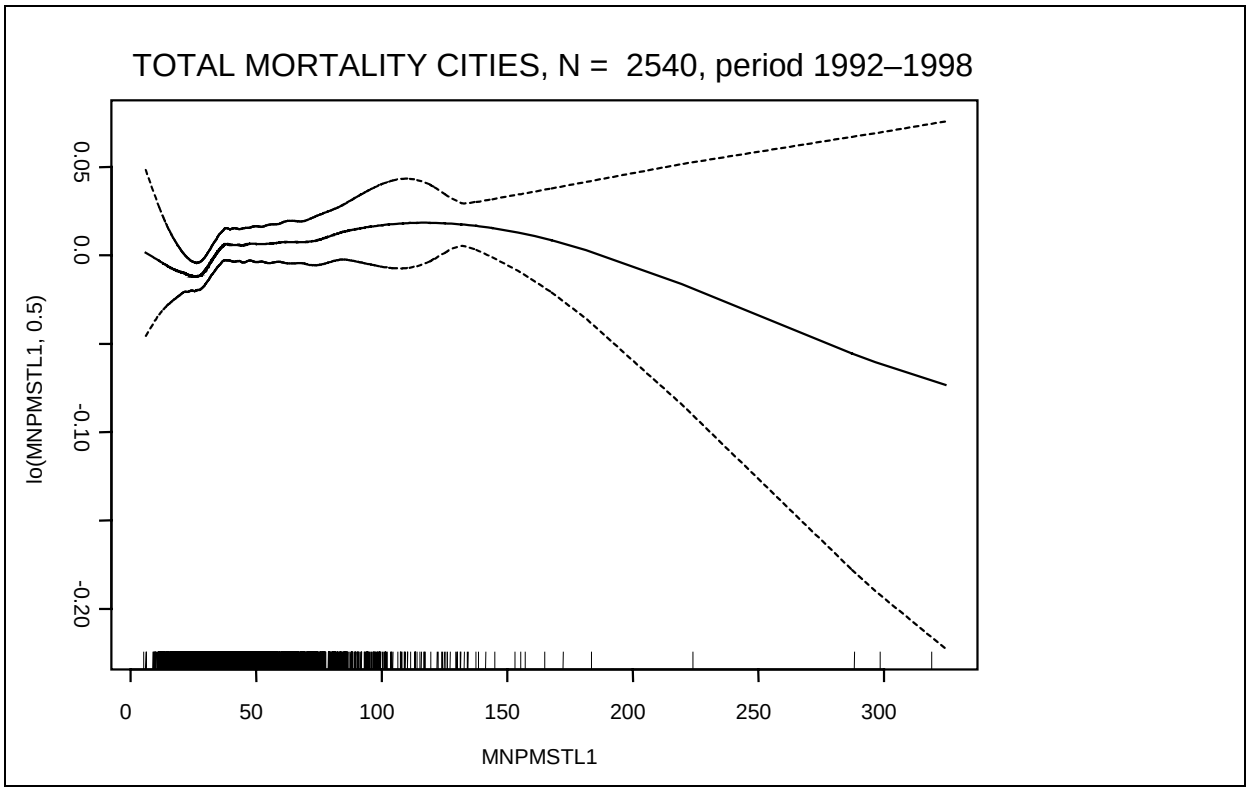


Figure 3.3 Total mortality 1992–1998 and PM_{10} for lag 1 in the four large cities.

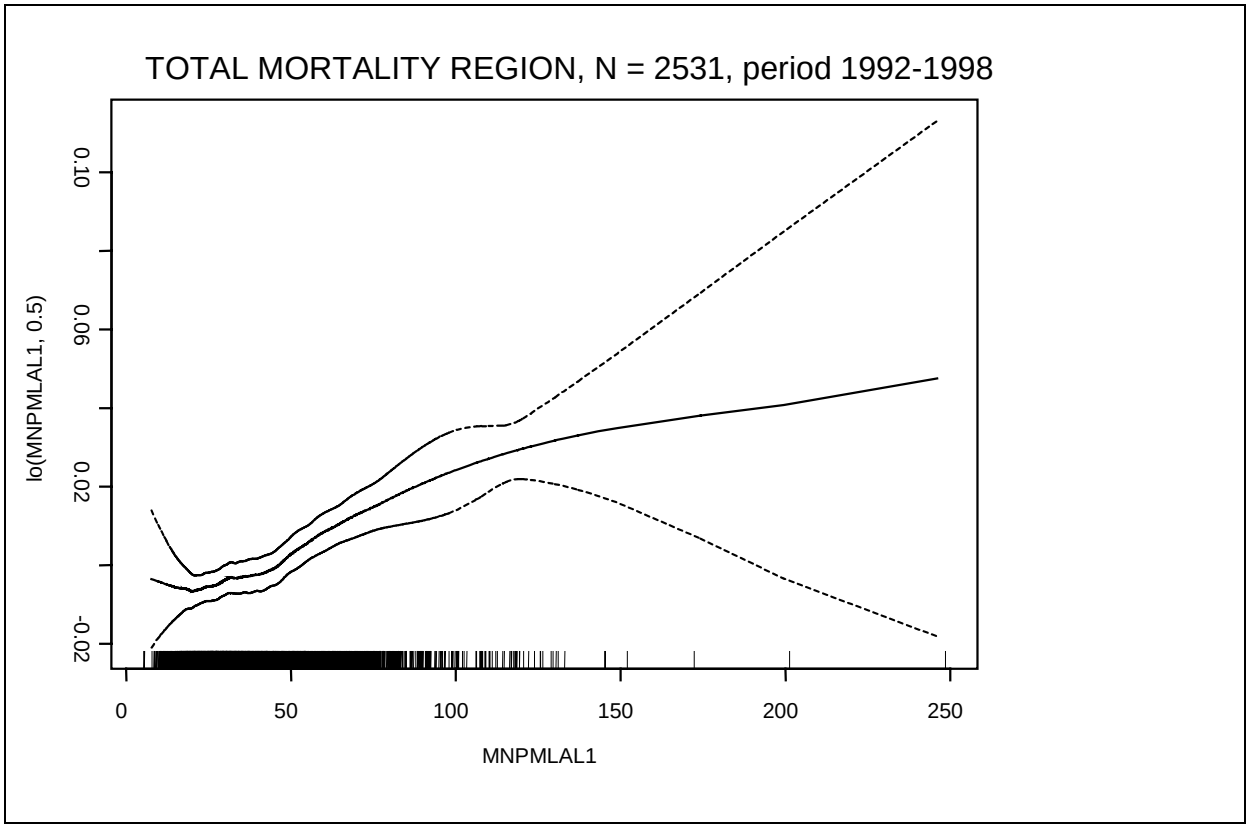


Figure 3.4 Total mortality 1992–1998 and PM_{10} for lag 1 in regional part of the Netherlands.

Principal-component analyses of different fractions of PM in the US point towards traffic as a source of the health-relevant fraction of PM. Other Dutch and foreign studies also provide evidence that health effects are related to the distance from a major road or to traffic density, and to heavy-duty traffic more specifically. The results of a Dutch cohort study (Hoek *et al.*, 2002) clearly point to more health effects at addresses near major roads.

A not yet understood phenomenon is that during the summer season, when PM and gaseous concentrations (apart from ozone) are lower in the Netherlands, the RR become higher (cf. Tables A4, A7, A10, A13 and A16). Hoek and Brunekreef (1999) evaluated whether this phenomenon was an interaction effect of PM with ozone, which it was not, or whether it was caused by pollen. Both authors reported an independent pollen effect in the previous analysis, so pollen has now been included in the new models. Nevertheless, the resulting RR for all pollutants (except nitrate) are higher in summer.

3.4.1.5. Conclusions on the time-series mortality update

Analyses of mortality data of the Dutch population for the period 1992–1998 resulted in a larger RR for all the pollutants studied than for the period 1986(1992)–1994. Differences in the current model specification are part of the reason for this increase in RR. The previously found lack of association concerning secondary inorganic aerosols (SIA) and associations with mortality disappeared in the time-series update.
A confirmation was obtained of the previously found larger RR in the summer season than in winter.
The previous finding that all particulate and gaseous pollutants were associated with short term fluctuations in mortality was confirmed, though there is some evidence that the gaseous pollutants act as surrogates for PM.
For total, respiratory, COPD and pneumonia mortality the most consistent associations were found with PM_{10}, BS, NO_2 and SO_2, while for cardiovascular mortality the most consistent associations were found for NO_2, SO_2 and O_3.
There were no larger RR in the four largest cities or the ‘traffic high’ and urban regions of the Netherlands. Although almost all RR were above unity, most RR in the four largest cities or the ‘traffic high’ region were not statistically significant.
Concentration-response curves suggest non-linear associations which level off at higher concentrations and did not show a threshold below which no effect on mortality was found.

3.4.2. Netherlands update of time-series analyses on hospital admissions

3.4.2.1. Study design

A follow-up study was conducted in order to investigate the short-term relationship between daily air pollution levels and the daily number of emergency hospital admissions in the Netherlands from January 1992 till September 1999 (Vonk and Schouten, 2002). This study is an update of Vonk and Schouten (1998). As this study is new for the Netherlands we will summarise the results. The main pollutant investigated was PM_{10}, but the effects of black smoke, SO_2, NO, NO_2, O_3 and CO were also studied. Hospital admissions due to the following diagnoses at discharge were investigated: all respiratory diseases (ICD-9: 460–519), upper respiratory diseases (ICD-9: 460–465, 470–478), pneumonia and acute bronchitis (ICD-9: 466, 480–486), COPD (ICD-9: 490–492, 494–496), asthma (ICD-9: 493) and

cardiovascular diseases (ICD-9: 393–414, 428, 430–438). The update has been reported in more detail by Vonk and Schouten (2002).

The data were analysed using Poisson regression controlling for trend, season, day of the week, influenza, temperature, relative humidity, air pressure, public and school holidays. Generalised Additive Models (GAM) were used to make the control for these confounders as flexible as possible.

Four single-day lags were evaluated (lag 0, 1, 2, 3) and four averages of lags (lag 0-1, 0-2, 0-3 and 0-6) for all pollutants. The analysis was performed for the whole year and for the warm (May to October) and cold season (November to April) separately. Different age groups (all ages, 0–14 years, 15–64 years and 65+ years) were examined. Concentration-response relationships were plotted for each analysis to see if the relation between air pollution level and number of hospital admissions was linear. Finally, two-pollutant models were specified to investigate the possible confounding effect of pollutants on each other.

3.4.2.2. Results

The results of the 'whole year/all ages' analysis show positive statistically significant associations between all measured air pollution components and the daily number of emergency hospital admissions due to all respiratory diseases, acute bronchitis or pneumonia, COPD and cardiovascular diseases. This means that on days with temporally elevated air pollution concentrations relatively more people are admitted to hospital with any of these mentioned diseases. The relative risks (RR) and 95% confidence intervals (95% CI) associated with an increase in air pollution from the 5th- to the 95th-percentile are summarised in Table 3.3. This table shows only the results of the best-fitting single-day lag, which is the lag with the most significant positive estimate or, if there is no positive estimate, the lag with the least negative estimate. Hospital admissions due to diseases of the upper respiratory system and hospital admissions due to asthma are not clearly influenced by air pollution and are not shown in Table 3.3. Results for O_3 are also not shown in this table because the concentration-response relationship between the daily number of hospital admissions and the daily level of O_3 is not linear.

Table 3.3 Summary of the results of the 'whole year/all ages' analysis.

	respiratory admissions			Acute bronchitis and pneumonia admissions			COPD admissions			cardiovascular admissions		
	lag	RR	95% CI	lag	RR	95% CI	lag	RR	95% CI	lag	RR	95% CI
PM_{10}	3	1.03	(1.02-1.05)	2	1.05	(1.03-1.07)	3	1.06	(1.04-1.08)	0	1.03	(1.02-1.04)
BS	3	1.05	(1.03-1.06)	3	1.05	(1.03-1.07)	3	1.08	(1.06-1.10)	0	1.03	(1.02-1.04)
SO_2	3	1.05	(1.04-1.07)	3	1.06	(1.04-1.08)	3	1.08	(1.06-1.10)	3	1.03	(1.02-1.04)
NO	0	1.03	(1.02-1.04)	0	1.03	(1.02-1.05)	3	1.03	(1.01-1.04)	0	1.02	(1.02-1.03)
NO_2	1	1.05	(1.04-1.07)	3	1.06	(1.03-1.08)	3	1.07	(1.05-1.10)	0	1.04	(1.03-1.05)
CO	3	1.04	(1.02-1.05)	0	1.04	(1.02-1.06)	3	1.05	(1.03-1.07)	0	1.04	(1.03-1.04)

Only RR of the best-fitting single-day lag are shown. RR are for an increase from the 5th- to the 95th-percentile in air pollution: PM_{10}: 63 µg/m^3, BS: 29 µg/m^3, SO_2: 16 µg/m^3, NO: 49 µg/m^3, NO_2: 37 µg/m^3, CO: 660 µg/m^3.

From Table 3.3 it can be seen that the highest estimates are found for admissions due to COPD, and for black smoke (BS) and SO_2. For cardiovascular admissions the effects of air

pollution are mostly on the same day, whereas for respiratory, acute bronchitis and pneumonia, and COPD admissions a longer time interval between exposure and admission is seen. This can be explained by the more acute nature of the cardiovascular diseases compared with the respiratory diseases.

Two-pollutant models with both PM_{10} and black smoke in the model show that for all diagnostic categories the PM_{10} estimate shrinks while the black smoke estimate remains the same. This is an indication that the more traffic-related particles (BS) have a greater influence on health than the mass of the particles (PM_{10}). Other two-pollutant models show that the effects of PM_{10} and black smoke on the number of respiratory, acute bronchitis or pneumonia and COPD admissions are much reduced when SO_2 is included in the model, while the SO_2 estimates remain unchanged.

The results of the analysis stratified by season are summarised in Table 3.4.

Table 3.4 Summary of the results of the 'by season/all ages' analysis.

	Respiratory admissions			Acute bronchitis and pneumonia admissions			COPD admissions			cardiovascular admissions		
	lag	RR	95% CI	lag	RR	95% CI	lag	RR	95% CI	lag	RR	95% CI
Warm season (May-October)												
PM_{10}	2	1.03	(1.00-1.06)	1	1.09	(1.05-1.15)	2	1.06	(1.02-1.10)	0	1.02	(1.01-1.04)
BS	3	1.04	(1.00-1.07)	0	1.06	(1.00-1.11)	3	1.09	(1.04-1.15)	0	1.05	(1.03-1.07)
SO_2	1	1.07	(1.03-1.11)	1	1.11	(1.05-1.17)	3	1.10	(1.05-1.15)	1	1.05	(1.03-1.07)
NO	0	1.03	(1.00-1.07)	0	1.06	(1.01-1.11)	3	1.05	(1.00-1.10)	0	1.05	(1.03-1.07)
NO_2	3	1.02	(1.00-1.05)	0	1.05	(1.01-1.10)	3	1.05	(1.01-1.10)	0	1.04	(1.02-1.06)
CO	0	1.05	(1.02-1.08)	0	1.08	(1.03-1.13)	3	1.04	(1.00-1.09)	0	1.05	(1.03-1.07)
Cold season (November-April)												
PM_{10}	3	1.03	(1.02-1.05)	2	1.03	(1.00-1.05)	3	1.06	(1.03-1.08)	0	1.02	(1.01-1.03)
BS	3	1.04	(1.03-1.06)	3	1.03	(1.01-1.06)	3	1.07	(1.05-1.09)	0	1.03	(1.02-1.04)
SO2	3	1.05	(1.03-1.06)	3	1.04	(1.02-1.07)	3	1.07	(1.05-1.09)	3	1.03	(1.02-1.04)
NO	1	1.02	(1.01-1.04)	0	1.03	(1.01-1.05)	1	1.03	(1.01-1.05)	0	1.02	(1.01-1.03)
NO2	3	1.04	(1.02-1.06)	3	1.03	(1.00-1.06)	3	1.06	(1.04-1.09)	0	1.04	(1.02-1.05)
CO	3	1.03	(1.01-1.05)	0	1.02	(1.00-1.05)	3	1.04	(1.02-1.06)	0	1.03	(1.02-1.04)

Only RR of the best-fitting single-day lag are shown. RR are for an increase from the 5th- to the 95th-percentile in air pollution: PM_{10}: 63 μg/m^3, BS: 29 μg/m^3, SO_2: 16 μg/m^3, NO: 49 μg/m^3, NO_2: 37 μg/m^3, CO: 660 μg/m^3.

In general, the relative risk of the effects of air pollution on the number of hospital admissions are slightly higher in the warm season, but with the exception of O_3 the pattern of the air pollution effects does not differ between the two seasons. The linear effect estimates for O_3 for admissions due to respiratory diseases, acute bronchitis or pneumonia, and COPD are positive in the warm season and negative in the cold season. Inspection of the concentration-response relationships reveals that this relationship is not linear; there is a threshold below which no effect or a negative effect of O_3 is seen. This threshold is located between 50 and 100 μg/m^3. Since O_3 levels are higher in the warm season, the linear effect estimates of O_3 will be more influenced by the values above this threshold and therefore will be positive in the

warm season. In the cold season most O_3 values are below the threshold and no effect or a negative effect of O_3 on the number of hospital admissions is seen.

Concentration-response relationships of the other pollutants are mostly linear, but some of the relationships show a very steep increase in the number of admissions at the lower air pollution levels followed by a less steep increase at the higher levels of air pollution (e.g. PM_{10} or SO_2 and pneumonia admissions). In other words, these concentration-response relationships level off at higher concentrations.

Table 3.5 shows the results of the analysis in the three different age groups (0–14, 15–64, 65+). The number of cardiovascular admissions in the youngest age group (0–14) is very low, so the effect of air pollution is not estimated.

Table 3.5 Summary of the results of the 'whole year/by age group' analysis for three different age groups (0–14, 15–64, 65+).

	respiratory admissions			Acute bronchitis and pneumonia admissions			COPD admissions			cardiovascular admissions		
	lag	RR	95% CI	lag	RR	95% CI	lag	RR	95% CI	lag	RR	95% CI
0-14												
PM_{10}	1	1.00	(0.98-1.02)	3	1.01	(0.98-1.04)	1	0.99	(0.93-1.05)			
BS	1	1.02	(1.00-1.04)	0	1.04	(1.01-1.07)	0	1.03	(0.96-1.10)			
SO_2	3	1.04	(1.02-1.06)	3	1.06	(1.03-1.10)	0	1.03	(0.97-1.11)			
NO	0	1.03	(1.01-1.05)	0	1.05	(1.03-1.08)	0	1.03	(0.97-1.09)			
NO_2	1	1.04	(1.01-1.07)	3	1.07	(1.03-1.11)	0	1.07	(0.99-1.16)			
CO	0	1.03	(1.01-1.05)	0	1.06	(1.03-1.10)	0	1.04	(0.97-1.11)			
15-64												
PM_{10}	3	1.03	(1.01-1.05)	1	1.06	(1.03-1.10)	2	1.05	(1.01-1.09)	1	1.03	(1.02-1.05)
BS	3	1.04	(1.02-1.06)	3	1.04	(1.00-1.07)	3	1.08	(1.05-1.12)	0	1.03	(1.01-1.04)
SO2	3	1.03	(1.01-1.05)	1	1.03	(1.00-1.07)	3	1.07	(1.03-1.10)	1	1.03	(1.02-1.05)
NO	3	1.02	(1.01-1.04)	1	1.03	(0.99-1.06)	3	1.06	(1.03-1.09)	0	1.01	(1.00-1.02)
NO2	1	1.04	(1.02-1.06)	1	1.09	(1.05-1.13)	1	1.09	(1.04-1.13)	0	1.04	(1.02-1.05)
CO	3	1.04	(1.02-1.06)	1	1.05	(1.01-1.09)	3	1.07	(1.03-1.11)	0	1.02	(1.01-1.04)
65+												
PM_{10}	3	1.05	(1.03-1.07)	2	1.07	(1.04-1.09)	3	1.07	(1.05-1.10)	0	1.03	(1.02-1.04)
BS	3	1.06	(1.04-1.07)	3	1.05	(1.03-1.08)	3	1.09	(1.06-1.11)	0	1.04	(1.02-1.05)
SO2	3	1.07	(1.05-1.09)	2	1.08	(1.05-1.11)	3	1.10	(1.07-1.12)	3	1.03	(1.02-1.04)
NO	0	1.02	(1.01-1.04)	0	1.03	(1.01-1.06)	3	1.02	(1.00-1.04)	0	1.03	(1.02-1.04)
NO2	3	1.06	(1.04-1.08)	0	1.08	(1.05-1.12)	3	1.09	(1.06-1.12)	0	1.04	(1.03-1.06)
CO	0	1.05	(1.03-1.06)	0	1.05	(1.02-1.08)	3	1.05	(1.03-1.07)	0	1.04	(1.03-1.05)

Only RR of the best-fitting single-day lag are shown. RR are for an increase from the 5th- to the 95th-percentile in air pollution:
PM_{10}: 63 µg/m^3, BS: 29 µg/m^3, SO_2: 16 µg/m^3, NO: 49 µg/m^3, NO_2: 37 µg/m^3, CO: 660 µg/m^3.

The highest effect estimates are found in the '65+' group and for the COPD admissions. It is striking that in the youngest age group no effect of PM_{10} is seen.

3.4.2.3. Conclusions on the Dutch hospital admissions update

All investigated pollutants are related to the daily number of emergency hospital admissions due to all respiratory diseases, acute bronchitis or pneumonia, COPD, and cardiovascular diseases. Emergency hospital admissions due to upper respiratory diseases and asthma are not clearly influenced by air pollution. The highest estimates are found for admissions due to COPD, for black smoke and SO_2 in the oldest age group and in the warm season. For cardiovascular admissions the effects of air pollution are mostly on the same day, whereas for respiratory, acute bronchitis and pneumonia and COPD admissions a longer time interval between exposure and admission is seen.

Two-pollutant models with both PM_{10} and black smoke suggest that the more traffic-related particles (i.e. black smoke) have more influence on health than the mass of particles. Other two-pollutant models show that the effects of PM_{10} and black smoke disappear when SO_2 is included in the model, while the estimates of SO_2 remain unchanged. The concentration-response relationship between O_3 and the number of admissions is not linear; there is a threshold below which no effect or a negative effect of O_3 is seen. This ozone threshold is located between 50 and 100 $\mu g/m^3$. In the youngest age group no effect of PM_{10} on the number of admissions is seen.

3.4.3. Assessing determinants of heterogeneity in study results

The heterogeneity of effects across study sites offers the potential to identify factors that cause heterogeneity and thus provide valuable insights into the mechanisms by which PM causes adverse health effects.

In the multi-city studies in the US (NMMAPS study; Samet *et al.*, 2000) and Europe (APHEA2 study; Katsouyanni, 2001) special attention was paid to the heterogeneity issue. In the NMMAPS study, in which up to 90 US city data were analysed for effects on mortality of PM_{10} and other pollutants, some differences in the PM_{10} effect were seen by region of the US: for the 90 cities the largest RR were found in the northeast. Factors assessed were ‘mean level of other pollutants’, ‘median income’, ‘percentage of adults without high school diploma’, ‘percentage of households below poverty level’, ‘percentage of persons using public transportation’ and ‘crude mortality rate’. The investigators did not identify any factor or factors that might explain these differences. In a later analysis of the NMMAPS morbidity study it was found that effects of PM were stronger in areas where the PM contribution from traffic was high (Janssen *et al.*, 2002). Some of the heterogeneity within the APHEA study of the effects of air pollution between western and central-eastern cities were caused by the statistical approach used and the inclusion of days with pollutant levels above 150 $\mu g/m^3$. Within the APHEA2 study, important effect modifiers were found to be long-term average NO_2 concentration, the ratio of PM_{10} to NO_2, followed by temperature, humidity, age-standardised mortality, the proportion of the elderly, and geographic area. The differences in relative risk for two periods in the time-series of Dutch mortality (1.027 versus 1.047, cf. 3.4.1.4) cannot be explained by differences in long-tem average NO_2 or the ratio of PM_{10} to NO_2, humidity, age-standardised mortality or the proportion of elderly in the Netherlands. This indicates that these effect modifiers, which offer an explanation for the APHEA2 data, need to be interpreted with some caution for a different time and spatial domain.

In the Netherlands we also tried to gain a better understanding of the relevant variables that cause heterogeneity in study results. We therefore assessed the scientific literature up to 2000 and selected all the studies in which PM_{10} was **measured** and in which its association with mortality was assessed. Studies in which PM_{10} or $PM_{2.5}$ data were calculated from other PM indices (like TSP, BS or Coefficient of Haze) were not selected for analysis. A number of potentially relevant variables in the selected papers were assessed. No US multi-city studies were evaluated in this inventory because the results were not available at that time. The variables used were continent, total population size, age distribution (< 5 yrs, > 65 yrs), season, gender and race distribution, mortality rate, the average background levels of the air pollution components PM_{10}, $PM_{2.5}$, SO_2, O_3, NO_2 , CO, and type of statistical model. Table 3.10 gives an overview of the availability of this information in the specific studies. An * indicates when information on a variable was available. As the table shows, from the 32 studies that were selected none of the variables of interest were sufficiently distributed over the different studies to allow any meaningful analysis. As the multi-city studies have meanwhile been published and as heterogeneity has been assessed in these studies, the added value of this multi-study inventory is limited. It was therefore concluded that there is currently insufficient data to explain intercontinental differences in PM_{10} heterogeneity. It was further concluded that the feasibility of a 'regression-type' approach, as was intended, is very low due to the high diversity of the information available in the different studies. A multi-city or multi-country type of approach with standard analytical protocols and with ad hoc selected variables to include in the heterogeneity models is therefore a more efficient approach.

3.5. Quantification of the estimated health risks

3.5.1. Health risks for PM

One of the purposes of our study was to estimate the actual number of premature deaths in the Netherlands associated with current air pollution levels. We used the relative risks from our time series analyses, the daily numbers of deaths, and the current PM_{10} levels to calculate the numbers of deaths in the Netherlands attributable to PM_{10}. Table 3.6 shows a recapitulation of the study's main results. The results are presented as Relative Risks (RR) per 100 $\mu g/m^3$ PM_{10}. We used the lag 1 RR in the calculation in order to make the calculations comparable with a previous estimation of the PM_{10}-attributable mortality.

Table 3.6 Relative Risk for total and cause-specific mortality in the Netherlands per 100 $\mu g/m^3$ PM_{10} at lag 1

Cause of death	Relative Risk per 100 $\mu g/m^3$ PM_{10}	95% CI
Total mortality	1.036	1.025–1.047
Respiratory	1.120	1.084–1.157
COPD	1.111	1.064–1.161
Pneumonia	1.121	1.065–1.180
Cardiovascular	1.025	1.009–1.042

Based on the relative risks for PM_{10} and on the average PM_{10} concentration in the Netherlands in 2000 (33 $\mu g/m^3$), estimations can be made of the number of deaths in the Netherlands attributable to PM_{10}. The results are presented in Table 3.7.

Table 3.7 Estimated annual number of deaths in the Netherlands associated with exposure to PM_{10}

Cause of death	Number of deaths per year	95% Confidence Interval
Total	**1700**	1150–2200
Respiratory	**650**	470–870
COPD	**250**	140–350
Pneumonia	**250**	140–400
Cardiovascular	**400**	150–680

Bold numbers indicate a statistically significant effect

As can be seen from Table 3.7, we have estimated that on average the death of approximately 1700 people in the Netherlands per year is associated with exposure to air pollution (PM_{10}).

Parallel to the mortality analyses, associations between hospital admissions throughout the Netherlands and air pollution were analysed by the University of Groningen (Vonk and Schouten, 1998 and 2002). Statistical models were comparable with the models used in the mortality analyses. Based on the relative risks, the number of hospital admissions for respiratory and a number of other causes were calculated; the results are presented in Table 3.8.

Table 3.8 Estimated annual number of hospital admissions in the Netherlands associated with exposure to PM_{10}. Based on data from Vonk and Schouten (2002).

Cause of admission	Number of admissions per year	95% CI
Respiratory	**1000**	550–1450
Cardiovascular	**1350**	800–1900
COPD	**550**	350–750

Bold numbers indicate a statistically significant effect

It was estimated that each year in the Netherlands more than 2000 hospital admissions for respiratory or cardiovascular causes and approximately 1000 deaths from these causes were associated with exposure of the Dutch population to PM_{10}. In general the results of these two Dutch studies corroborate foreign studies reported in the scientific literature.

In conclusion, relative risks estimated for hospital admissions and daily mortality were generally positive, statistically significant and consistent with previously reported associations.

3.5.2. Health risks for other pollutants

Table 3.9 gives the associations for other air pollutants from the update of the most recent Dutch time-series study. The relative risks (RR) are expressed per relevant concentration range. Statistically significant associations are presented in bold.

Table 3.9 Relative Risk for total and cause-specific mortality in the Netherlands (1992–1998) for other air pollutants (lag 1).

Pollutant	per $\mu g/m^3$	Total mortality	Respiratory mortality	COPD mortality	Pneumonia mortality	Cardiovasc. mortality
PM_{10}	100	**1.036**	**1.120**	**1.111**	**1.121**	**1.025**
BS	50	**1.040**	**1.121**	**1.089**	**1.147**	**1.032**
NH_3	30	**1.023**	**1.065**	1.018	**1.116**	1.015
CO	1,500	**1.048**	**1.145**	**1.084**	**1.224**	**1.044**
NO	100	**1.018**	**1.056**	1.012	**1.104**	1.017
NO_2	50	**1.035**	**1.107**	**1.063**	**1.141**	**1.029**
SO_2	50	**1.042**	**1.191**	**1.118**	**1.195**	**1.029**
O_3	150	**1.041**	1.011	1.046	1.035	**1.032**
SIA	30	**1.039**	**1.073**	**1.071**	**1.088**	**1.035**
Nitrate	10	**1.030**	**1.062**	**1.071**	1.051	**1.025**
Sulphate	10	**1.028**	**1.058**	**1.073**	**1.052**	**1.018**

Bold numbers indicate a statistically significant effect

Table 3.9 shows that other major air pollutants (both particulate and gaseous) are, like PM_{10}, associated with mortality in the Netherlands. All air pollution components, except ozone, correlate highly in the Netherlands (Van der Wal and Janssen, 2000). Meteorology is, of course, the driving force in this correlation, resulting in a simultaneous daily increase or decrease of pollutants in the mixture of air pollution. Owing to this interdependency of the various pollutants, it remains questionable whether the mortality effects as found in relation to PM_{10} can also be attributed specifically to ambient PM_{10} or to a part of PM_{10}. The generally moderate to high correlation among air pollutants makes it problematic in statistical analyses to separate effects from specific components of the air pollution mixture. Despite these difficulties some efforts were made to analyse the independent effects of single pollutants by using models in which two air pollutants were analysed simultaneously. The results of these analyses showed that PM_{10} associations were influenced when gaseous components were added into the model. The previously discussed finding of Sarnat *et al.* (2001) indicates that the gaseous pollutants act as a surrogate and that particles were more consistently associated with mortality. Some remarks on two-pollutant models and the PM-associated health effects have already been made in 3.4.1. It could even be hypothesised that if the causal factor(s) is or are only a part of the PM (whether $_{10}$ or $_{2.5}$), as suggested by the factor analysis reported in 3.1.1.1 and in the toxicology sections 4.3.2.3.a and c, this causal factor probably has a high correlation with PM_{10} as well. Because this correlation will never be perfect, part of the correlation will also turn up with the gaseous components independently of PM_{10}. It could be hypothesised that in two-pollutant models this part of the causal PM fraction might appear to show up as a PM-independent gaseous correlation. If it could be demonstrated that such an

explanation was valid, it could accommodate the current results of the two-pollutant models and those of Sarnat *et al.* (2001).

Because of the difficulties caused by correlation between pollutants, we opted for an approach in which the different air pollutants are evaluated in terms of plausibility of causality. This will be based on the temporal and spatial correlation and different indoor/outdoor ratios of the various gaseous and particulate components of PM_{10}. An example of this approach was recently presented by Buringh *et al.* (2000). This study investigated associations between SO_2 and daily mortality at different levels of exposure and found the effects to be stronger (per unit concentration) at lower levels. The study then showed that in geographic areas with low levels the effects were in fact smaller (per unit) than in areas with high levels, suggesting that SO_2 in this case is a surrogate rather than an active ingredient. These analyses circumvent the above described co-linearity that often exists between PM and gaseous co-pollutants. An analysis similar to the one for SO_2 could in principle be carried out for NO_2, ozone and CO as well. Of course, BS is part of PM and might be indicating some 'sooty' part of PM like EC or OC that might be more specifically responsible for the health effects.

In a continuation of the mortality study, Hoek *et al.* (1999) examined interactions between pollutants (ozone and PM_{10} in summer), confounding by airborne pollen in the summer, and the associations between secondary aerosol components sulphate and nitrate (SIA) with daily mortality. It was concluded that there was no interaction between ozone and other pollutants, that pollen did not confound the associations between air pollution and mortality, and that significant associations were found between aerosol sulphate and nitrate and daily mortality

*Table 3.10 Epidemiological studies on the association between mortality and **measured** PM_{10} and the availability of additional information on study characteristics.*

Author	Location	Continent	Total population	> 65	< 5	Season	Sex	Racial	Total mortality	Resp. mortality	Card.vasc. mortality
Dockery *et al.*, 1992	St. Louis,	NA	*						*		
	Kingston-Harriman	NA	*						*		
Fairly, 1990	Santa Clara County	NA	*			*			*	*	*
Styer et al.,	Cook County, Ill.	NA	*	*			*	*	*	*	*
	Salt Lake County, Utah	NA	*	*							
Pope *et al.*, 1999	Ogden	NA	*						*	*	*
	Salt Lake City	NA	*						*	*	*
	Provo / Orem	NA	*						*	*	*
Gwynn *et al.*, 2000	Buffalo, New York	NA	*						*	*	*
Mar *et al.*, 2000	Phoenix	NA		*					*		*
Ito and Thurston, 1996	Cook County, Ill.	NA	*				*	*	*	*	*
Pope *et al.*, 1992	Utah	NA	*						*	*	*
Ito et al.,	Cook County	NA	*			*			*		
	LA County	NA									
Schwartz, 1993	Birmingham, Alabama	NA	*						*	*	*
Ostro *et al.*, 1999	Coachella Valley, CA	NA	*	(*; > 50)		*			*	*	*
Klemm and Mason, 2000	Atlanta	NA	*	*					*		
Chock *et al.*, 2000	Pittsburgh	NA		(*; > 75)		*			*		
Lipfert *et al.*, 2000c	Philadelphia	NA	*	*					*	*	*
Laden *et al.*, 2000	6 cities (Watertown, Kingston-Harr., St. Louis, Steubenville, Portage, Topeka)	NA	*						*		
Moolgavkar, 2000	3 US counties (Cook, LA, Maricopa)	NA	*						*	*	*

*Table 3.10 cont'd: Epidemiological studies on the association between mortality and **measured** PM_{10} and the availability of additional information on study characteristics.*

Author	Location	Continent	Total population	> 65	< 5	Season	Sex	Racial	Total mortality	Resp. mortality	Card.vasc. mortality
Schwartz, 2000	10 US cities (New Haven, Pittsburgh, Birmingham, Detroit, Canton, Chicago, Minneapolis-St.Paul, Colorado Springs, Spokane, Seattle)	NA				*			*		
Daniels *et al.*, 2000	20 largest US cities (partly same studies as above)	NA	*	*					*	*	*
Spix and Wichman, 1996	Cologne, Germany	WE	*						*		
Bremner *et al.*, 1999	Greater London, UK	WE	*	*					*	*	*
Wordley *et al.*, 1997	Birmingham, UK	WE	*						*	*	*
Zmirou *et al.*, 1996	Lyon, France	WE	*						*	*	*
Dab *et al.*, 1996	Paris, France	WE	*							*	
Ponka *et al.*, 1998	Helsinki, Finland	WE	*	*					*		*
Verhoeff *et al.*, 1996	Amsterdam, Netherlands	WE	*	*		(*)			*		
Hoek *et al.*, 2000	Netherlands	WE	*						*	*	*
Peters, Skorkovsky, Kotesovec, Brynda	Coal Basin, Czech	EE	*						*		
Ostro *et al.*, 1996	Santiago, Chile	SA	*	(*)			(*)		* (*)	(*)	(*)
Saldiva *et al.*, 1995	São Paulo, Brazil	SA		*		*			*		
Cifuentes *et al.*, 2000	Santiago, Chile	SA	*			*			*		
Gouveia, Fletcher, 2000	São Paulo, Brazil	SA	*	*	*				*	*	*
Hong *et al.*, 1999	Inchon, South Korea	A	*						*	*	*

*Table 3.10 cont'd: Epidemiological studies on the association between mortality and **measured** PM_{10} and the availability of additional information on study characteristics.*

Author	Location	RR (tot. mort.) (per 50 µg/m^3)	PM_{10}	$PM_{2.5}$	O_3	SO_2	NO_2	CO	Lag time	Remarks	Analysis
Dockery *et al.*, 1992	St. Louis,	1.08	*	*	*	*	*		1		Poisson
	Kingston-Harriman	1.08	*	*	*	*	*		1		Poisson
Fairly, 1990	Santa Clara County	0.99 / 1.08 (#)	*	*	*		*	*	0-1	(#)=lag 1 / lag 0	Poisson / GAM
Styer et al.,	Cook County, Ill.		*						2--3--5		Poisson- semi param. anal.
	Salt Lake County, Utah		*								
Pope *et al.*, 1999	Ogden	1.08	*						5 mov av		Poisson
	Salt Lake City	1.04	*						5 mov av		
	Provo / Orem	1.05	*						5 mov av		
Gwynn *et al.*, 2000	Buffalo, New York	1.05 / 1.12 (#)	*		*	*	*	*	0-2	filled / unfilled data	Poisson / GAM
Mar *et al.*, 2000	Phoenix	1.06 (#)	*	*	*	*	*	*	0-1	> 65	Poisson
Ito and Thurston, 1996	Cook County, Ill.	1.03	*		*	*		*	0 / 1		Poisson / GAM
Pope *et al.*, 1992	Utah	1.08	*						5-d moved av		Poisson
Ito et al.,	Cook County	1.03	(*)						0		
	LA County										
Schwartz, 1993	Birmingham, Alabama	1.05	*						3 av		Poisson
Ostro *et al.*, 1999	Coachella Valley, CA	1.02 (#)	*			*		*	*	0-1-2-3-	lag 1

									mov3/5		

*Table 3.10 cont'd: Epidemiological studies on the association between mortality and **measured** PM_{10} and the availability of additional information on study characteristics.*

Author	Location	RR (tot. mort.) (per 50 µg/m^3)	PM_{10}	$PM_{2.5}$	O_3	SO_2	NO_2	CO	Lag time	Remarks	Analysis
Klemm and Mason, 2000	Atlanta		*	*	*	*	*	*	0-1-mov2		Poisson / GAM
Chock *et al.*, 2000	Pittsburgh		*	*		*	*	*	*	0-1-2-3	
Lipfert *et al.*, 2000c	Philadelphia		*	*	*	*	*	*	0/1		
Laden *et al.*, 2000	6 cities (Watertown, Kingston-Harr., St. Louis, Steubenville, Portage, Topeka)			*					2 day av		Poisson
Moolgavkar, 2000	3 US counties (Cook, LA, Maricopa)		*	(*)	*	*	*	*	0-1-2-3-4-5		Poisson / GAM
Schwartz, 2000	10 US cities (New Haven, Pittsburgh, Birmingham, Detroit, Canton, Chicago, Minneapolis-St.Paul, Colorado Springs, Spokane, Seattle)		*		(*)	(*)		(*)	2 day av		Poisson
Daniels *et al.*, 2000	20 largest US cities (partly same studies as above)		*						0-1		log linear GAM
Spix and Wichman, 1996	Cologne, Germany	1.02									
Bremner *et al.*, 1999	Greater London, UK	1.01	*		*	*	*	*	0-1-2-3		Poisson
Wordley *et al.*, 1997	Birmingham, UK	1.06	*		(*)	(*)	(*)		1		multivariate linear regression

Zmirou *et al.*, 1996	Lyon, France	1.01	(*)		*	*	*		0-2	Poisson

*Table 3.10 cont'd: Epidemiological studies on the association between mortality and **measured** PM_{10} and the availability of additional information on study characteristics.*

Author	Location	RR (**tot**. mort.) (per 50 µg/m^3)	PM_{10}	$PM_{2.5}$	O_3	SO_2	NO_2	CO	Lag time	Remarks	Analysis
Dab *et al.*, 1996	Paris, France	1.08 (#)	(*)		*	*	*		0/1	only resp. mort.; log PM13	Poisson, with log (PM13)
Ponka *et al.*, 1998	Helsinki, Finland		*		*	*	*		0-1-2-3-4-5-6-7		Poisson
Verhoeff *et al.*, 1996	Amsterdam, Netherlands	1.03 / 1.01 / 0.99	*		*	*		*	0-1-2	lag 0 / 1 / 2	Poisson
Hoek *et al.*, 2000	Netherlands	1.01	*		*	*	*	*	1-- av 6		Poisson/GAM
Peters, Skorkovsky, Kotesovec, Brynda	Coal Basin, Czech	1.02 / 1.05 / 0.99	*	*	*	*	*	*	0-1-2	lag 0 / 1 / 2	Poisson, with log (PM10)
Ostro *et al.*, 1996	Santiago, Chile	1.06	*		*	*	*		0		univariate / Poisson
Saldiva *et al.*, 1995	São Paulo, Brazil	1.06 (#)	*		*	*	(*)	*	2-d moving	>65	Gaussian / Poisson/GAM
Cifuentes *et al.*, 2000	Santiago, Chile		*	*	*	*	*	*	0-1-2-3-4-5 / av 2-3-4-5		Poisson / GAM
Gouveia, Fletcher, 2000	São Paulo, Brazil		*		*	*	*	*	0-1-2		Poisson
Hong *et al.*, 1999	Inchon, South Korea	1.04	*		*	*	*	*	1-5 mov av		Poisson / GAM

3.6. Relevance for PM as a proxy or causal factor

There is a general tendency in the epidemiological literature to conclude that ambient PM is associated with all kinds of health effects, especially in the respiratory and cardiovascular system. Although there was scepticism in the early 1990s about the validity of these associations, nowadays there is overall agreement that the observed associations are consistent, coherent and in agreement with Hill's postulates about causality (Dab *et al.*, 2001). Some studies were performed explicitly to re-analyse epidemiological studies in order to evaluate the robustness of model specifications and confounding factors. None of these re-analyses have lead to rejections of the previously published associations, although effect estimates may differ slightly between the original study and the re-analysis.

Hill (1965) formulated a number of postulates in order to establish a causal relation between environment and disease. An attempt can be made to use these to answer the question of whether PM is causal or a proxy. However, Hill's criteria are not very hard and for PM they can be considered to have been met. This gives us no clues as to the causal fractions that need to be abated. Rothman and Greenland (1998) concluded their chapter on causation and causal inference in 'Modern Epidemiology 2' with the statement that some authors argue that using a checklist of causal criteria might actually cloud the inferential process. Another approach is to transform the criteria into deductive tests of causal hypotheses, thus avoiding the temptation to use causal criteria to buttress pet theories and instead allowing scientists to evaluate competing causal theories using crucial observations, as Rothman and Greenland remark.

The question of causal fractions could in principle be answered by conducting large-scale experiments, eradicating the presumed causal fraction and evaluating whether the health effects have disappeared. Such experiments are not easy to perform and whenever they are, they are mostly the result of some historical event and not of a planned social science experiment.

Two recent events which have been reported in peer reviewed literature were the result of such large-scale experiments and could be evaluated epidemiologically and toxicologically. Recent experiments are necessary for this type of analysis because a generalisation of the historical events of more than half a century ago in the Meuse Valley, Donora or London to current ambient levels of PM is a fallacy. Two large experiments that can be mentioned are Utah Valley, where a heavily polluting steel mill was closed for over a year in 1986–1987, and the reunification of Germany in 1989, after which a great number of polluting industries were shut down and air pollution decreased in the former German Democratic Republic. The underlying question is whether these decreases in ambient pollution also resulted in a subsequent decrease in health effects.

The short answer is yes, reducing air pollution results in fewer health effects. Frampton *et al.* (1999) indicated that in **Utah Valley** the principal point source of particle pollution was an integrated steel mill built during World War II in a valley that has frequent inversions in winter time. When in operation this mill contributed 50–70% of total Utah Valley PM_{10} emissions. A labour dispute caused the mill to be closed for a 13-month period in 1986–1987. While it was closed during the winter of 1986–87, the PM concentrations were reported to have been approximately one half of the levels with the steel mill in operation. Frampton *et al.* (1999) reported that the reductions in PM_{10} levels were accompanied by an improvement in

health indicators among the local population, including decreases in 1) daily mortality, 2) age-adjusted death rates for malignant and non-malignant respiratory diseases, 3) respiratory hospital admissions for pneumonia, pleurisy, bronchitis and asthma, 4) bronchitis and asthma admissions for preschool-age children, 5) pulmonary function abnormalities and 6) elementary school absences. For Utah Valley the lucky coincidence was that afterwards pooled filters for the years 1986, 1987 and 1988 could be analysed toxicologically. These analyses generally supported the original epidemiological findings. These results and the link between epidemiology and toxicology are discussed in 4.5.2, as is the question of the extent to which the Utah Valley situation can be generalised to the ambient PM situation in the Netherlands.

Heinrich *et al.* (2000) described the decline in air pollution and respiratory symptoms in children in former **Eastern Germany**. More than two thousand children in three areas participated in a regional cross-sectional study. While annual average TSP levels in the three areas fell from 65–44 $\mu g/m^3$ in 1993 to 43–36 $\mu g/m^3$ in 1995 and to 33–25 $\mu g/m^3$ in 1998, the age-adjusted prevalence of bronchitis decreased from 54% in 1993 to 41% in 1995 and 38% in 1998. For the prevalence of bronchitis, otitis media, frequent colds and febrile infection the odds ratios (OR) decreased significantly after adjustment for several potential predictors. In conclusion, Heinrich *et al.* (2000) found that the prevalence of non-asthmatic respiratory symptoms decreased between 1993 and 1995 in all three study areas. In a later study conducted by Heinrich *et al.* (2002), when an additional survey in 1999 was included in the analysis, these results were confirmed. These results too are discussed in 4.5, as well as the question of the extent to which the air pollution situation in the former GDR, where heavy industry was virtually uncontrolled and brown coal was used for heating, can be generalised to the ambient PM situation in the Netherlands.

During the 1996 **Summer Olympic Games in Atlanta**, the influence of temporary changes in transportation behaviours was investigated (Friedman *et al.*, 2001). Ambient air quality and childhood asthma during the 17 days of the Games were compared with those during a baseline period consisting of the four weeks before and the four weeks after the Games. Asthma-related relative risks during the Games were less than unity, and for some medical sources the decrease was significant. These findings suggest that in Atlanta in the summer of 1996 a temporary improvement in ambient air quality contributed to a temporary reduction in the severity of pre-existing asthma. This reduction could not be attributed specifically to any individual pollutant, since PM_{10}, CO, NO_2 and O_3 were reduced in parallel. In the opinion of the authors, reductions in morning rush-hour traffic played an important part in the reduction of asthma-related visits and hospitalisations.

Apart from the reported PM associations, there is also a general tendency in the epidemiological literature to find that gaseous components do not confound the PM associations. In several studies, both PM and gaseous air pollution components were analysed in so-called two-pollutant models. The results of these analyses showed overall that PM associations remained statistically significant when a gaseous pollutant was added to the regression model. Therefore, it is concluded that PM effects cannot be explained by confounding by gaseous components. However, this does not prove that the PM associations are caused by PM. In Europe, PM and gaseous components, apart from O_3, are highly correlated. Sometimes even, very high correlation between the components makes it impossible to analyse two-pollutant models. In the Netherlands we analysed two-pollutant models for PM_{10} and O_3, SO_2, NO_2 and CO respectively, between BS and O_3, SO_2, NO_2 and CO respectively, between SO_4^{--} and O_3, SO_2, NO_2 and CO respectively, and between the sum

of SO_4^{--} and NO_3^- and O_3, SO_2, NO_2 and CO respectively. We concluded that statistically no distinction could be drawn between specific air pollution components on the basis of the statistical analyses. However, Sarnat showed recently that ambient gaseous pollution concentrations correlate well with personal $PM_{2.5}$ concentrations but not with personal gaseous pollutant concentrations (Sarnat *et al.*, 2001), which provides strong evidence that ambient gaseous concentrations act as surrogates for personal $PM_{2.5}$. This might also be the case for other PM fractions like PM_{10}, but no such data were measured in the Sarnat study. However, it is worthwhile interpreting the gaseous associations, found in numerous studies, in the light of the $PM_{2.5}$ findings.

The question concerning the part of the PM that could be responsible for the associations with health effects remains unsolved. While there was a strong tendency to regard the fine particles ($PM_{2.5}$) as having the highest correlation with health effects (compared with PM_{10} or TSP), more recently studies have also been published which show that the coarse fraction (PM_{10}–$PM_{2.5}$) cannot be excluded from health effects. On the other part of the size distribution (PSD), in the ultrafine (UF) domain, associations with health effects were also reported. However, these studies are small in number and the results sometimes inconsistent. The next Section, on the dosimetry and toxicology of PM, explores the extent to which the pure chemical components of, for instance, SIA and sea salt can be deemed causal for PM-associated health effects (cf. subsection 4.3.2.3).

Health effects are associated with PM in time-series studies and these associations are not based on ineffective control for potential confounders. However, the most troubling result from the epidemiological literature is that we do not know yet what part(s) of PM (size, composition or source) cause(s) these effects. Also, we do not know whether there is an independent association with gaseous pollutants as a causal factor for the health effects reported in the scientific literature, though this is probably less likely, based on the work of Sarnat *et al.* (2001). This lack of information generally makes exact scientific predictions of the expected health gains with decreasing concentrations of PM or air pollution virtually impossible. However, based on the enormous number of epidemiological studies, we can expect a reduction in air pollution levels, including PM, to result in a better ambient air quality and a better living environment, and as a consequence better health. The exact gain in health is, however, not quantifiable. A greater understanding of the causal agents is therefore needed

4. Dosimetry and toxicology of PM

4.1 Introduction

Epidemiology consistently indicates that health effects are associated with daily variations in ambient PM levels. Section 4.1 explores the plausibility of specific risk groups being involved and possible mechanisms of action. Risk groups can be identified either by their specific susceptibility or purely by the fact that they receive a higher dose at the target organ compared with the average individual (dosimetry) while breathing PM at a similar concentration. The current toxicological database was reviewed for its support of the causal relationship between mass concentrations of PM and a wide variety of health outcomes. Toxicological mechanisms of action that might explain the observed health effects were also explored and summarised.

Epidemiological associations between PM levels and health outcomes are found throughout the world. This suggests it is unlikely that only one specific source or constituent will be found. Either some common underlying mechanisms of action affected by multiple inducers or a common denominator (causal factors) – present in all areas but not necessarily in PM itself – need to be identified. Toxicologists need to address their wide range of resources and techniques in close collaboration with epidemiologists (mechanistic epidemiology) to solve this complex issue.

Two more or less consistent pictures of air pollution serve together in the epidemiological and toxicological domain. These are the health effects studies performed in Utah Valley and in former Eastern Germany (Frampton *et al.*, 1999; Heinrich *et al.*, 2000). These two places have become synonyms for what could be referred to as two large-scale ‘historical experiments’ that allow a joint epidemiological and toxicological analysis. They currently form a bridge between the fields of epidemiology and toxicology, but the ambient concentrations and the fractions involved are so site- and time-specific that a generalisation of these historical events to the ambient situation in the Netherlands is highly implausible. Therefore, these results cannot explain the prevailing associations between PM and health effects also found in the Netherlands.

In spite of the epidemiologically consistent observations that ambient PM concentrations correlate with human mortality and morbidity, toxicological evidence regarding the exact mode of action of these air pollutants is scarce. PM_{10}, $PM_{2.5}$ and even $PM_{0.1}$ are all ‘containers’ of PM of a certain size range with thousands of individual chemical constituents with specific physical, chemical and biological properties. PM may also contain products of biological origin, such as fragments of pollen, moulds, spores, viruses, and molecules such as endotoxins (LPS), myco- and phytotoxins, and even whole bacteria. Although each ambient air sample is distinct, epidemiological studies have indicated that the toxic potency of PM (defined in terms of relative risk [RR] per unit mass concentration of PM) does not differ more than one order of magnitude between various geographical locations (variations less than a factor of 12 [RR = 1.01 to 1.12 per 50 $\mu g/m^3$ PM_{10}] – see Table 3.10). Concentration differences in various constituents of PM are generally larger than one order of magnitude (Visser *et al.*, 2001).

At least two explanations for this phenomenon can be hypothesised.

- Ambient PM contains a certain amount of a highly potent chemical or group of chemicals which directly cause adverse health effects. If so, this common factor may one day be found in toxicological studies with ambient PM or PM fractions.
- Several fractions or components trigger similar toxicological modes of action resulting in common pulmonary and/or cardiovascular effects. A strong synergy among air pollutants as a minimum is required to explain the associated mortality and morbidity.

The truth is probably more complex, but the two general hypotheses could contribute towards closing the gap between epidemiological observations and toxicological explanations and serve as guidelines for mechanistic research. It is inconceivable that ambient particles from different locations neither share an important toxicological fraction nor share a biological-toxicological mechanism, albeit that traffic emissions in general are omnipresent and come close to being a common factor.

The toxicological literature in the field of ambient PM was recently reviewed (US-EPA, 1996, 2002). A large volume of *in vitro* and *in vivo* animal and human clinical studies using model compounds, surrogate PM as well as real ambient PM were evaluated. These recent reviews are not repeated here, but important findings are used and combined with other available information. The latter includes Dutch studies, some of which have not been reported previously.

With the current low levels of ambient PM it is likely that susceptible individuals, or only these individuals, are affected, or that the toxicity is the combined effect of a mixture of components that is not yet understood. Unfortunately, however, susceptibility data are very scarce in the literature. Also, research on susceptibility can only be carried out when the pathophysiology of the toxicant is, to some extent, known. The latter restriction, however, is at present not met for PM. In recent years much progress has been made, as has been illustrated in the two criteria documents on PM (AQCD) from the US-EPA (1996 and 2002), and future research can be expected to concentrate largely on susceptible individuals.

Section 4.1 describes the role of dosimetry with respect to the relationship between exposure as used in epidemiology and dose as a concept used in toxicology. In addition, the factors possibly related to the causality of PM-associated effects are exemplified and summarised. As hardly any information is available on chronic exposure studies using ambient particulate matter constituents, this topic will not be addressed at all, in spite of the fact that these types of effect should be regarded as important.

4.2. Dosimetry of PM

Section 4.2 addresses issues that affect PM deposition in humans, e.g. the effect of age, breathing pattern, exercise and particle properties, and is based on Winter and Cassee (2002).

In general, epidemiological studies associate ambient concentrations with health effects, mostly assuming a linear relationship with the delivered dose. However, it has been clearly demonstrated that for PM this relationship is not a simple linear function of particle size (Figure 4.1), but that the PM dose depends to a great extent on the particle diameters. This phenomenon is not considered in epidemiological studies, but it has profound implications for risk assessment, which should be based on delivered dose rather than on exposure concentration.

The biological effects of ambient PM_{10} particles may depend on the dose at critical target sites and organs. Estimating or measuring this deposited particle dose is called dosimetry and it forms the link between the external exposure concentration (mostly expressed as mass) and the effective dose at the target site. Health effects of PM_{10} and its constituent fractions in the airways and lungs may depend on the specific dose metric, usually expressed as particle mass or particle number.

Dose assessment and dosimetry are essential in PM risk assessment in relation to particle size. It is therefore important to know:

- the deposited dose of the particles
- which of the specific PM fractions (size) result in the highest deposited dose
- which particle dose metrics have the closest link with health effects, and
- whether certain human subpopulations may be at greater risk, due in part to an enhanced PM deposition and larger dose.

Unlike PM_{10}, the choice of $PM_{2.5}$ as the basis for a standard was not primarily fostered by dosimetric considerations, as might have been expected from the point of view of protecting public heath. Figure 4.1 shows that three size fractions can be distinguished, namely a coarse, a fine and an ultrafine mode. However, debate about the exact cut-off diameter between fine and coarse PM continues.

In the 16th century Philippus Aureolus Theophrastus Bombastus von Hohenheim better known as Paracelsus, and pictured on the IRAS letterhead, coined the phrase that “All substances are poisons. There is none which is not a poison. The right dose differentiates a poison and a remedy.” This prhase has never been more discussed than in the field of ambient PM health effect research. This historical quote indicates on the basis of classical toxicology that there must be something of a dose-effect or concentration-response curve for PM. Since epidemiological studies do not provide a threshold level and acute mortality effects have been identified at present ambient concentration levels (which are generally lower than the standards), the toxic potency of the causative fraction must be extremely high. Irrespective of chemical composition, the EU toxicity criteria for the classification of toxic substances call a 4-hr lethal concentration (LC50) of less than 250 000 $\mu g/m^3$ obtained in rats ‘very toxic’, the highest toxicity class. Extrapolating this ‘very toxic’ value to a 24-hr exposure and adjusting for inter- and intra-species differences using the factors in normal use for toxicological standard setting of 6 x 10 x 10, a human equivalent would be in the order of 400 $\mu g/m^3$ for a 24-hr exposure. This estimate may seem an oversimplification, but it does illustrate the fact that PM_{10} appears to possess an extremely high toxic potency.

Current epidemiological studies indicate that for the Dutch situation approximately five people die per day from effects associated with exposure to ambient PM at an annual average level of 30–40 µg/m^3, producing an annual figure of nearly 2000 excess deaths (cf. Table 3.7). From this perspective, ambient PM should indeed be regarded as a 'very toxic' aerosol. This can be elucidated by a simple calculation. Assuming no local but only systemic effects from PM, an extreme dose of 140 µg/m^3 (the current daily standard in the Netherlands) and 100% deposition efficiency (which is also extreme), an absolute maximum estimate for an average person weighing 70 kg with a breathing minute volume of 15 l/min. predicts a deposited daily dose of 3000 µg of PM. The more usual average ambient levels of PM would result in an estimated maximum dose of about 200 µg per person. Only a few chemicals are known to have such high acute toxicity. It is therefore difficult to understand how subjects with cardiorespiratory diseases could die from current short-term exposures to ambient air PM. The mechanisms and chemical and physical fractions for ambient PM effects are unclear.

A comparison with historical data does not really help us with the current enigma of PM either. During the peak of the historic London fog episode from 7–8 December 1952, not only were the concentrations of PM extremely high compared with current standards, so too were those of SO_2 (and probably other non-measured gases as well) (Wilkins, 1954). Approximately 900 people died in London during this period, compared with a regular mortality of 295 in the days before the event. The PM concentrations measured at that time were 1600 µg/m^3. If all the mortality effects of these two days of the 1952 episode had been ascribed solely to PM, then translating the 1952 figures into figures that are comparable with the current results of epidemiology it would have resulted in a RR of 1.06 per 50 µg/m^3 of PM. Compared with the RR values in Table 3.10, this value for acute mortality from PM in 1952 does not seem extreme. The snag is that apart from the fallacy of ascribing all health effects solely to PM, general principles of toxicology do not allow us to extrapolate from these high concentrations to the current low concentrations, assuming a similar biological effect right down to near zero concentrations. However, from a precautionary principle an extrapolation of this kind should be – and in practice currently is – used by standard-setting agencies to put the PM enigma on the research agenda.

The larger PM fraction with diameters between 1.0 and 10 µm is captured quite efficiently in the upper airways. However, a significant portion of this fraction will still be deposited in the lower airways and lungs (depending on breathing parameters and particle characteristics). The finer particle fraction (0.1 and 1.0 µm), with a low overall deposition rate, is almost exclusively deposited in the lower airways and gas exchange regions. These data suggest that, from a deposition point of view, both more coarse (> 1 µm) and UF (< 0.1 µm) particles may play a role in lower airway effects and that larger particles may also contribute to upper airway effects (cf. Figure 4.1).

The amount of particles deposited in various parts of the airway and lung sections depends on particle size. The upper (extrathoracic) respiratory tract (URT) (mouth, nose and larynx), the tracheobronchial airways (TB), and the alveolar region or pulmonary region (P) all receive different doses, depending on the particle size. Inhalability, oronasal breathing, and tracheobronchial path length and acinar volume

influence the deposition of particles between experimental animals and humans to a different extent (Miller, 2000). Still, major similarities as well as extensive morphological data allow the extrapolation of data for risk assessment purposes. However, care must be taken when investigating particles above approximately 5 µm in animal studies, since the inhalability (the probability of entering the airways through nasal breathing) in rats decreases from 65% to 0% as aerodynamic particle size increases from 5 to 10 µm. In addition, rodents are obligate nasal breathers, but humans switch to oronasal breathing when work or exercise requires a minute ventilation that exceeds about 35 l/min. Also, a significant portion of ultrafine particles will be removed from the air stream in the nose by diffusion. As a result, oral breathing will receive a significantly higher dose of PM in the lower respiratory tract. These differences have significant implications for particulate risk assessments and lead to specific human-rat extrapolation coefficients for PM.

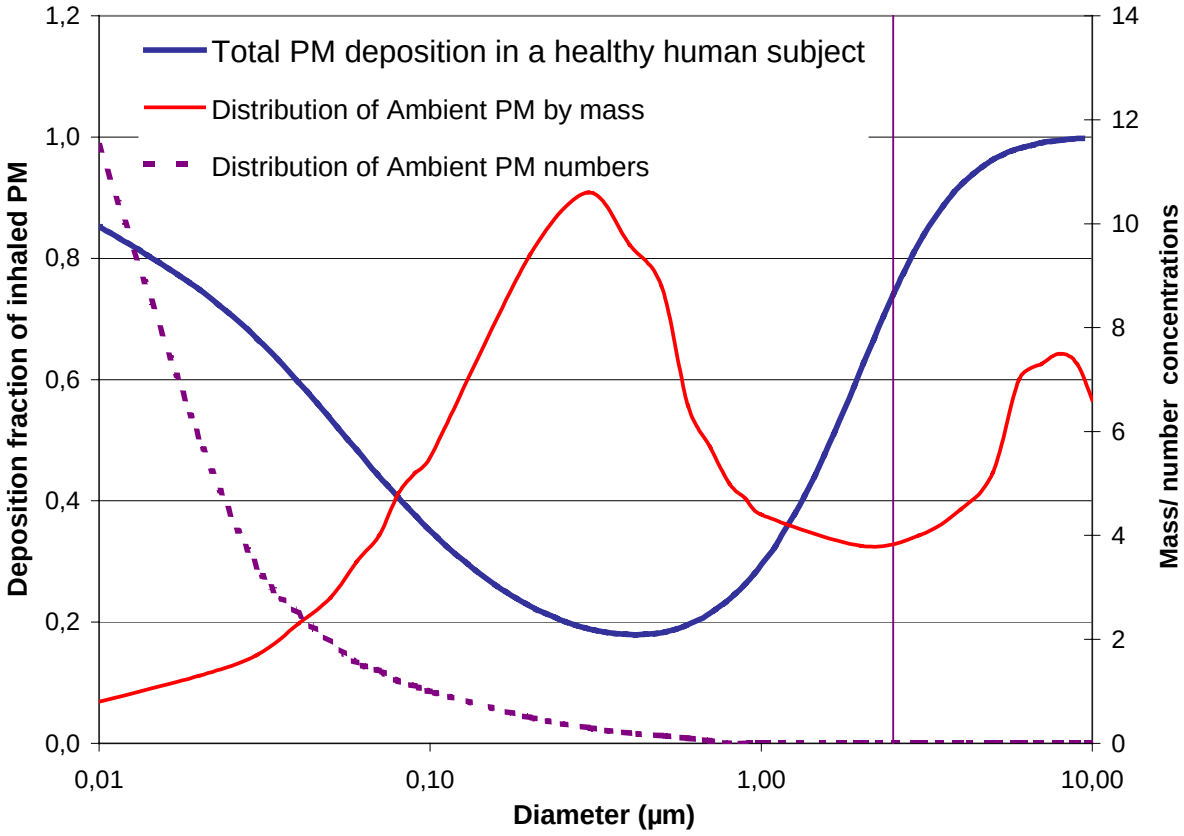


Figure 4.1 Deposition of particles in the human respiratory tract as well as an average ambient particle mass and number distribution as function of the diameter.

Table 4.1 Calculations of total deposited dose of ultrafine, fine and coarse PM fractions in human airways and lungs using the ICRP model and expressed relative to the dose in a healthy adult male at light exercise and standardised to respiratory tract tissue mass (Freijer et al.,1997).

Subjects*	Ambient PM fractions Ultrafine < 0.1 µm	 Fine 0.1–2.5 µm	 Coarse 2.5–10 µm
Adult male	100%	100%	100%
Children, < 10 years old	120–170%	110–250%	140–170%
Adult male			
- at rest or sleep	30–40%	20–40%	30–40%
- at heavy exercise	190–200%	140–170%	200%
Adult male with COPD	400–440%	370–500%	420–430 %

* at light exercise, unless stated otherwise

Existing particle deposition models use the airway and lung geometry data of a healthy, normal adult. In the lower airways and lungs of COPD patients, however, the deposition of particles is greatly enhanced, partly because of an altered, often oral, breathing pattern. COPD patients have about three quarters of the inhaled air going over only about one quarter of their lung surface. Thus, altered ventilation patterns and tissue pathophysiology can be expected to result in increased local PM doses in some parts of the lungs of COPD and asthma patients, as presented in Table 4.1. Remarkably, besides the fine and ultrafine fractions, the coarse fraction of PM is also deposited to a substantial extent in both lower and upper airways. This indicates that from the perspective of deposition and dose, the coarse fractions might, at least from the dosimetry viewpoint, play a role in causing adverse health effects.

The Chemical Substances Threshold Limit Values Committee of the American Conference of Governmental Industrial Hygienists (ACGIH, 1996) and the Organisation for Standardisation/European Standardisation Committee (ISO/CEN) use a median cut-off point of 10 µm aerodynamic diameter for thoracic (lung airways plus gas-exchange region) and 4 µm for respirable (gas-exchange region) particulate mass. If the potential health effects are predominantly caused by fine PM, as has been hypothesised, biologically speaking one would expect a PM_4 rather than a $PM_{2.5}$ standard. With a PM_4 standard, greater account would be taken of the potential hazard of suspensions of solid particles and droplets in this region of the lungs than with a 50% cut-off point of 2.5 µm.

The CIIT/RIVM particle deposition model allows calculation of PM deposition fractions and exposure doses for humans and rats, and includes age-specific human lung models. Human regional depositions for different particle sizes were calculated for a default set of physiological and breathing parameters as well as lobar depositions and alveolar deposition distribution for a stochastic human lung. Large individual variations exist in regional deposition due to the differences in fractional residual capacity (FRC) and breathing parameters, and in lobar and alveolar deposition due to differences in lung geometry.

For coarse particles (5 µm and 10 µm) tracheobronchial and thoracic deposition fractions are significantly larger for children (aged from 0 to 15 years old) than for adults, mainly due to the increase in head deposition from children to adults. The difference in tracheobronchial and thoracic deposition fractions between children and adults increases with particle size. Pulmonary or alveolar deposition fractions of 5 µm particles for 8- to 14-year-old children are higher than for adults. For coarse particles deposited aerosol mass rates and exposure doses in the thoracic region of 8- to 14-year-olds for 5 µm particles and of 2- to 14-year-olds for 10 µm particles are higher than in adults (18 and 21 years old). Deposited aerosol mass rates and exposure doses in the tracheobronchial region per unit surface area decrease with progressing age. However, individual differences in the tracheobronchial surface area and in the tidal volume can disturb this dependency, leading to even equal exposure doses per unit surface area for infants and for adults. Pulmonary deposition fractions per alveolus are higher for babies compared with adults, for children aged about 2 years and older pulmonary deposition fractions per alveolus do not change significantly. Deposited pulmonary mass rates and exposure doses per alveolus are lower for children aged 2–3 years compared with adults for ultrafine and 2.5 µm particles, and higher for 8- to 14-year-old children compared with adults for 5 µm particles. The pulmonary exposure doses per unit surface area increase almost linearly by approximately 30% between the ages of 20 and 80 years.

Aerosol deposition in human adults was calculated at different levels of physical exertion. Tracheobronchial deposition for the ultrafine and coarse particles fractions decreases slightly from rest to light exercise. Tracheobronchial deposition for coarse particles rises when breathing is changed from nasal to oronasal and increases from modest to heavy exercise. Pulmonary deposition for the ultrafine particles fraction increases from rest to light exercise; deposition for the coarse particles fraction decreases from rest to light exercise, rises when breathing is changed from nasal to oronasal and decreases slightly from modest to heavy exercise. Deposited thoracic mass rate increases with increasing physical exertion, faster for heavy exercise.

Variations in fractional residual capacity (FRC) mostly affect the pulmonary deposition fractions. An increase in FRC of about 1.5 l results in a minimum 25% decrease in pulmonary deposition fractions. For oral breathing, tracheobronchial and pulmonary deposition of coarse particle fractions are larger (3 times for 5 µm particles) than for nasal breathing. A longer than normal expiratory fraction leads to a slightly lower tracheobronchial deposition of ultrafine and coarse particles and to a slightly lower pulmonary deposition of coarse particles. However, the pulmonary deposition of ultrafine particles is slightly larger. Tracheobronchial deposition is slightly larger for ultrafine and coarse particles when there is a pause between inhalation and exhalation. Pulmonary deposition is larger for all particle sizes when there is a pause between inhalation and exhalation.

Tracheobronchial and thoracic deposition fractions are larger for humans than for rats. Pulmonary deposition fractions are larger for humans than for rats for fine (larger than 0.3 µm) and coarse particles. Deposited thoracic mass rates are 45–200 times larger for humans than for rats at rest. Coefficients for the rat-human extrapolation of exposure dose and standardised exposure dose were determined for three levels of human physical exertion (sleep, awake rest and light exercise), which allows equal

exposure levels to be achieved for rats and humans by changing the aerosol concentration or exposure time.

4.3 Toxicological evidence for causality

The toxicological information has been extensively reviewed elsewhere (Van Bree and Cassee, 2000; US-EPA, 1996, 2002). In their critical review Van Bree and Cassee (2000) focused on ambient particulate air pollution (PM) toxicity and particle hypotheses. Mechanisms were evaluated to investigate the causality and plausibility of acute health effects associated with ambient exposure. High-dose studies indicated that PM induces oxidative pulmonary inflammation and cardiorespiratory malfunctioning, which could contribute to a disease exacerbation mechanism. PM surface reactivity appears to be more important than PM mass, thus prudently suggesting an important role for the anthropogenic (carbonaceous) fine fraction. The limited number of low-dose PM inhalation studies supported this suggestion. Coarse PM may still be important in health effects related to upper airways (e.g. worsening of asthma). However, a role for secondary components (sulphates, nitrates) or ultrafine PM at levels occurring in ambient air have not yet been established. Evidence that exhaust particles, particularly diesel, played a role in acute PM health effects is still marginal. Studies have indicated that mixtures of particles and gases like ozone may result in more toxicity than the individual components. Current dosimetry models predicted that people with cardiorespiratory diseases, especially older people, may receive increased PM doses upon exposure. Ambient PM toxicity studies have been intensified in recent years, but the current limited data have not yet produced sufficient evidence to demonstrate convincingly: 1) a specifically important and causal role for one form of PM fraction or composition and 2) mechanisms explaining PM health effects in people considered to be at increased risk, as Van Bree and Cassee reported in 2000.

Section 4.3 summarises some more recent information and highlights possible factors based on physical, chemical or biological properties of PM that can be linked to epidemiological observations. The focus is on short-term exposure studies and acute effects. In addition, although ultrafine PM is receiving increased attention, this fraction is not extensively reviewed due to the lack of correlation with the mass of PM.

4.3.1. Ambient particles and disease models

Studies using concentrated ambient $PM_{0.15\text{-}2.5}$ (CAPs) have been conducted on experimental animals since 1998. Initially, experiments focused on 3-day inhalation exposures in healthy and compromised rats and mice, thereby mimicking possible human risk groups. About half of the studies were carried out in an industrialised area of the city of Utrecht, while the other half were conducted on the RIVM premises in Bilthoven (Tables 4.2 and 4.3). High variability in ambient PM and in exposure concentrations resulted in the decision to focus on 1-day exposure only (Table 4.4). In late 2001, a third contrasting location subject to high motorway emissions was selected. In addition, other disease models in rats were applied, focusing primarily on lung inflammation and blood hypertension. The following paragraphs summarise

these CAPs studies. A detailed analysis of the RIVM CAPs exposure studies is expected to be published at the end of 2002.

Respiratory allergy

People with respiratory allergy (asthma) are believed to be more susceptible to air pollution in general and PM in particular. Several animal models were applied in which asthmatic symptoms such as increased IgE levels and increased airway reactivity upon exposure to a non-specific challenge were induced. For example, ovalbumin-treated mice (Hessel *et al.*, 1995) were exposed in two independent experiments to ambient PM (CAPs) concentrated by a factor of about twenty for four hours a day and for three consecutive days. The smooth muscle contraction of isolated trachea caused by increasing concentrations of metacholine exposure was measured one and four days after CAPs exposure. The CAPs mass concentration varied significantly between each experiment (mean mass concentration of 1,161 and 556 $\mu g/m^3$) and also within the three exposure days (350 $\mu g/m^3$ up to 2 mg/m^3). Sulphate levels were roughly 15–20% and nitrate levels between 40 and 50% of the total mass. Histopathological examination showed that asthmatic mice had an inflammation in lung tissue (as shown by an increased number of eosinophils) as well as a hypertrophy of the bronchiolar epithelium. However, CAPs did not seem to affect the pathological conditions of the bronchoalveolar regions. The ovalbumin treatment resulted in a low hyperreactivity of the trachea by metacholine compared with healthy controls. This difference did not become statistically significant and was lower than observed in previous studies. No biologically significant effect of CAPs on tracheal reactivity could be observed. We concluded that CAPs in this study did not affect airway pathobiology. Neither a significant hyperreactive nor a hypersensitive response was observed despite the relatively high exposure levels of CAPs. These data did not provide support for the hypothesis that CAPs can exacerbate a pre-existing asthmatic condition in a relevant experimental animal model, but neither did they refute the hypothesis. The latter was concluded as CAPs themselves appear to have been ineffective and, in addition, the asthmatic effects are present though not very observable.

In addition to a mouse model, another asthma model was developed in rats. Brown Norway rats were sensitised with house dust mite antigen (HDM) as a model for atopic asthma (Gilmour *et al.*, 1996; Lambert *et al.*, 2001) and subsequently exposed to CAPs for six hours. Specific lung function parameters like pulmonary resistance (RL), dynamic compliance (Cdyn) and airway responsiveness were measured after exposures to CAPs. Only in one of two independent experiments were marginal ($0.05 < p < 0.02$) effects on RL and no effects on Cdyn observed after CAPs exposures ranging from 80–400 $\mu g/m^3$. The effects, however, were rather mild and most probably not clinically relevant. Notably, while mass concentrations of CAPs in two studies were comparable, these effects could not be related to the mass concentrations of CAPs. These studies were also among the first at the RIVM to show moderate impacts on health-effect indicators found in blood rather than in the lung. Increased blood fibrinogen, reduced numbers of red blood cells and haemoglobin levels were observed in rats exposed to two different days of CAPs, whereas a clear pulmonary response was absent.

It was concluded from these studies that CAPs could result in minor systemic effects and minor effects in compromised airways of an animal model mimicking a human PM risk group. Similar effects were observed in healthy animals, suggesting that the allergy models are either not sensitive enough or simply do not respond more pronouncedly than healthy animals. This means that this study design could not provide evidence for increased sensitivity of asthmatics (as a possible risk group) to PM, although PM itself seems to provoke toxicity.

Pulmonary *hypertension*

Two identical studies were carried out in which rats were pre-treated with monocrotaline to induce pulmonary inflammation, followed by muscularisation of the vascular system of the lung leading to pulmonary hypertension (Gordon *et al.*, 1998; Kodavanti *et al.*, 1999; Schultze and Roth, 1998; Watkinson *et al.*, 1998). This type of hypertension can result in right ventricular hyperplasia and hence heart failure. The animals were exposed after the pulmonary inflammation had disappeared.

The adverse effects were studied in healthy and compromised rats four and eleven days after short-term (4 hr/day for 3 consecutive days) exposure. Exposure levels were on average 350 and 1100 µg/m3 respectively, ranging between 200 and 1700 µg/m^3. This range illustrates the significant differences between exposure days. Health effects were studied using bronchiolar lavage fluid analysis and histopathological examinations. However, no statistically significant differences were observed between sham and CAPs exposure groups regardless of the time following exposure or the treatment with monocrotaline.

Occasionally, small changes in heart rate and peripheral blood cell differential counts were observed in both normal and monocrotaline-treated rats within a few hours of CAPs exposure (Gordon *et al.*, 1998; Gordon *et al.*, 2000a,b). However, these effects were not noted twenty-four hours after, suggesting an acute but possibly systemic effect. This rapid recuperation might be the explanation for the absence of CAPs effects in our own studies with a longer interval between exposure and health effects measurements, as well as the fact that Gordon *et al.* (1998, 2000a,b) observed the effect in the blood rather than in the lungs.

This hypertension model was also used in a study involving concentrated freshly generated diesel exhaust particles (DEP) mixed with ambient air (CDP) (Cassee *et al.*, 2002). It was hypothesised that a single six-hour exposure to PM exacerbated inflammatory processes, affecting health parameters in the blood. Histopathology of lung and nose, bronchiolar lavage (BAL) and blood analyses were performed one, two and four days after the CDP exposure. Morphometry of BrdU-labelled cells in lung and nose was performed four days after exposure. MCT-treated rats showed hypertrophy of the media of the pulmonary muscular arteries that was not effected by CDP. BrdU-labelling of predominantly Clara cells in the terminal bronchioles was markedly enhanced after an additional exposure to CDP. However, no increases in Clara cell protein (CC16) levels were measured in either BAL or blood. CDP exposures did not induce significant irreversible toxic effects in the lungs. In addition, blood fibrinogen levels were enhanced in pulmonary hypertensive rats exposed to CDP. The study demonstrates that very high CDP concentrations are needed to result

in pulmonary changes in animal models, with pulmonary hypertension that continues for days after a single exposure. Also, CDP has the potential to induce changes in blood. It must be stressed, however, that these concentrations were rather high. Dosimetric comparison indicates that rats need only be exposed to 2–10 times higher levels than humans to arrive at the same internal dose in the lungs. This would mean that the applied level of 8–9 mg/m^3 is consistent with at least 1000 µg/m^3 for humans.

Pulmonary inflammation

A hypothesis for the mechanisms of PM-induced health effects departs from the idea that an already existing pulmonary inflammation is required to result in PM effects. In a series of experiments, ozone exposures were applied to induce inflammation. In this case, the use of ozone should not be confused with ambient ozone exposure or combined ozone-PM exposure. The type of inflammation can be related to a type of pneumonia. The inflammatory time-response curve was shown to reach an optimum at twenty-four hours post-exposure. Therefore, a single 8-hour exposure to 1600 µg/m^3 of ozone (twenty-four hours before CAPs exposure) was applied in this model.

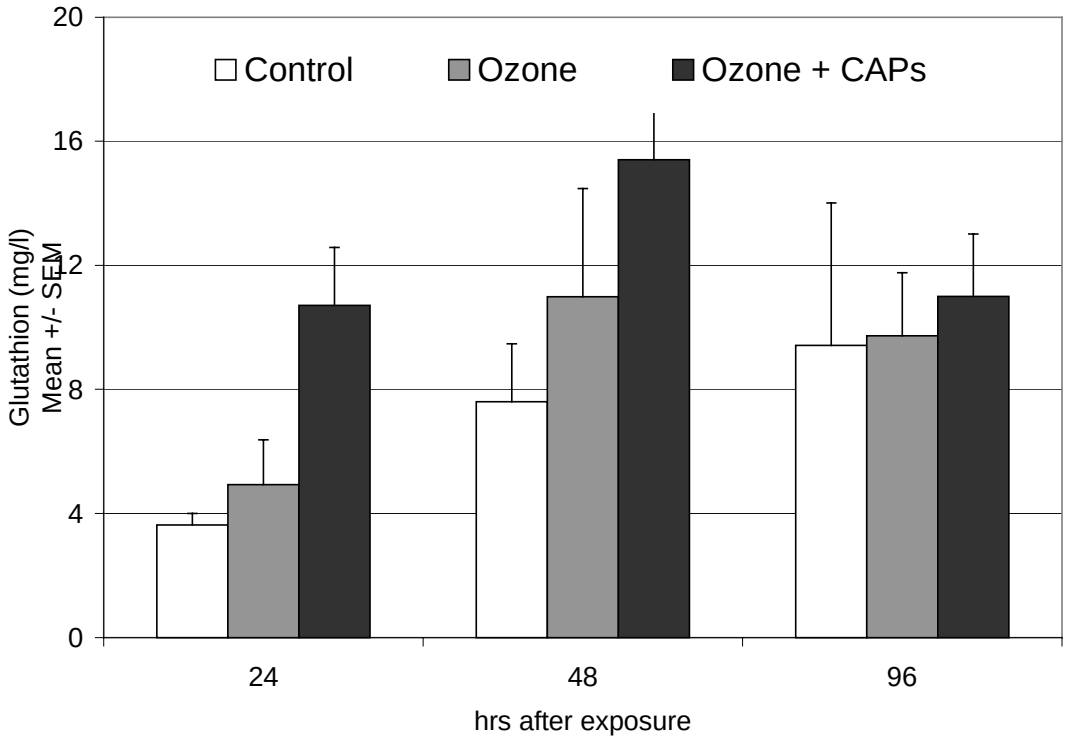


Figure 4.2 Effects on anti-oxidant glutathione at different intervals after exposure to CAPs in rats pre-exposed to 1600 µg/m^3 ozone.

[Because the previous studies have shown limited effects due to CAPs exposures, and the reviewers have expressed the need for a positve control]. In 2000, diesel exhaust particles were used as a potentially positive control. This positive control is intended to help us validate and test the usefulness of the effect parameters for determining effects caused by PM. To examine the effects of particle air pollution in animals, rats with a pre-existing pulmonary inflammation (induced by 1600 µg/m^3 ozone) were

nose-only exposed to concentrated freshly generated diesel exhaust particles (DEP) mixed with ambient air (CDP). It was hypothesised that a single 6-hour exposure to PM would exacerbate respiratory inflammatory processes, which affect health parameters in the blood. Histopathology of lung and nose, bronchiolar lavage (BAL) and blood analyses were performed one, two and four days after CDP exposure. Morphometry of BrdU-labelled cells in lung and nose was performed four days post-exposure. CDP exposure did not influence this pattern, with the exception of alveolar macrophages that were loaded with CDP. The proliferation of Clara cells in the terminal bronchioles significantly increased after exposure to CDP, but no changes in Clara cell protein were observed in either BAL or blood. This cell proliferation might very well be correlated with the oxidative stress. CDP exposures clearly induced an oxidative stress, indicated by increasing glutathione levels in BAL with time. Although some slight changes in blood parameters were seen, these effects were not consistent. How the effects found with CDP would compare with similar levels of ambient particles has not yet been established.

Similar results were observed in a similar study applying the ozone inflammation model, using only ambient air without added diesel exhaust (Figure 4.2). Increased levels of antioxidant shortly after CAPs exposure and enhanced levels of markers for toxicity at 1000–2000 $\mu g/m^3$ indicate that CAPs can potentially cause health effects. So far, these are the most prominent health effects observed in the RIVM PM-toxicity studies. More recent studies using this animal model have indicated that oxidative stress markers such as glutathione can be affected by CAPs, but there is no clear relationship with the mass concentrations. These studies are currently being evaluated and results will soon be presented in a separate report.

Systemic hypertension

Lately, toxicology appears to observe systemic effects due to PM exposure even without a marked effect in the pulmonary region. Recent studies have focused on the effects of CAPs spontaneously developing systemic hypertension in SHR rats. Initially, same-age animals (7–8 weeks old) were used, but this was then followed by exposures in older animals (12–13 weeks old) after consultation with our US-EPA colleagues. We also moved to a location dominated by traffic exhaust. We captured PM right next to the entrance to an aqueduct crossing a busy highway near Rotterdam, the Netherlands. Because of the length of time it takes to collect all the analytical data from a CAPs study, no detailed results can yet be provided on these latter studies. However, the overall picture is that the SHR rats are less sensitive to CAPs exposures as regards pathological or biochemical changes in the lung when compared with the ozone-induced inflammation model: cell proliferation, cell viability, cell differentials in BALF or blood, biochemical markers for toxicity in BALF and histopathology were generally not affected by CAPs. Some parameters were highly variable and have not yet been correlated with the exposure characteristics. Also, the cell proliferation rate, a very sensitive marker in ozone-exposed animals, did not increase.

Tables 4.2 to 4.4 summarise the exposure characteristics of the RIVM CAPs studies and show that rather high exposure levels can be achieved, not only on a total mass basis but also with respect to sulphate and nitrate. In addition, a wide range of

exposure conditions will allow a search for an existing concentration/dose-effect relationship.

Table 4.2 Exposure concentrations of RIVM CAPs studies 1998. Animals were exposed for four hours a day for 3–4 consecutive days.

Model	Species	Exposure Duration	CAPs µg/m^3	Sulphate µg/m^3	Nitrate µg/m^3	Chloride µg/m^3	Ammonia µg/m^3
		6	731	222	60	-	-
Healthy	Wistar rat	6	227	45	30	-	-
Hypertension	SHR rat	6	188	22	45	-	-
		6	396	115	52	-	-
Healthy	Wistar rat	6	366	87	73	-	-
Hypertension	SHR rat	6	401	68	93	-	-
Asthma	BN rat	4	415	49	22	-	-
(Timothee		4	86	9	3	-	-
pollen)		4	96	9	3	-	-
		4	-	12	4	-	-
Asthma	BN rat	4	45	3	2	-	-
(Timothee		4	77	43	15	-	-
pollen)		4	204	24	8	-	-
		4	393	31	28	-	-
Asthma	BN rat	4	812	226	202	-	-
(Timothee		4	514	99	64	-	-
pollen)		4	1225	259	293	-	-
Asthma	BN rat	4	1016	140	321	-	-
(Timothee		4	863	-	-	-	-
pollen)		4	528	120	117	-	-
Asthma	Balb/C mouse	4	418	121	68	-	-
(ovalbumin)		4	691	187	173	-	-
		4	388	72	65	-	-
Asthma	Balb/C mouse	4	457	52	79	-	-
(ovalbumin)		4	286	93	21	-	-
		4	165	15	18	-	-
Asthma	Balb/C mouse	4	101	14	10	-	-
(ovalbumin)		4	187	3	3	-	-
		4	464	107	65	-	-
Healthy	BN rat	4	877	266	108	2	123
		4	400	211	122	5	107
		4	852	264	203	9	153

Table 4.3 Exposure concentrations of RIVM CAPs studies 1999. All animals were exposed for four hours a day for three consecutive days.

Model	Species		CAPs µg/m^3	Sulphate µg/m^3	Nitrate µg/m^3	Chloride µg/m^3	Ammonia µg/m^3	MMD µm
Allergy model (ovalbumin)	Mouse (Balb/C)		530	85	209	15	87	0.87
			1007	185	363	13	166	0.84
			1947	276	769	24	318	0.73
		Mean	**1161**	**182**	**447**	**17**	**190**	**0.81**
		Sd	721	95	289	6	117	0.07
Allergy model (ovalbumin)	Mouse (Balb/C)		-	-	-	-	-	0.85
			407	76	62	16	31	0.82
			734	86	250	12	89	0.75
		Mean	**571**	**81**	**156**	**14**	**60**	**0.81**
		Sd	231	7	133	3	41	0.05
Pulmonary hypertension (monocrotaline)	Rat (Wistar)		634	36	24	46	6	0.94
			297	41	28	14	10	1.02
			29	91	157	18	55	0.66
		Mean	**320**	**56**	**70**	**26**	**24**	**0.87**
		Sd	304	31	76	17	27	0.19
Pulmonary hypertension (monocrotaline)	Rat (Wistar)		2413	320	676	23	283	0.58
			817	121	43	23	35	0.87
			543	149	67	25	52	0.83
		Mean	**1258**	**196**	**262**	**24**	**123**	**0.76**
		Sd	1010	108	359	2	138	0.16

Table 4.4 Exposure concentrations of RIVM CAPs studies 1999–2000. All animals were exposed for a single 6-hour period.

Model	Species	CAPs µg/m^3	Sulphate µg/m^3	Nitrate µg/m^3	Chloride µg/m^3	Ammo-nia µg/m^3	MMD µm
Allergy model	Rat	1464	328	328	12	180	0.97
(house dust mite)	(Brown	752	57	105	18	27	0.72
	Norway)	1874	298	120	24	118	0.66
Healthy	Rat	680					0.78
	(Wistar)	1261					0.96
		3049					0.81
		1015					0.46
		883					0.84
		414					1.40
		278					0.52
		201					0.99
		328					0.79
		206					0.82
Lung inflammation	Rat	7833[1)]					0.32
(ozone-induced)	(Wistar)	8943					0.19
		9313					0.22
		8563					0.23
		8784					0.22
		7346					0.30
Pulm hypertension	Rat	2129[1]					0.30
(monocrotaline)	(Wistar)	2199					0.18
		9680					0.16
Lung inflammation	Rat	978	137	183	4	100	0.70
(ozone-induced)	(Wistar)	1755	326	471	4	254	1.30
PART I		1040	170	241	3	132	0.64
		1392	256	424	8	217	0.64
		967	192	255	7	145	0.64
Lung inflammation	Rat	1104	222	221	8	147	0.80
(ozone-induced)	(Wistar)	1075	305	161	13	157	1.53
PART II		671	143	109	3	81	1.62
		1348	267	109	20	192	1.51
		1204	275	176	33	150	0.80
		740	133	169	14	98	0.59
Lung inflammation	Rat	1228					0.72
(ozone-induced)	(Wistar)	1842					0.59
PART III		414					0.77
		2085					0.52
		393					0.94
Hypertension	Rat	377					0.71
	(SHR 7-8)	271					0.91
		3721					0.87
		1161					0.65
		335					0.71
		511					0.59
Hypertension	Rat	2400					0.73
	(SHR 12-13)	556					1.14
		471					0.56
		690					0.48
		1931					0.60

1) Exposures were dominated by diesel exhaust particles: the inlet of the concentrator was located near the exhaust of a power supply driven by a diesel motor.

Other investigators using similar concentrator technologies in a wide variety of experimental designs (Godleski *et al.*, 1996, 2000; Gordon *et al.*, 1998, 2000a,b; Zelikoff *et al.*, 1998; Clarke *et al.*, 1999; Wellenius *et al.*, 2002; Brook *et al.*, 2002; Harder *et al.*, 2001; Kodavanti et al., 2000a,b; Batalha *et al.*, 2002) have indicated that

statistically significant changes in health parameters can sometimes be observed. However, the reproducibility as well as the clinical or biological impact is under discussion. Owing to the large number of variables in studies of this type (no control over composition of PM) at present, the small number of experiments do not allow sound conclusions on the role of PM, although plausibility for PM-associated health effects is provided. Interestingly, a common feature in all of these studies seems to be an effect on heart function, which could not be related to the PM mass concentration or the duration of the PM exposures. Godleski and colleagues suggested that people with ischaemic heart disease are predisposed to developing life-threatening cardiac effects. Very recently, the same group showed vasoconstriction following short-term exposure (3 days, 5 hr/day; 70–750 μg/m3) in a healthy rat and a rat with chemically induced bronchitis. These results and results from studies in which animals were exposed by intratreacheal instillation support the hypothesis that PM affects the vascular system by, for instance, inducing endothelial damage. These effects could also explain the changes in heart rate and heart rate variability.

Human clinical evidence concerning PM and resulting dose-effect relationships is scarce. In fact, as yet, such studies have only been performed in the USA (US-EPA, USC) and Canada (Health Canada). The results of these three research groups also provide some evidence for a cardiovascular mechanism indicated by changes in heart function parameters (heart rate variability, high and low frequency) or blood coagulation (fibrinogen). For instance, concentrated $PM_{2.5}$ levels up to 300 $\mu g/m^3$ resulted in mild inflammation in the lower respiratory tract and an increased concentration of blood fibrinogen (Ghio *et al.*, 2000) but had no effect on immune phenotype or macrophage function under the conditions tested (Harder *et al.*, 2001). In a following study in which healthy volunteers received a 500 μg dose of PM extracts from Utah Valley PM filters by instillation in the lungs, Ghio *et al.* (2000) (Ghio and Devlin, 2001) concluded that mass may not be the most appropriate metric to use in assessing health effects following PM exposure but rather that specific components should be identified and assessed. Two-hour inhalation exposure of healthy volunteers to up to 150 $\mu g/m^3$ combined with 240 $\mu g/m^3$ ozone in Toronto was linked to acute conduit artery vasoconstriction (Brook *et al.*, 2002). This at least provided a plausible explanation for epidemiological findings, because these alterations may be a major pathway of PM-provoked acute cardiac events. The mixture is also relevant to ambient conditions. It seems that the evidence for PM causing systemic effects (or at least being able to induce effects that go beyond the lung as target organ) is growing.

Although a systematic review of the CAPs studies has not yet been performed, signs of PM being able to cause adverse effects in both healthy and compromised animals and humans are emerging. From the evidence available from studies using CAPs exposure in either animals or human subjects, it appears that PM mass is not evidently associated with adverse health effects. This suggests that other metrics might be more appropriate, e.g. chemical composition or physical properties. This is discussed in 4.3.2.

Data from field studies with contrasting traffic density and PM levels in São Paulo and Florence (experimental animals) and Mexico City (humans) have shown that

serious histopathological changes in upper and lower airways can occur. These effects may be linked with effects on the immune system and respiratory morbidity and mortality as found at these locations in epidemiological studies (Calderon-Garcidueñas *et al.*, 1999, 2000; Lemos *et al.*, 1994; Saldiva *et al.*, 1992, 1994; Gulisano *et al.*, 1997). The relative contribution of traffic-related PM exposure compared with gaseous pollutants is not clear but is expected to be substantial. Swedish research recently revealed that exposure of human healthy and asthmatic volunteers for up to two hours to diesel exhaust in controlled chamber studies induced airway inflammation (Nordenhäll, 2002). The asthmatics appeared to be no more sensitive to diesel exhaust, with the exception of some inflammatory markers (IL-10). Exposure also caused lung function change, but it was similar in magnitude in both healthy and compromised subjects. The investigators also pointed out that some, but not all, of the health outcomes were in accordance with the lag effects observed in most time-series studies. In these studies 100 $\mu g/m^3$ was usually used (at a 50% efficiency at the cut-off of 10 µm), and the results could not be evaluated for the effects of either the gases or particulates alone. Overall, these studies showed that traffic-related air pollution and the levels that can be achieved in a busy street could induce health effects, in particular in the case of longer lasting exposures.

Apart from these real-time ambient PM exposures, a number of studies were conducted using either PM that is re-suspended in air or that is administered by using intratracheal instillation techniques. Although intratracheal instillation studies have a drawback in the sense that PM is delivered as a bolus in the lung with a supposedly uniform distribution regardless of the size of the particles, they can be used for comparison of the toxicological potential of collected PM samples or for mechanistic research. For example, Ulrich *et al.* (2002) used a rat model in which the animals were pre-treated with ozone (1600 $\mu g/m^3$) to induce a mild inflammation, followed by instillation with 500 µg, 1500 µg or 5000 µg ofparticulate matter (PM) from Ottawa, Canada (ECH- 93) per rat. They subsequently determined changes in health indicators at two, four and seven days after PM exposure. A crucial inflammatory mediator, TNF-α, was elevated approximately four times two days after instillation in lung lavage fluid with the highest ECH-93 concentration, whereas protein levels of another cytokine, MIP-2, were approximately three times higher during the entire time period studied. As increased plasma endothelin-1 (associated with elevated blood pressure), decreased ACE activity, as well as decreased mRNA levels (endothelial dysfunction) were also observed, it was suggested that endothelial damage probably caused by NO-induced toxicity occurred in the animals. Together with the 20%-elevated fibrinogen levels in blood, these changes provide evidence for the plausibility of PM-induced heart failure in human high-risk groups.

Recent intratracheal instillation in experimental animal studies show that in healthy animals and animals with cardiopulmonary diseases a high dosage of 1000–5000 µg/rat of fine and coarse particles (~ 0.4–10 µm; mineral dusts, PM collected from ambient air or from anthropogenic, combustion sources (fly ash)) cause cytotoxicity, inflammation and the production of oxygen radicals in lungs, as well as myocardial toxicity and arrhythmic changes (Hatch *et al.*, 1985; Ghio *et al.*, 1992; Ghio and Hatch 1993; Li *et al.*, 1996, 1997; Watkinson *et al.*, 1998; Killingworth *et al.*, 1997;

Kodavanti *et al.*, 1997; Murphy *et al.*, 1998; Ulrich *et al.*, 2002). In principle, this provides a plausible explanation for PM-associated health effects, although the exposure levels exceed the maximum obtainable 24-hour dose by inhalation of ambient PM. These studies also indicate that chemical composition is an important determinant with respect to the toxic effects.

4.3.2. Health effects of PM model compounds

In epidemiological studies ambient PM seems to be a very potent agent, but the nature of the causative fractions of ambient particles is unclear. To answer the question of whether the causal fraction is of a physical, chemical and/or biological nature, studies were conducted using specific constituents or groups of constituents with distinct diameters. A better understanding of the role of the nature of PM could provide answers to the following questions:

- Are toxic effects a result of the intrinsic toxic potential or the dose or both?
- Is there evidence of an exposure-response relationship with the assumed toxicant(s), especially at low concentrations?
- Can results obtained with a restricted number of PM samples and/or locations be generalised from one PM sample or locale to another?

Table 4.5 presents the prevailing hypotheses for different causative fractions of PM. They will be examined in more detail in the following sections.

Table 4.5 Prevailing hypotheses of responsible fractions of PM

PM acidic properties	Schlesinger, 1996
Ultrafine (UF) fraction (< 0.1 µm) of PM_{10}	Oberdörster *et al.*, 1994
Organic fraction	Lewtas, 1993; Sagai *et al.*, 1993
The presence of biological materials	Targonski *et al.*, 1995
Particle-gas combinations	Kleinman *et al.*,1998; Last *et al.*, 1991a,b; Brook *et al.*, 2002
Transition metal content	Pritchard *et al.*, 1996; Dreher *et al.*, 1997; Costa and Dreher, 1997; Kodavanti *et al.*, 1998
PM deposition leads to 'hot-spots' and exaggerated regional damage in the lung	Bennett *et al.*, 1996
Electrical charge of PM	Cohen *et al.*, 1998; Oortgiesen *et al.*, 2000; Veronesi *et al.*, 2002a, b

This section highlights the parts of PM that have received most attention recently, but does not pretend to cover all the available evidence regarding possible causal fractions.

4.3.2.1. Effects caused by physical characteristics

In the last few decades attention has focused mainly on mass concentrations as being the best descriptive parameter in epidemiological studies. Recently, however, surface

area or number concentrations have also received more attention. In particular, number concentrations for ultrafine particles have been used successfully in several epidemiological studies (Section 3).

In the field of toxicology little is known about the relevance of surface area or number concentrations. The diameter as well as the shape (spheres versus fibres) of insoluble particles determines whether or not the particles can cross (respiratory) epithelia in significant amounts. Two mechanisms of direct activity of particles have been proposed, however. Bennet *et al.* (1996) proposed that specific deposition of PM in the lung leads to 'hot spots'. They showed a distinctly non-homogeneous retention pattern of particle aggregates, with an accumulation of this material along lymphatics in lung sections of elderly people. The issue of 'hot spots' is a part of studies focusing on the dosimetry of particles. Another mechanism of direct action might be the interaction of particle charge with receptor molecules.

a. PM charge or radiation

Oortgiesen *et al.* (2000) hypothesised that negatively charged PM, like ROFA and synthetic polymer microspheres, trigger irritant receptors in airway target cells *in vitro*. This in turn might be related to neurogenic airway inflammation. The same group also described that the surface charge carried on PM was highly correlated with the release of inflammatory mediators, i.e. cytokine IL-6 (Veronesi *et al.*, 2002a) in cultures of human respiratory epithelial cells. Cohen *et al.* (1998) investigated the effect of particle charge with ultrafine PM and concluded that about five times higher deposition can occur compared with zero-charged (at Boltzman equilibrium) particles. Since most ambient particles carry one, or a few, charges, deposition models might underestimate the pulmonary deposition of these small particles. Charged nano particles, whose surface was modified with cationic compounds, have also been shown to be more effective in cardiovascular drug delivery (Labhasetwar *et al.*, 1998). The toxicity of a uniform exposure atmosphere of silica particles has been described to be caused by the positive charge they carry (Bagchi, 1992). Titanium oxide discharge reduced particle deposition in the respiratory tract by 20% (Ferin *et al.*, 1983). Altogether, these few studies call for a better exploration of the role of particle charge as an important denominator in the delivered target dose of PM and also as an inducer of inflammatory processes. This phenomenon has been reviewed (Veronesi and Oortgiesen, 2001) and it has been suggested that PM initiates airway inflammation through sensory neural pathways, in particular by activating acid-sensitive irritant receptors. The released neuropeptides may affect both immune and non-immune airway target cells. However, PM-induced health effects have not sofar been shown to be induced by neurogenic pathway.

Recently, Marcazzan *et al.* (2001) described in a paper on the characterisation of PM_{10} and $PM_{2.5}$ in the ambient air of Milan (Italy) that they had observed a high temporal correlation between ^{222}Rn concentrations and PM_{10} and $PM_{2.5}$ concentrations. The correlation between daily averages (N=8) of both particulate measures and radon daughters could be estimated to be more than 0.9 for the month of February 1998 in Milan. A similar correlation could be expected in the Netherlands too: a correlation coefficient of 0.8 in Wijnandsrade, the Netherlands between daily PM_{10} and the average equilibrium equivalent decay concentration of the radon daughters (EEDC)

during an episode (Smetsers *et al.,* 1996). Radon daughters are closely linked with free radical formation, the mechanism of which is described in 4.4.
Exposure to these α particles has recently been associated with the generation of extracellular and intracellular reactive oxygen species (ROS) (Narayanan *et al.,* 1999). In addition, Narayanan *et al.* (1997) found significant increases in the generation of IL8, an oxidative stress-related compound as early as thirty minutes after the irradiation of cultures of normal human lung fibroblasts.

b. PM diameters

One of the hypotheses that is receiving increasing attention is that UF (< 0.1 μm) insoluble particles freshly generated in ambient air by combustion processes or by gas-particle conversion reactions are responsible for adverse health effects. The early studies with polytetrafluoroethylene fumes (Oberdörster *et al.,* 1996) in particular drew attention to this size fraction. Moderate support for this hypothesis was recently provided by epidemiological studies (Peeters *et al.,* 1997; Pekkanen *et al.,* 1997) as well as by animal inhalation studies (Oberdörster *et al.,* 1992; Oberdörster *et al.,* 1996; Ferin *et al.,* 1992). However, Arts *et al.* (2000a,b) and Cassee *et al.* (2002a) compared the effect of particle diameter and number concentrations for both insoluble carbonaceous particles and highly soluble ammonium salts and came to the conclusion that if any effect was observed, this was more likely to occur with fine mode particles than ultrafine mode particles regardless of the chemical composition. Studies using ultrafine TiO_2 , however, showed marked pulmonary inflammation, possibly related to their large surface area and increased interstitial access (Oberdörster *et al.,* 1992). An inhalation study with short-term exposures to near ambient mass and number concentrations of ultrafine metal oxide and carbon particles failed to show any pulmonary biological response in healthy rats, suggesting that the generic, physical nature of ultrafine particles is not important for pulmonary effects (Roth *et al.,* 1998; Ziesenis *et al.,* 1998; Utell and Frampton, 2000; Oberdörster *et al.,* 1996).

To study the role of particle size in the toxicity of inhaled PM, studies using the highly toxic chemical cadmium chloride ($CdCl_2$) were conducted (Cassee *et al.,* 2002b). Subsequently, the multiple path deposition (Cassee *et al.,* 1999) dosimetry model was applied to explain the differences in toxicity in rats that have been exposed to cadmium chloride aerosols at equal mass concentrations but with different particle diameters. The results of the study showed that the pulmonary toxicity of $CdCl_2$ did not depend on particle size as such but on the actual dose deposited, and therefore related more to the intrinsic properties of the aerosol. Studies using aggregates of ultrafine and fairly inert particles such as TiO_2 and carbon black (Ferin *et al.,* 1992; Mauderly *et al.,* 1994; Nikula *et al.,* 1995; Oberdörster *et al.,* 1992) suggest that the concentrations that cause adverse health effects are usually above 1000 $\mu g/m^3$.

Whereas the epidemiological studies associate PM_{10} or $PM_{2.5}$ with health effects, a rapidly increasing number of toxicological studies focus on the three size fractions within PM_{10}. Most of these studies apply either concentrators for inhalation studies or novel PM sampling techniques for *in vitro* or *in vivo* health effects studies.

In contrast with other views frequently presented, coarse particles (2.5–10 μm) may therefore still serve as important causal candidates for some of the health effects

occurring in upper airways, lower airways and lungs. Apart from the outcome of the dosimetry models, recent data from a small number of epidemiological studies indicate that health effects are also associated with the coarse PM fraction and sometimes even to a larger extent than with the fine PM fraction (Loomis *et al.*, 1999). It is not yet known which type of PM-associated health effects may be linked with this fraction, but some of the asthma and infection responses might be a possibility.

Becker *et al.* (2002) studied the potency to induce inflammatory mediators of coarse, fine and ultrafine ambient PM. They observed the strongest effects in the coarse fraction and found an absence of effect from UF. The authors suggest that the effects are linked with the presence of microbial cell structures and endotoxins. Since bronchoconstriction is a clear symptom in people with COPD or asthma, and dosimetry models predict that the tracheobronchial airways are also a target for the PM deposition of particles > 1 µm, there may be a relationship between coarse mode PM and bronchoconstriction. The same group (Dailey *et al.*, 2002) also studied the effects of the three size fractions in airway epithelial cells. Interestingly, coarse and UF mode PM induced stronger responses (cytokine production) than the fine mode, again with the coarse mode PM being the most potent fraction. Li *et al.* (2002) described that coarse and fine mode particles collected in Downey, California produced different effects in an oxidative stress model. Also, the effects of coarse mode particles were most effective when collected in the autumn and winter.

Some suggestions have been made that particles in the micron size range may be loaded with ultrafine particles on the surface. The implications are not yet clear: either the large particles carry the ultrafines to different locations or target tissues (e.g. macrophages instead of epithelial cells) or larger particles act as filters for ultrafines so that they will not be a threat for human health anymore. Clearly, more attention needs to be devoted to this phenomenon.

c. PM surface area

Very little information has been published on the role of the surface area of PM. It is not a parameter used in air quality measurements. The surface to mass ratio increases with decreasing particle diameter. As Donaldsen *et al.* (1998) mentioned, the large surface area provides the opportunity for increased contact with the lung surface and subsequently for the chemicals on the particle surface to have a significant effect. For example, Oberdörster *et al.* (1992, 1994) have demonstrated that particle-induced lung injury caused by ultrafine TiO_2 is related to their larger surface area.

4.3.2.2 Effects caused by biological characteristics

It is sometimes speculated that biological agents are the main cause of the effects in human populations. Pollen concentrations, for instance, have been shown to correlate well with mortality in the Netherlands (Hoek and Brunekreef, 1999). A limited number of data are available from mechanistic animal toxicity studies using diesel exhaust particles in combination with allergens. Submicron fragments of pollen grains suggest that small particles act as transport vectors (Knox *et al.*, 1997; Behrendt *et al.*,

1992, 1997). Significant attention has been paid to the adjuvant action of particles (see 4.3.2.2a below).

Mohn and Becker (1999) presented *in vitro* data suggesting that coarse ambient PM was more potent than fine PM in provoking inflammatory responses in which endotoxin may play an important role. Recently, Becker *et al.* (2002) concluded that PM recognition by human alveolar macrophages involved receptors evolved to recognise microbial cell structures, and that microbes preferentially found in the coarse particle fraction of PM might be involved in inflammatory events associated with exposure to pollution particles. In the areas around rubbish tips high concentrations of bacteria and fungi are usually found in the air. Although this often results in complaints about stench, it is not associated with airway or cardiovascular diseases in people living in the neighbourhood. It is well-known that endotoxin (LPS) inhalation may induce a neutrophilic airway inflammation (Pauwels *et al.*, 1990). Several papers have looked at the effect of endotoxin in PM samples (Van Eeden *et al.*, 2001). In these studies, it was made very clear that cytokine induction was not explained by the presence of endotoxins (20–30 ng/mg PM). The overall conclusion is that although PM contain biological material such as endotoxins, ambient levels cannot induce pulmonary effects.

PM as adjuvant – infection sensitivity

Over the past 20–30 years the prevalence of asthma has increased some 50% every ten years. In the same period, the prevalence of hay fever has increased rapidly (Jarvis and Burney, 1998). Perennial and seasonal allergens, e.g. house dust mite (HDM) and pollen, are potentially able to trigger allergic asthma reaction (Jarvis and Burney, 1998; Newson *et al.*, 1998, Marks, 1998). Of all allergens, exposure to HDM might be the most important as a risk factor for developing asthma (Marks, 1998).

Diesel exhaust particles (DEP) are a major component of ambient air pollution and respirable particles. A large part consists of freshly generated DEP < 0.1 µm in diameter and this fraction can be carried deep into the lungs when inhaled. One of the first to show epidemiological evidence for the adjuvant activity of DEP was Ishizaki *et al.* (1987). He showed that the incidence of allergic rhinitis caused by pollen was higher in areas with air pollution than in non-polluted areas, while the pollen counts in both areas were almost the same. It is likely that the pollen (fragments) are coated with DEP rather than the other way around. Ormstad *et al.* (1998) showed that DEP had the ability to adsorb cat, dog, birch pollen and HDM allergen *in vitro*. Moreover, DEP aggregates have been identified on pollen grains isolated from ambient air (Behrendt *et al.*, 1997; Knox *et al.*, 1997).

Experiments have shown that specific IgE responses in mice and rats immunised with ovalbumin (OA) and HDM mixed with DEP were stronger than when they were immunised with only OA or DEP (Muranaka *et al.*, 1986; Takafuji *et al.*, 1989; Fujimaki *et al.*, 1997; Nilsen *et al.*, 1997; Steerenberg *et al.*, 1999; Van Zijverden *et al.*, 2000). This agrees with the observations of Diaz-Sanchez *et al.* (1994 and 1997), who demonstrated that in non-allergic and in ragweed-allergic persons DEP increased total and specific IgE production respectively, and increased the production of

cytokines. In addition, many studies have shown that DEP exposure without allergen increases the production of cytokines and cells, making the host more vulnerable for reaction to allergens (Nel *et al.*, 1998). It therefore seems that adsorption of allergens to airborne particles is not always a prerequisite for adjuvant activity of allergens. In addition, antioxidants seem to prevent the adjuvant effect (Whitekus *et al.*, 2002).

4.3.2.3. Effects caused by chemical characteristics of PM

a. Secondary inorganic (acidic) particulate matter (SIA)

A large portion of ambient PM in the Netherlands consists of secondary inorganic aerosol (SIA). These are produced in the atmosphere via chemical reactions involving gaseous pollutant precursors, and are dominated by such chemical species as acidic sulphates and nitrates and their salts. The next section is to a large extent based on a recent survey of the literature (Schlesinger, 2000, Annex B; Schlesinger and Cassee, 2002) in which the health effects arising from exposure to specific fractions of secondary inorganic aerosol constituents are based on available data from peer review published papers as well as publicly available reports on controlled animal and human clinical exposure studies involving these chemical species. The acidity of ambient particulate matter in the Netherlands is relatively low, due to neutralisation by ammonia, and the major components of particulate matter are ammonium sulphate and ammonium nitrate. The evaluation was made for pure sulphates, nitrates and ammonium, and does not consider any possible interactions between other gaseous or particulate pollutants.

Sulphate and sulphuric acid

The toxicological database for sulphate and sulphuric acid is sufficient for a risk assessment from a toxicological perspective. Exposures to acid sulphates have produced transient changes in pulmonary function in asthmatics, including enhanced non-specific airway responsiveness, and epidemiological evidence indicates that the exacerbation of symptoms in asthmatics may be related to atmospheric particulate acids. The concentrations of acidic sulphates needed to produce any effects in controlled exposure studies are generally well above those found in the ambient air in the Netherlands. Short-term exposures of healthy animals (with the exception of the guinea pig) to H_2SO_4 concentrations of 1000 $\mu g/m^3$ (< 1 μm diameter) generally do not alter standard lung function tests (US-EPA, 1989). Similarly, healthy adult humans show no consistent effects on pulmonary function or respiratory symptoms with acute exposure to H_2SO_4 at < 1000 $\mu g/m^3$, even with exercise (Avol *et al.*, 1988a; Frampton *et al.*, 1992; US-EPA, 1989). On the other hand, there is some evidence that asthmatics may be more sensitive than healthy individuals to effects on lung function, and that they may experience modest bronchoconstriction following exposure to H_2SO_4 at < 1000 $\mu g/m^3$ (Avol *et al.*, 1988a,b ; Koenig *et al.*, 1993; Linn *et al.*, 1989; US-EPA, 1989). While basic lung functional indices may not be affected by acid exposure in normals, an increase in airway responsiveness was observed with bronchoprovocation challenge following acute exposure to 1000 $\mu g/m^3$ H_2SO_4, and in

some adult asthmatics following exposure to 100 μg/m^3 (Utell *et al.*, 1983). But, as above, there appears to be no consistent effect of acute exposure on airway reactivity in either healthy or asthmatic individuals (Avol *et al.*, 1988a,b ; Linn *et al.*, 1989).

Although epidemiological studies suggest there may be segments of the population that are susceptible to inhaled acidic sulphates, toxicological studies generally used healthy adult animals, and very limited data are available to allow evaluation of the effects of different disease states, other than asthma, upon response to acid sulphate particles. Acute exposure to H_2SO_4 at levels as low as 100 μg/m^3 alters mucociliary transport in normal humans (Leikauf *et al.*, 1981; Spektor *et al.*, 1989), without altered particle clearance from the alveolar region. These effects may ultimately be reflected in alterations in the ability of these cells to adequately perform their role in host defences. There is also some evidence that sulphuric acid reduces resistance to bacterial infection, but this seems to depend upon the animal model used (Zelikoff *et al.*, 1994).

The exact characteristics of ambient acidic PM in epidemiological studies are not certain, and those used in most controlled studies may differ from those to which populations are actually exposed. The relative potency of acidic sulphate aerosols is related to their degree of acidity, i.e. the H^+ content within the exposure environment (Koenig *et al.*, 1993; Schlesinger, 1984, 1989; Schlesinger *et al.*, 1990). Furthermore, the number concentration of particles within an exposure atmosphere, as well as the total mass concentration of H^+, is an important factor in determining response following inhalation of acidic sulphates (Chen *et al.*, 1995). Most controlled exposure studies involved pure acidic sulphate droplets. However, in the ambient air acidic sulphates may also occur as a coating on the surface of other particles, such as metals or carbon, especially within combustion aerosols. It has been shown that up to an order of magnitude higher exposure levels of pure acid aerosols were needed to produce comparable biological results in animals than when the exposures involved acid coated on a solid particle (Amdur and Chen, 1989). Such exposures to surface-coated acid produced biological effects at concentrations as low as 20 μg/m^3 (as H_2SO_4). Thus, it is likely that the physical nature of the inhaled acid particle is a key factor in determining ultimate response.

Many of the controlled animal studies and all of the human clinical studies involved acute exposures. The available evidence suggests that the minimally effective concentration of sulphuric acid to alter pulmonary mechanical function, including non-specific airway responsiveness, in normal humans following acute exposure is > 1000 μg/m^3, but in asthmatics it may be around 68–100 μg/m^3. However, effects on asthmatics, especially at these low concentrations, are quite inconsistent. This may be due to the normally large variability in asthmatic responses to low level air contaminant exposure and also to susceptibility differences within segments of the asthmatic population. For example, elderly asthmatics do not seem to be especially susceptible, but adolescent asthmatics may be more susceptible. In any case, the extent of effect on pulmonary function is small at low acid exposure concentrations, with changes of < 10% in one commonly measured parameter, namely FEV1, following exposures of asthmatic subjects to sulphuric acid aerosols at concentrations 500 μg/m^3. There are likely to be no adverse or irreversible effects, as far as

cardiopulmonary function is concerned, from ambient levels of sulphate or nitrate aerosols, even in presumably more sensitive asthmatics.

Another biological endpoint which has been extensively examined with acute exposure of humans is mucociliary clearance from the tracheobronchial tree. This has been shown to be transiently altered in normal individuals by sulphuric acid aerosol at a concentration of 100 $\mu g/m^3$, with no evidence for any susceptibility difference for asthmatics. The nature of the effect at this concentration can be acceleration or slowing of clearance, depending upon the region within the tracheobronchial tree that is being examined. However, the pathological significance, if any, of such transient effects is not certain.

Perhaps more relevant to repeated ambient exposures are the results of longer-term controlled exposure studies. These all involved animals and indicate the potential for the production of non-specific airway hyperresponsiveness, persistently retarded mucociliary clearance and changes in airway secretory cell function with repeated exposures to sulphuric acid at concentrations ranging from 100–250 $\mu g/m^3$. The development of hyperresponsive airways in healthy animals at exposure levels below that producing any change in standard lung function indices may have implications for the pathogenesis of airway disease, and alterations in mucociliary function could have implications in terms of the development of chronic obstructive pulmonary disease. However, as noted, considerations of dose equivalency must be included in any evaluation in this regard.

The suggestion can be made that the controlled exposure studies do not adequately resemble the ambient conditions in which PM should be regarded as a very complex mixture of particulate matter. For a few years now, both animal and human exposure studies have been performed using concentrated ambient PM (see 4.3.1). Most of these studies determine sulphates and nitrates in a manner similar to that for air quality measurements. Unfortunately, very little information has so far been published in the literature, but a few examples can be discussed here.

Recently, twelve normal subjects were exposed to CAPs at an overall mass concentration ranging from 99 to 215 $\mu g/m^3$ at the University of Southern California. The corresponding nitrate measurements ranged from 16 to 72 $\mu g/m^3$, and sulphate from 1.7 to 17.5 $\mu g/m^3$. In California, nitrate was positively correlated with total PM mass ($R = 0.58$), but sulphate was not ($R = -0.19$). The group showed small and equivocal biological responses, if any, to CAPs exposure. Comparing these results with Table 2.1 (total SIA 10–14 $\mu g/m^3$), one might assume that SIA did not cause a serious adverse effect in healthy humans, albeit that the number of measurements is limited (Linn, personal communication).

The daily dose for inhaled Dutch ambient sulphates is at most 200 µg. Sulphates are highly soluble components that will cross the lung-blood barrier rapidly. The amount that is deposited in the airways is most likely only a fraction of what is already present in the body. For instance, sulphate itself is already present in concentrations ranging between 240 and 420 micromol/litre in serum. There are numerous endogenous forms of sulphate known, including some that are used in a medicinal capacity. For example, the infusion of magnesium sulphate ($MgSO_4$) in healthy volunteers has been studied

for its ability to inhibit arterial wound healing (Ravn *et al.*, 1996). Although a transient decrease in blood pressure was observed during the initial bolus infusion of $MgSO_4$, this effect was due to magnesium rather than sulphate. These facts also make it less likely that airborne sulphate at ambient levels is a serious threat to human health.

All these studies have focused on the role of acidity of H_2SO_4. In the Netherlands, about 80% of the secondary inorganic aerosol is found within the fine mode of PM_{10} (Visser *et al.*, 2001). The annual average PM_{10} mass concentration ranges from 27 to 42 $\mu g/m^3$ and of this the secondary inorganic particulate component accounts for about 10–14 $\mu g/m^3$ (Keuken *et al.*, 1999; Weijers *et al.*, 2000; Visser *et al.*, 2001). Since the acidity of ambient particulate matter in the Netherlands is relatively low, due to neutralisation by ambient ammonia, the major secondary components are ammonium sulphate and ammonium nitrate.

Nitrate

The toxicological database on health effects from inhaled nitrates is very limited. Those studies which did evaluate toxicological responses generally involved exposure to nitric acid in the vapour state (e.g. Abraham *et al.*, 1982; Koenig *et al.*, 1983; Nadziejko *et al.*, 1992; Aris *et al.*, 1993; Schlesinger *et al.*, 1994; Wong *et al.*, 1996). These exposures (in the g/m^3 range) were noted to produce various effects on pulmonary functional and lung defence parameters. These effects can be attributed to the acidity rather than to nitrate itself. As mentioned in the previous paragraph, the acidity of secondary inorganics is largely reduced by ammonia. Dose estimates for inhaled Dutch ambient nitrate are in the order of 100 µg or less. There are not likely to be any adverse effects, as far as measured cardiopulmonary function is concerned, from ambient levels of nitrate aerosols, even in presumably more sensitive asthmatic members of the general population (Annex B). It should, however, be noted that some of the potentially more sensitive cardiopulmonary indices of response, such as heart rate variability (HRV), have not been assessed in controlled studies. Nitrate in itself is not very toxic, but the principal effects may arise as a result of the conversion of nitrate to nitrite. Nitrite is formed from nitrate by bacterial conversion in the oral cavity and/or stomach and not in the respiratory tract. Nitrite oxidises haemoglobin to methaemoglobin, which interferes with the transport of oxygen by the blood (methaemoglobinaemia). There are also large uncertainties about the conversion of nitrate together with other gaseous pollutants in ambient air into more potent substances like radical forming agents. This evaluation focuses only on the pure nitrate as a causative agent. Clearly, information on atmospheric chemistry and the actual nitrogen species to which people are exposed are largely unknown. Consequently, it is not yet possible to estimate the risks of nitrogen species covered by nitrate as a proxy.

Although the route of exposure may have profound implications for the effect at the target site, systemic[1] effects based on systemic nitrate delivery as a result of airborne

[1] Assuming a 50% deposition efficiency in the respiratory tract, a 24-hour outdoor exposure, a minute volume of 15 lpm and an ambient nitrate level of about 10–14 µg/m3, average daily intake of nitrate by inhalation is in the range of 100–150 µg/day. However, the exposure pathway for nitrate is estimated to be 95% through the diet. Dietary nitrate intake has been estimated to be 50 000–140 000 µg/day,

nitrate exposures do not seem likely. This alone does not rule out the fact that nitrate may very specifically affect the lung tissue. The studies that have been published on nitrate exposure do not provide clear evidence for risk of exposure to ambient aerosolised nitrate.

Ammonia

The average ammonia concentration in air in the Netherlands of 3.6 $\mu g/m^3$ is substantially below the threshold limit value (TLV) of 25 ppm (about 18 000 $\mu g/m^3$) recommended by ACGIH (1996), which is intended to protect against ocular and respiratory irritation and may serve as the basis for a Reference Dose. Since 360 $\mu g/m^3$ is the lower bound limit of the range of concern, it should not be necessary to apply an uncertainty factor of 10 for unusually sensitive individuals. Using a 70 kg human reference body and an inhalation rate of 20 m^3/day, the corresponding dosage is 100 µg/kg/day or 7000 µg/day for ammonia. This is an estimate for a healthy worker. An additional safety factor of 10 may be applicable to protect risk groups.

Considerations

In the evaluation of effects from ambient pollution, an important consideration is the potential for special susceptibility of specific subgroups within the general population. Adolescent asthmatics may represent a sensitive segment of the population with respect to the bronchoconstrictive effects of fine mode acidic aerosols. Controlled exposures have produced transient changes in pulmonary function in asthmatics, including enhanced non-specific airway hyperresponsiveness in some cases. Although epidemiological evidence indicates that the exacerbation of symptoms in asthmatics may be related to atmospheric particulates, the contribution of chronic ambient particulate exposure to the development of airway hyperresponsiveness in normal individuals remains unclear.

There are, however, some caveats in this overall evaluation of potential health effects from exposure to ambient secondary aerosols. It is important to consider the relationship between animal exposure studies and actual human exposures, both in terms of particle size and inhaled dose. It is also necessary to consider the physicochemical characteristics of chemical species in the ambient air compared to those used in controlled studies. Although sulphates themselves may seem harmless, we cannot rule out the fact that they carry other, reactive constituents on their surface. Clearly, atmospheric chemistry should try to find out in what way sulphates appear in ambient air. As a final point, the potential for interactions between particulate matter and ambient gases must be considered in developing conclusions as to effective levels of the former.

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depending on the quantity of vegetables, which implies that intake through the respiratory system is roughly 70 µg/day. The Acceptable Daily Intake (ADI) for nitrate set by the European Commission's Scientific Committee for Food is 1600 µg/kg/day (Ysart *et al.*, 1999). These values are even lower for sulphate. In addition, assuming a drinking-water consumption of 2 l/day and a daily consumption of 100 g of vegetables, overall daily nitrate consumption may easily range from 200 000–400 000 µg. It is clear that nitrate consumption exceeding the ADI is not just a hypothetical issue but will occur frequently. From a statistical exposure model, Slob (1995) has shown that in adults 15% of daily intakes regularly exceed the ADI. In young children this can rise to 45 %.

The issue of exposure concentration (C) and duration (T) comes into play when evaluating acute exposure responses in terms of circumstances where repeated exposures are the pattern of concern. Evaluation of the role of C and T must be performed using the acute exposure database, since the database for chronic exposures is much too sparse in this regard. The response to acute exposures, at least in terms of their effect on two of the most commonly evaluated endpoints, namely tracheobronchial mucociliary clearance and respiratory region clearance, appears to be a function of both C and T. In the rabbit, for example, a concentration of 75 $\mu g/m^3$ sulphuric acid was at or below the no-observed-effect level for altering mucociliary clearance, even when the C x T was equivalent to that obtained using concentrations at 100 or 200 $\mu g/m^3$. Exposure to 50 $\mu g/m^3$ for two hours a day for fourteen days was found to be ineffective in altering alveolar region clearance. It appears that a threshold exists for both the number of deposited acid particles as well as the mass concentration needed to produce any biological response, at least for some endpoints.

One important consideration in evaluating the health effects from secondary inorganic aerosols is this issue of threshold. The current epidemiological database for ambient particulate matter suggests that the concentration-response relationship presents no indication of a clear threshold. However, for some specific chemical constituents of ambient particulate matter, a threshold may exist. For example, the occurrence of a threshold concentration, exposure to which would not result in any effect regardless of the exposure duration, is not unexpected for acidic particles. The response to acid sulphates and other inorganic acidic chemical species is likely due to the deposition of hydrogen ions on airway surfaces. The extent of available hydrogen ions may be altered in the inhaled air, or once the aerosol deposits on airway surfaces, due to the presence of endogenous ammonia in the respiratory tract and buffers in airway surface fluids respectively. Thus, it is likely that a certain threshold concentration is needed to overcome these processes and result in deposition of sufficient hydrogen ions so as to alter localised airway surface or cellular pH. The occurrence of such a threshold may also contribute to the inconsistencies which are often noted in human exposure studies involving low acid concentrations.

In addition to a threshold for exposure concentration, a threshold for exposure duration also seems to occur, such that acute exposure for longer than this critical time is needed at some effective exposure concentration in order to overwhelm the buffering capacity of the airway surface fluids and to produce an observable response. For example, exposures to sulphuric acid aerosol for 30–40 min at 1000 $\mu g/m^3$ were required before any effect on mucociliary clearance could be observed in animal studies (Schlesinger *et al.*, 1978). Thus, a normal individual should be able to tolerate certain exposure regimes with no observable effect. However, the particular C and T values which do not result in observed responses could differ for people with airway disease, and the product is probably not a constant value. Furthermore, the specific threshold for C or T probably depends upon both the sensitivity of the endpoint being examined and the exact exposure protocol.

The available database for acidic sulphates strongly suggests that long duration exposures to low concentrations and short duration exposures to high concentrations may not necessarily be toxicologically equivalent. This is likely to apply to other inhaled acidic particulates as well. Given this database, it can be concluded that acute

effects at 100 ug/m^3 on potentially sensitive human asthmatics are small and very inconsistent, and that the observed change in airway epithelial secretory cells in rabbits chronically exposed at 125 µg/m^3 was not associated with a persistent or permanent alteration in mucociliary transport function. Taking into consideration similarities in the relationship between animal and human effects when responses have been directly compared, a threshold NOAEL level of 50 µg/m^3 may be appropriate for the irritant effects of pure sulphuric aerosol droplets in humans. However, there is the caveat that effects may occur at lower levels if acid is coated on a solid "carrier" particle.

The toxicological database does not as yet support a role of ambient acidic sulphates in adverse health outcomes noted in epidemiological studies, at least from the viewpoint of effective exposure concentrations. Levels of acidic sulphates needed to produce any effect in controlled studies are well above those found in ambient air. However, the exact physicochemical characteristics of ambient acid particulate matter in the epidemiological studies, and to which human populations are normally exposed, may differ from those used in the controlled toxicological studies, and such characteristics will probably affect the exposure concentration-response paradigm. For example, as noted, most controlled studies used pure sulphuric acid, droplets, but in ambient air these may be neutralised or occur as a pure acid surface coating on other particles. Furthermore, the most direct measure of acid aerosol is the strong hydrogen ion concentration of ambient particulate matter, and this parameter is difficult to accurately assess at the low ambient levels at which it occurs. Thus, it has not been widely used and this limits those analyses that can directly assess the potential health effects from strongly acid aerosols (Gwynn *et al.*, 2000). However, the acidity of ambient particulate matter in the Netherlands is in any case relatively low, and the major components of secondary ambient particulate matter are ammonium sulphate and ammonium nitrate.

It should be noted again that many controlled exposure studies used concentrations of particulate matter that were much higher than those occurring in ambient air. Thus, some of the mechanisms elicited may not occur with exposures to lower levels. Clearly, controlled exposure studies have not as yet been able to unequivocally determine the particle characteristics and the toxicological mechanisms by which ambient particulate matter may affect biological systems. Furthermore, one must always consider that with ambient exposures, health outcomes may be the result of interactions between secondary particulate matter with other co-existing ambient gaseous pollutants, such as ozone (Thurston *et al.*, 1992, 1994; Bates and Sizto, 1995; Schlesinger, 1995; Frampton *et al.*, 1995). The role of gaseous pollutants in association with particulate matter in eliciting health effects seems to be restricted (Sarnat *et al.*, 2001).

Conclusions on SIA

Based on a review of more than 150 publications (Schlesinger and Cassee, 2002), it is concluded that the toxicological database does not support a causal role for ambient acidic sulphates in adverse health outcomes noted in epidemiological time-series studies. Levels of acidic sulphates needed to produce any effect in controlled studies in experimental animals or human volunteers are well above those found in ambient

air, especially in the Netherlands. The ranking of irritant potency of the sulphates in terms of alterations in various aspects of lung effects (clearance function) appears to be sulphuric acid > ammonium bisulphate > ammonium sulphate, sodium sulphate. In Annex B (Schlesinger, 2002) the conclusion is drawn that a threshold exists for the most potent component of the list: sulphuric acid. This implicitly leads to the conclusion that for the other less potent fractions such a threshold should exist too. Also, preliminary data from controlled human exposure to CAPs does not support a dominant role of SIA in inducing adverse health effects at ambient levels. Toxicological studies of these other constituents of the secondary fraction, as well as metal sulphates, ammonium bisulphate and ammonium nitrate, suggest that these have little toxic potency at environmentally relevant levels in normal humans or animals or in the limited compromised animal models used.

b. Crustal material

Very little is known about the toxicity of the crustal fraction within PM. *In vitro* studies (Green *et al.*, 1982; Holian *et al.*, 1998) showed that Mount St. Helen dust did not induce any effect at concentrations where two urban PM samples caused apoptosis of alveolar macrophages. In the same study, ROFA was shown to be at least ten times more toxic than the urban dust samples. Inhalation studies in rats exposed to 9400 $\mu g/m^3$ size-fractionated Mount St. Helen ash for five days (2hr/day) revealed no changes in pulmonary function and histology for periods of up to one year. Similar results were reported in mice (Grose *et al.*, 1982). In addition, intratracheal instillation studies in the rat with fine and coarse mode Mount St. Helen dust produced only minor pulmonary functional changes, histologically detectable alveolitis, and small increases in lung weight with coarse-mode PM. The weak toxic potency of crustal material is also reflected in epidemiological studies (Merchant *et al.*, 1982). Laden *et al.* (2000) concluded that fine crustal PM is not associated with increased mortality in the six cities study. Places where PM_{10} is composed primarily of crustal material and volcanic ash show similar risk ratios for morbidity (Gordian *et al.*, 1996) and it was concluded that the coarse fraction of PM_{10} may affect the health of working people. According to some authors, the coarse fraction PM does not appear to be associated with mortality (Schwartz *et al.*, 1999).

c. Sea salt

A part of PM consists of sea salt, a source that cannot be influenced by emission control strategies. Sodium chloride, and to a lesser extent potassium chloride, are the dominant constituents of sea salt and are generally considered to be harmless at ambient concentrations of a few $\mu g/m^3$. Due to its high solubility, sodium chloride is rapidly dissolved in the fluid lining the epithelia of the respiratory system. A recent review (Annex B) on the toxicity of sea salt spray suggests that ambient levels of sodium chloride are well below those shown to produce any effects in toxicological studies. In fact, these aerosols are generally used in toxicological studies as the presumably negative or vehicle control.

d. Organics

The organic, non-biological constituents of PM comprise a wide range of chemicals. However, the amount of scientific literature on health effects of organic particulate matter is far from sufficient to judge its hazard. Most of what has been published is related to diesel exhaust particles, but the total contribution of DEP to the total carbonaceous fraction (OC) is not known. It is also not valid to use gas-phase short-term exposure studies for this purpose, since the kinetics are usually very different from particle-associated organics. For instance, Bevan and Ruggio (1991) revealed that 50% of the benzo[a]pyrene on diesel exhaust particles was retained in the lungs three days after intratracheal instillation, whereas the remainder was either excreted or distributed throughout the other organs of the rat. Results like these imply that organic material adsorbed to the surface of insoluble particles are only slowly released once deposited in the respiratory tract.

e. Transition metals and fly ash

As previously described in Section 3, two historical experiments, one in Utah Valley and the other in Eastern Germany, supplied epidemiological evidence for a role of transition metals or other air pollution as causal for PM-associated health effects (Pope *et al.*, 1999; Heinrich, *et al.*, 2000). However, the results of these two studies are very specific to the geographical location. Usually, the concentrations of transition metals in the Netherlands are orders of magnitude lower (Visser *et al.*, 2001). It is therefore very unlikely that the presence of these metals in all kinds of ambient PM could explain observed human health effects in the Netherlands.

Transition metals, primarily iron, greatly enhance the generation of oxygen-derived free radicals, which can destroy a large variety of biomolecules, in particular the lipid components of cell membranes. Nitric oxide has a high affinity for transition metals. Investigational evidence for the importance of oxidative damage to the mechanisms of metal toxicity and carcinogenicity is especially robust for the essential metals iron and copper at high concentrations and also for three indisputable human metal carcinogens (nickel, chromium, and cadmium) (Buzard and Kasprzak, 2000). The oxidative potency of these types of particles to induce lung toxicity *in vivo*, such as iron ions adsorbed to titanium oxide, ambient PM from various locations, or residual oil fly ash, seems to be proportional to the content of some of the ‘first-row’ transition metals associated with the particles (Keeling *et al.*, 1994; Becker *et al.*, 1996; Gilmour *et al.*, 1996, Li *et al.*, 1996, Pritchard *et al.*, 1996; Costa and Dreher, 1997; Kodavanti *et al.*, 1997; Dreher *et al.*, 1997). These findings support the view that ambient PM_{10} has free oxygen radical activity causing lung inflammation and tissue injury. Also humic-like (organic) substances of a quinone-/semi-quinone-type molecular structure have been suggested to be the similar causal factor for diesel and other carbonaceous particles as well as for ambient PM (Sagai *et al.*, 1993; Ichinose *et al.*, 1995; Kumagai *et al.*, 1995; Ghio *et al.*, 1996). All these organic and metallic components are able to participate in electron transfer and redox cycling reactions resulting in free radical generation ($O_2.^-$, OH., lipid peroxides). Solubility or ionisability of these components seems to be a prerequisite for oxidative reactivity. This has been demonstrated by

extracting or chelating PM samples. Due to the loss of soluble transition metals, PM samples lose their biological activity.

Several papers have described the potency of aqueous extracts of total suspended particulates (TSP) collected prior to, during and immediately after the closure of a steel mill during a strike in Utah Valley in the winter of 1987 (Soukup *et al.*, 2000; Frampton *et al.*, 1999; Dye *et al.*, 2001). All of the studies indicate that the dust collected during the closure was much less active in biological test systems compared with the other two years. *In vitro* studies showed reduced activity in the host defence system and an increase in inflammatory mediators (Soukup *et al.*, 2000; Frampton *et al.*, 1999), whereas *in vivo* studies indicated increased but reversible airway reactivity and inflammation (Dye *et al.*, 2001). The PM samples differed in their metal contents, with yr 1 = yr 3 > yr 2, and metals correlated well with the sulphate contents. The water soluble metal content was less than 1% of the total mass. Nonetheless, the health effects were associated with the metal contents. Indeed, adding chelating resins to the filter extracts usually reduced the toxic effects, suggesting a prominent role for transition metals. However, Soukup *et al.* (2000) also observed that even on removing the metals, the effect of the PM extracts on phagocytosis of alveolar macrophages differed among the three years. This indicates that the metals are most probably not the only active ingredients within the complex PM mixture. The PM situation in Utah Valley during wintertime inversions is not comparable with that in the Netherlands. In Utah Valley, approximately 50–70% of PM_{10} emissions were from the steel mill. In the winter of the closure, PM concentrations were reduced by half (Frampton *et al.*, 1999). That one industrial source has such a large influence on PM levels is not seen in the Netherlands.

A number of *in vitro* studies using PM_{10} fractions sampled from ambient air also show that residual oil fly ash (ROFA), TSP, PM_{10} and $PM_{2.5}$ have soluble transition metal- or endotoxin-mediated free radical activity, causing oxidative stress, inflammation, and immunotoxic responses (Donaldson *et al.*, 1996, 1997; Becker *et al.*, 1996; Gilmour *et al.*, 1996; Samet *et al.*, 1997; Vincent *et al.*, 1997; Becker and Soukup, 1998; Bonner *et al.*, 1998). Recently, ROFA-induced cardiovascular arrhythmias have also been linked with rats suffering from myocardial infarction, but not with carbon black, suggesting that the chemical constituents play a major part in inducing these health effects (Wellenius *et al.*, 2002). Metal solubility may play a dominant role, and PM fractions < 2.5 μm show greater solubility and metal release than PM fractions > 2.5 μm (Smith *et al.*, 1998). Recently, however, Mohn and Becker (1999) presented *in vitro* data suggestive of coarse ambient PM being more potent than fine PM in provoking inflammatory responses, in which endotoxin may play an important role. Also, *in vitro* data from Hornberg *et al.* (1998a,b) on the genotoxicity of ambient fine and coarse PM collected from an urban area characterised by high traffic density suggests that coarse PM may have comparable or even higher activity than fine PM. The data indicate that biological effects of coarse fraction PM cannot be excluded yet.

Transition metals have been linked with various effects on the immune response (Gavett *et al.*, 1999; Gavett and Koren, 2001). On the other hand, metal salts (cadmium sulphate, lead nitrate, chromium nitrate and nickel sulphate) were found to exert no effect on NK cell function in the human concentration range (Yucesoy *et al.*, 1999).

4.3.2.4. PM in combination with gases

The US-EPA criteria document (2002) evaluates a variety of studies in which toxicity experiments with 'other PM' constituents were tested. Usually, concentrations are in the order of 1000s of $\mu g/m^3$. Almost none of these studies indicates any toxicant potency which makes it relevant for ambient exposure. There are only a few studies that might indicate exceptions.

Real-world exposures to air pollutants are rarely to single pollutants, but rather are a mix reflecting the combination of many sources, emission constituents, and/or ongoing photochemical processes in the atmosphere. Since the early 1950s, Amdur and colleagues have examined the potential for synergistic interactions of pollutant gases and PM. Her sentinel studies demonstrated that fossil fuel irritants such as acidic SO_2 could interact physicochemically with soluble metal salts to yield particles inherently more toxic than the original (Amdur, 1989). Later, using the combustion-related ultrafine particulate ZnO, SO_2 (20 $\mu g/m^3$) appeared to react with the ZnO in a partially humidified atmosphere to yield acid sulphate that carried deep into the lungs of experimental animals (guinea pigs). Other experimental studies have supported the potential of gas-particle interactions – some involving fine carbon or diesel, acidic, or dispersed ambient particles combined with organic/inorganic gases or vapours (e.g. O_3, NO_2, SO_2, HNO_3, aldehydes) (Last *et al.*, 1986; Warren and Last., 1987; Last and Warren, 1987; Jakab, 1992; Jakab *et al.,* 1993; Jakab and Hemenway, 1994; Creutzenberg *et al.*, 1995; Vincent *et al.*, 1997; Bolarin *et al.*, 1997; Madden *et al.*, 2000). The results of these studies suggest that particles can act as reactants and/or carriers to deliver toxicants to the deep lung. A limited number of human studies has been conducted along the line of Amdur's SO_2 interaction experiments and with acids and oxidants, but little evidence of synergism between PM and gases has been observed, perhaps due to the small numbers of subjects tested in most studies.

These interaction studies have been conducted at relatively high PM mass concentrations and, at present, there is no scientific indication for this type of interaction at environmentally relevant concentrations of gases and PM. Recently, it has been demonstrated that in a 4-week inhalation study of 0.5 μm particles combined with exposures to carbon black (50–100 $\mu g/m^3$), ammonium bisulphate (70 $\mu g/m^3$), and ozone (0.15 ppm or 300 $\mu g/m^3$) showed more deleterious effects than exposure to the components alone (Kleinman *et al.*, 1996; Bolarin *et al.*, 1997). The effects included decreased alveolar macrophage function, increased lung collagen concentration and lung cell turnover rates, although there were no indications for increased lung permeability and inflammation. An *in vitro* study that focused on pre-exposure to ozone and subsequent exposure to mineral particles revealed that ozone directly stimulates the uptake of particles in airway cells, which suggests that impaired particle clearance from the lungs and greater opportunity to interact with inter- and intracellular environments may be detrimental (Churg *et al.*, 1996).

Although the evidence is limited, these collective data suggest that the complex mixtures comprising air pollution atmospheres (PM plus various oxidant and photochemical product gases) may ultimately be responsible for PM-associated health

effects. Overall, however, no single compound or chemical class appears as yet to be a plausible candidate for the generic causative fraction.

4.4 Hypotheses on mechanisms of PM

The chemical, physical and biological composition of PM is complex, differing not only from place to place, but also from time to time. An understanding of what in this complex mixture determines its toxicity would aid in PM monitoring and control. Studies of the mechanisms of PM toxicity may contribute to the identification of these components. The mechanisms discussed here relate only to the acute effects of PM, because only a few studies have been conducted of chronic PM exposures while approximately one hundred time-series studies have currently been performed. As such, the mechanisms for lung cancer and other diseases related to chronic exposure will not be discussed here.

4.4.1. Respiratory tract pathway

The first few generations of the respiratory tract as well as the uppermost part (nose, trachea) are also covered with ciliated cells. This cell type can remove particles efficiently and rapidly. The majority will be swallowed, but it has been suggested that, depending on the diameter, the particles can still cross this defence barrier and reach the underlying tissue. Also, nothing is known as yet about the fate and effect of swallowed PM. A systemic effect cannot be excluded.

Upon inhalation, particles make contact with lung epithelial lining fluid (ELF) and may interact chemically with its components. ELF is composed of surfactant lipids, proteins and antioxidants (such as ascorbic acid (AA), glutathione (GSH) and uric acid (UA)) and serves as the first line of defence against inhaled toxins and infective agents (Sun *et al.*, 2001). In a next step, two key lung cell types, the macrophages and bronchial epithelial cells, are targeted by PM and inflammation is induced as a protective response. Reactive oxygen species (ROS) induced in these cells activate the promoters of cytokines and chemokines involved in inflammation through activator protein-1 (AP-1) and nuclear factor-kappa B (NF-κB) signalling pathways. The release of these inflammatory mediators induces the migration of granulocytes and lymphocytes into the tissue along with the local activation of these cells. The inflammatory reaction caused by the inhalation of particles may lead to the aggravation of existing lung disease and enhance the sensitivity to allergens of people with hay fever and asthma. It may also have the capacity to alter blood coagulability and circulating red cells and platelets, which can lead to myocardial infarction (Seaton *et al.*, 1995). This mechanism could explain the adverse influence of inhaled particles on cardiovascular morbidity and mortality. When this crucial and normally beneficial inflammatory response occurs in an uncontrolled manner, the result is excessive cellular/tissue damage leading to chronic inflammation and the destruction of normal tissue.

4.4.2. Cardiovascular pathway

Associations between PM and heart failure assume that, besides the pulmonary system, air pollution might also affect the cardiovascular system. Whether this is an indirect effect caused by messengers initially produced in the lungs or because the particles themselves actually reach the bloodstream remains to be established. An increasing body of evidence suggests that the effects of PM need not necessarily be restricted to or always have to be found in the lungs. As mentioned in previous sections, PM can cause systemic effects, although at present the applied dose appears rather high, with the exception of the human exposure studies.

As mentioned briefly, the ROS, as well as reactive nitrogen species (RNS), can cause damage to the surrounding tissue epithelium and endothelium. Damage to endothelium results in an imbalance between the proteins and peptides that affect blood pressure, like endothelins, angiotensin converting enzyme (ACE) and endothelial cell NO synthase, and the activation of the coagulation cascade. A critical parameter might be the ratio between endothelins and NO. The impact of the particles could be a decreased secretion of NO to the smooth muscle cells and increased secretion of ET-1 from the endothelial cells, with spillover in the systemic circulation. The higher vasoconstrictor is coupled with lower vasodilator, and this could be a malicious combination with respect to haemodynamics. Endothelial dysfunction has several implications. It can lead to vascular inflammation, in which cytokines are induced. Cytokines like IL-6, IL-8 and TNF-α have been linked to coronary heart disease and the progression to heart failure (Shan *et al.*, 1997). TNF-α also induces cardiomyocyte apoptosis (Song *et al.*, 2000). Endothelial dysfunction can also lead to plaque disruption in patients with atherosclerosis (Kinlay and Ganz, 2000), which in turn may cause myocardial infarction. During the last few years it has become clear that cardiac dysfunction and heart failure are not just a problem of the heart itself, but that peripheral organ dysfunction has an additional effect on cardiac function. Both the role of systemic (chronic) inflammation and endothelial dysfunction has been described in the development of heart failure (Kleber and Petersen, 1998). Elevated blood coagulation proteins like fibrinogen also increase blood viscosity. Circulating cytokines and neutrophils might have deleteriorous effects on the heart. Increased blood pressure, procoagulant state, tissue damage and elevated blood viscosity have a negative effect on cardiac function. A sensitive and early marker of cardiac dysfunction is ANF expression in the ventricles.

4.4.3. Mechanisms

The underlying biological mechanisms of the health effects associated with air pollution are most likely diverse and complex, and may not be a consequence of a single toxicological pathway. For example, it seems implausible that worsening of asthma in children could result from the same injury as the increased risk of cardiac death in elderly people. It also seems unlikely that a single toxic fraction responsible for all effects will be identifiable, although it is possible that a universal process, such as oxidative stress, may initiate different reactions in different individuals with different susceptibilities.

Several hypotheses have been proposed to explain the epidemiological findings of PM-associated health effects. Besides the fact that toxicity is assigned to a specific chemical part of PM, rather than the total mass, two leading hypotheses regarding the

mechanism of PM are known: firstly, that of modulation via sensory neural pathways (stimulation of receptors increases) and secondly, that of oxidative stress (reactive oxygen species (ROS) increases) (see Figure 4.3)

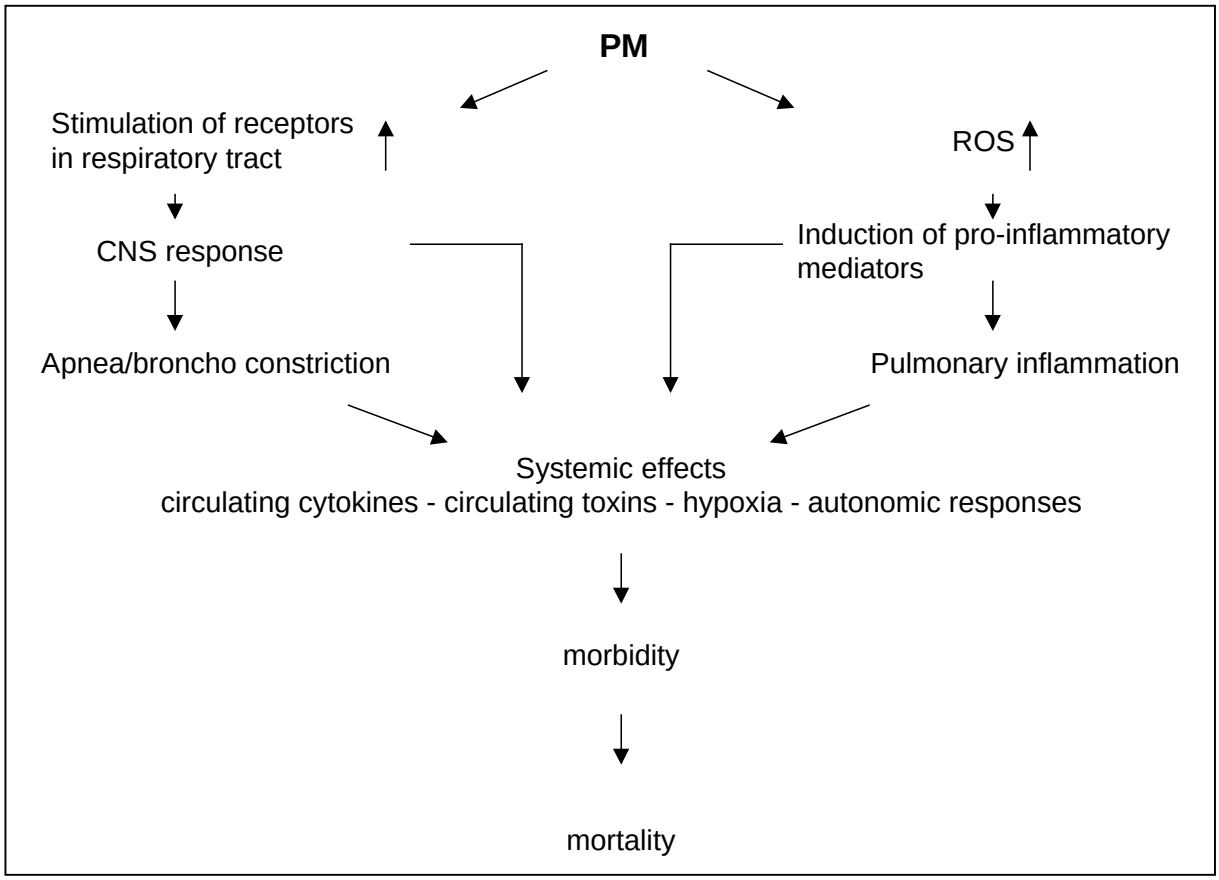


Figure 4.3 A schematic impression of possible mechanisms underlying PM-induced effects related to morbidity and mortality. On the left, the neurogenic inflammation, on the right, that of oxidative stress. After Godleski et al., 2000.

The suggested mechanism acting via modulation of sensory neural pathways includes effects like autonomic neurological responses, myocardial hypoxia, systemically circulating inflammatory cytokines and toxins. However, the leading hypotheses on the mechanism of PM toxicity focus on the induction of oxidative stress via specific PM properties either directly or through the induction of inflammation. Reactive oxygen intermediates generated by immune cells recruited to the sites of inflammation are a major cause of cell damage. A situation of this kind is characteristic of oxidative stress. It is also hypothesised that PM can modulate the immunosystem. For example, diesel exhaust particles (DEP) are able to induce aspecific inflammatory responses in the lung, which together with the allergen present result in stronger immune response (Diaz-Sanchez *et al.*, 1997). Other studies have indicated that allergens, such as free grass pollen allergen molecules, can form agglomerates with fine particles present in air. In this way they are able to deposit deep in the lung, resulting in a more effective antigen presentation and immune response (Knox *et al.*, 1997).

4.4.3.1. Neurogenic inflammation

It is hypothesised that one way in which PM initiates airway inflammation is through the neurological pathway, or neurogenic inflammation (NGI). The activation of specifically capsaicin-sensitive vanilloid (e.g. VR1) irritant receptors appear to play a part in this process (Veronesi and Oortgiesen, 2001). These acid-sensitive receptors are located on the sensory C nerve fibres that innervate the airways, as well as on various immune and non-immune airway target cells. The activation of these receptors results in the release of neuropeptides from the sensory terminals that innervate the airways. Their interactions with airway target cells result in signs of inflammation (e.g. bronchoconstriction, vasodilatation, histamine release, mucous secretion etc.) (Veronesi and Oortgiesen, 2001). Recently, differential increases in $[Ca^{2+}]_i$ and IL-6 release have been described to occur in human bronchial epithelial cells (BEAS-2B) and dorsal root ganglia (DRG) sensory neurones when exposed to PM derived from different sources (Veronesi *et al.*, 2002b).

PM induced autonomic neurological responses may lead to direct effects on heart cells, with subsequent changes in the cardiac electrical systems, including arrhythmia/bradyarrythmia, low frequency/high frequency ratio increases, heart rate variability increases, impaired atrioventricular conduction, fatal ventricular fibrillation (with T-wave flipping as a possible indicator), and ultimately death. The systemic effects may be caused by receptor stimulation, CNS responses, apnoea, bronchoconstriction. However, the evidence for this cardiotoxic mechanism is extremely limited and is only suggested by a few epidemiological studies (Dockery *et al.*, 1999) and *in vivo* inhalation or instillation studies on experimental animals (Gordon *et al.*, 1998). The data in experimental studies show that effects are sometimes larger in animal models for cardiopulmonary diseases compared with healthy animals. Killingsworth *et al.* (1997) studied the effects following inhalation of fly ash in rats and found that chemokines (MIP-2) in heart tissue were released and that macrophages in the heart contain in fact true fly ash particles. Although studies of the neurological pathway are not comprehensive, a neuroimmunological mechanism in the initiation of PM-associated airway inflammation could be plausible, since sensory receptors are highly sensitive indicators of noxious and potentially damaging chemical exposure (Veronesi and Oortgiesen, 2001).

4.4.3.2. Oxidative stress

Oxidative stress most likely plays a major role in inducing the effects of air pollution on health. Oxidative stress is defined as a process in which the balance between oxidants and antioxidants is distorted due to an excess of oxidants. This shift can lead to antioxidant depletion. Consequently, the oxidants can induce biological damage. Such a state of oxidative stress is thought to contribute to the pathogenesis of a number of human diseases, including those of the lung. Among them are atherosclerosis, diabetes mellitus, chronic renal failure, rheumatoid arthritis and neurodegenerative disease. In many cases the investigation of parameters of oxidative stress and radical damage has brought substantial insights into their pathogenesis. It has been shown that oxidative stress can be mediated by, for instance, transition metals (Costa and Dreher, 1997; Jimenez *et al.*, 2000) or ultrafine particles of

unknown composition (Brown *et al.*, 2000). Either PM contains already reactive oxygen or nitrogen species or these are formed in the lung after provocation with particles with or without gases.

a. Reactive oxygen species (ROS)

Oxygen is often the source of radicals, as partially reduced species are generated through normal metabolic processes and some of these reactive species can escape (Kehrer, 2000). Continuous production of a small amount of free radicals and reactive oxygen derivatives is therefore a basic feature of aerobic life. Within certain boundaries, the generation of ROS is even essential to maintain homeostasis. For example, ROS generation by phagocytic cells constitutes an essential host defence mechanism necessary to combat infection. Likewise, cytosolic ROS produced in response to stimulation by growth factors are involved in regulating the proliferative response. Under a certain situation of metabolic stress, even mitochondrial-derived oxidants seem to function as signalling molecules. However, a more extreme rise in intracellular oxidant levels has two potentially important effects: damage to various cell components and triggering of the activation of specific signalling pathways (Finkel and Holbrook, 2000).

ROS can be derived from numerous sources *in vivo*. Among them are autooxidation and photochemical reactions. However, *in vivo* the most important sources are enzymatic-driven redox reactions involving oxygen. As a result, superoxide ($O_2^{\bullet-}$), being the major factor in oxygen toxicity (Fridovich, 1974), is generated *in vivo* as a product of the following metabolic processes.
First of all it is generated by the NADPH oxidase as part of the antibacterial function on phagocytic cells. Production of superoxide radicals is an important part of the bacterial killing mechanism of neutrophils. In this process, called phagocytosis, invading microorganisms are enclosed in phagosomes. During the respiratory burst the PMNs become activated, O_2 consumption increases and a membrane-associated NADPH oxidase reduces oxygen to superoxide anion ($O_2^{\bullet-}$) (Babior *et al.*, 1973), which subsequently dismutates to form H_2O_2 (Weening *et al.*, 1975). However, the expanding family of NADPH oxidases also include the nitric oxide synthase (NOS) family, illustrating the apparent purposeful and deliberate use of oxidant generation in normal cellular signalling and homeostasis. Another important metabolic process in which ROS are produced is the mitochondrial electron transport chain, where the synthesis of ATP is coupled to the reduction of O_2. Superoxide can be formed at Complex I by the donation of an electron to oxygen from NADH dehydrogenase. Superoxide and/or hydrogen peroxide may also be formed following the autooxidation of reduced ubiquinone at Complex II (succinate-coenzyme Q) and/or Complex III (coenzymeQH_2-cytochrome c reductases) sites. This makes the generation of ROS predominantly a function of metabolic rate.

Reactive species in biological systems comprise freely diffusible hydrogen peroxide (H_2O_2), organic hydroperoxides and hypohalous acids (HOX, X=Cl, Br, I) with half-lives in the range of minutes, the peroxyl radical and nitric oxide (NO) with half-lives of seconds, peroxynitrite (OONO-) with a half-life of a millisecond, superoxide anion ($O_2^{\bullet-}$), singlet oxygen (O_2) and alkoxyl radical with half-lives of about a microsecond, and the hydroxyl radical ($OH^{\bullet}$) with a half-life that is diffusion-limited at about a nanosecond. Having a short half-life, $O_2^{\bullet-}$, must react at the site at which it is generated or a short distance from its origin. In contrast, H_2O_2, is a mild oxidising agent, is relatively stable, moves easily through cell membranes and affects sites far from its origin. However, it is believed that H_2O_2 and $O_2^{\bullet-}$ produce their most potent effects by forming the hydroxyl radical ($OH^{\bullet-}$) via the Haber-Weiss reaction:

$$O_2^{\bullet-} + H_2O_2 \rightarrow OH^- + OH^{\bullet-} + O_2$$

In the following manner:

$$Fe^{2+} + H_2O_2 \rightarrow Fe^{3+} + OH^{\bullet-} + OH^- \quad \text{(Fenton reaction)}$$
$$Fe^{3+} + O_2^{\bullet-} \rightarrow Fe^{2+} + O_2$$

Transition metals such as iron and copper, being a part of PM, are capable of redox cycling and can induce Fenton reaction, resulting in the production of reactive oxygen

and nitrogen species (Jimenez *et al.*, 2000). It has been shown that equal masses of different PM samples can induce disparate lung injuries, suggesting that particle components may be relevant in assessing health effects after their exposure or that a specific susceptibility may be the cause of these differences in effects. In the case of the Utah Valley study, metals specifically have been related to the biologic effects of collected PM.

When cellular production of ROS overwhelms the protective antioxidant capacity, damage to molecules (and their accumulation) and disruption in cellular homeostasis may ensue. Free radicals, especially the $OH^{\bullet -}$ radical, which is a highly reactive species interacting with almost anything in its immediate vicinity, can react with macromolecules such as proteins and carbohydrates, polyunsaturated fatty acids and DNA.
Oxidation of proteins and carbohydrates may result in fragmentation and cross-linking, with subsequent loss of function. Protein repair systems do not appear to exist, since it is apparently more efficient either to prevent oxidation or simply to destroy the modified species.
Peroxidation of membrane lipids results in loss of integrity. During this process isoprostanes, which are considered to be a marker of oxidative stress and many disease states, are also produced. They may in fact mediate the effects of free radicals and reactive oxygen species (Janssen, 2000) and have the ability to affect signalling pathways, including those that regulate the apoptotic form of cell death (Kehrer, 2000).
Concerning damage to DNA, the DNA base guanine is particularly sensitive to oxidation making the detection of 8-hydroxy-deoxyguanosine a reasonable biomarker for oxidative injury.

b. Antioxidant mechanisms

The burden of ROS production is largely counteracted by a complex antioxidant defence system. An antioxidant is defined as any substance that, when present at low concentrations compared with those of an oxidisable substrate, significantly delays or prevents oxidation of that substrate (Halliwell, 1991).

Antioxidants can be classified by their method of action (enzymatic and non-enzymatic scavengers) or, for example, by their site of action (intra-cellular and extra-cellular). Intra-cellular scavengers include glutathione peroxidase (a Se dependent enzyme found in the cytosol and mitochondria), glutathione (present in millimolar concentrations within cells), catalase (located in the peroxisomes, where it reduces H_2O_2 to H_2O and O_2) and superoxide dismutase (SOD, which catalyses the dismutation of $O_2^{\bullet -}$ into H_2O_2; CuZn-SOD can be found mainly in the cytoplasm, Mn-SOD in the mitochondria, CuZn-SOD in the extracellular fluids). Extra-cellular antioxidants include vitamin E, Vitamin C, carotenoids, uric acid, plasma proteins such as metal chelators and albumin (Bunker, 1992).

It seems likely that especially in diseased persons, protective antioxidant mechanisms are reduced, leading to an imbalance between free radical production and removal, resulting in the accumulation of end products of free radical action and slow, progressive tissue damage (Bunker, 1992).

The lung has a well-developed antioxidant system, with reduced glutathione (GSH) being one of the key components (Rahman and MacNee, 2000a). GSH, being a vital

intra- and extracellular protective antioxidant, has been shown to be critical in protecting airspace epithelium from oxidative/free radical mediated injury and inflammation (Li *et al.*, 1994). It protects cells from the toxic effects of oxidants by maintaining the reduced state of the sulphhydryl groups of a number of sulphhydryl-dependent enzymes (Puglia and Powell, 1984). The induction of oxidative stress is evidenced by decreased levels of the antioxidant GSH in the lungs (Li *et al.*, 1996) and epithelial cells *in vitro*, whereas acute inhalation of oxidising agents such as ozone or NO_2 depletes the lungs of GSH (Sagai *et al.*, 1982). Depletion of GSH, affecting the GSH/GSSG ratio, influences a variety of cellular signalling processes, such as activation of the transcription factors activator protein-1 (AP-1) and nuclear factor-kappa B (NF-κB) (see Figure 4.4). It can also increase the permeability of lung

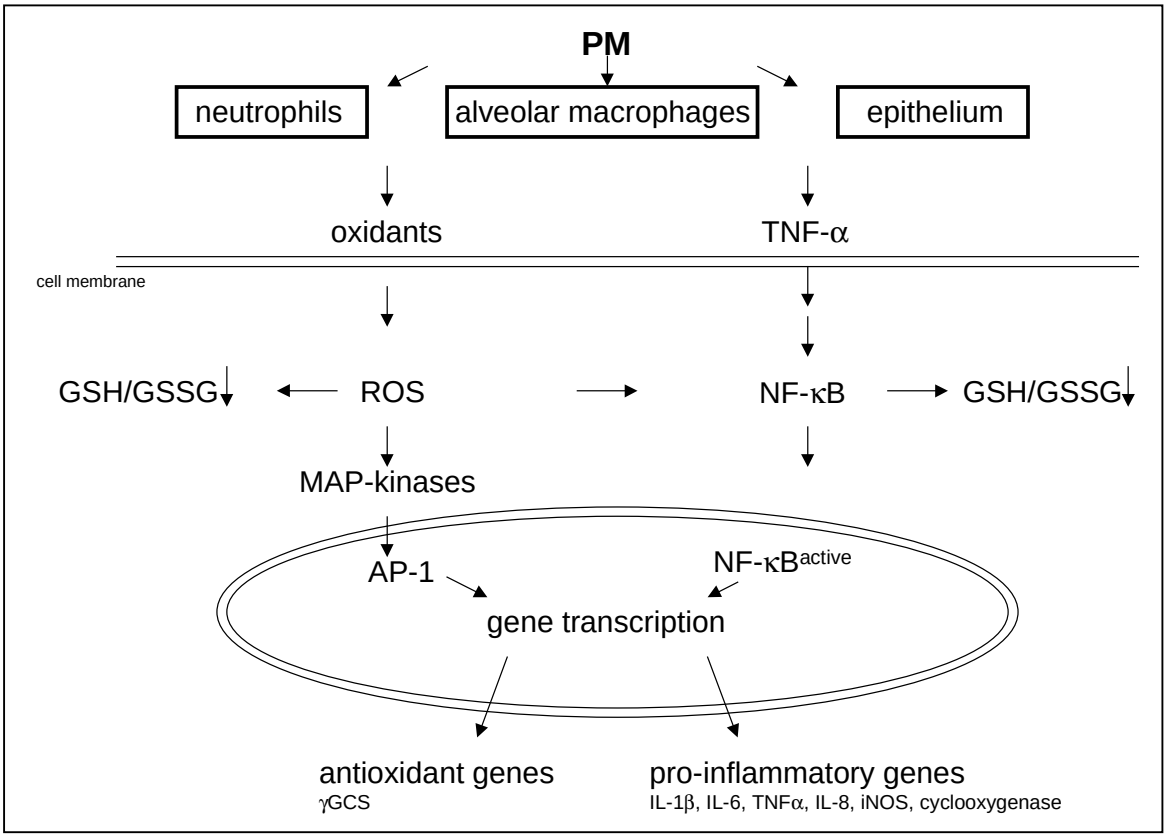


Figure 4.4 . Schematic overview of effects of PM on several signal transduction routes.

epithelium, which can lead to the passage of particles into the interstitium (Li *et al.*, 1994). Alterations in the alveolar and lung GSH metabolism are widely recognised as a central feature of many inflammatory lung diseases.

The intracellular GSH redox homeostasis is strictly regulated to govern cell metabolism and protect cells against oxidative stress (Rahman and MacNee, 2000b). The synthesis of glutathione requires two enzymes and three amino acids, glycine, cysteine and glutamic acid (with cysteine being the rate-limiting substrate). The tripeptide GSH is formed by the consecutive actions of γ-GCS and glutathione synthetase. Gamma-glutamylcysteine synthetase (γ -GCS) is the rate-limiting enzyme in GSH synthesis. γ-GCS consists of a catalytic heavy subunit (γ-GCS-HS) and a regulatory light subunit (γ-GCS-LS). The promoter regions of both corresponding genes contain putative AP-1 and ARE response elements necessary for expression in

response to diverse stimuli, such as oxidants, phenolic antioxidants, inflammatory, and anti-inflammatory agents in lung cells.

c. Signal transduction

Besides ROS being part of an oxidative stress situation, recent studies have also implicated ROS generated by specialised plasma membrane oxidase in *normal* physiological signalling by growth factors and cytokines (Thannickal and Fanburg, 2000). ROS are able to act directly on signal transduction pathways, or indirectly through the generation of bioactive mediators. They modulate quite a number of so-called redox sensitive transcription factors, such as AP-1 and NF-κB (Figure 4.10), while ROS are also able to affect the activity of calcium-signalling, antioxidant enzymes, ion transporters, various cell-growth-related genes and various kinases (c-jun N terminal kinase (JNK), extracellular regulated kinase (ERK) and the mitogen-activated protein kinase (MAP kinase) family) (Adler *et al.*, 1999; Sen, 1998). It seems that also controlling the thiol status of a cell is critical in altering the signalling pathway.

The development of an oxidant/antioxidant imbalance in lung inflammation may activate redox-sensitive transcription factors such as NF-κB and AP-1 (see Annex D for more details) (Gius *et al.*, 1999; Rahman and MacNee, 1998). Transcription factors are DNA-binding proteins that interact with specific sequence motifs in the promoter region of the gene, controlling the expression of these. In particular NF-κB, a multi-subunit transcription factor, plays a central role in the regulation of the expression of numerous genes involved in the inflammatory and immune response (resulting in upregulation of cytokines and chemokines) (Baeuerle and Henkel, 1994) as well as protective antioxidant genes, e.g. γ-GCS. The critical balance between the induction of proinflammatory mediators and antioxidant genes in response to oxidative stress at the site of inflammation is not known (Rahman, 1999). Cellular oxidative processes seem also to activate stress kinases (JNK, MAPK, p38). Exogenous and endogenous antioxidants are effective in blocking activation of NF-κB and preventing the consequences of proinflammatory gene expression. The most potent inducers of the NF-κB activation pathway on the other hand are TNF-α, IL-1β and gram-negative endotoxin or lipopolysaccharide (LPS). LPS can be found in the coarse fraction of PM, making it a plausible candidate for PM toxicity.

Via a different route, involving ROS, ultrafine particles are able to enhance Ca influx, inducing alterations in calcium signalling. This results in inflammatory responses via the activation of transcription factors, e.g. NF-κB (Dolmetsch *et al.*, 1998). Calcium oscillations increase the efficiency and specificity of gene expression. For example, oscillations reduce the effective Ca^{2+} threshold for activating transcription factors. Whereas rapid oscillations stimulate all three transcription factors, infrequent oscillations activate only NF-κB and not NF-At and Oct/OAP (Dolmetsch *et al.*, 1998).

4.4.4. Susceptibility

Associations of air pollution and health effects (hospital admissions and mortality) are found worldwide, regardless of the source and composition of PM, and of the local climate. This association appears to be independent of the population size and density. Mortality mainly involves elderly persons with one or more chronic diseases, usually of a cardiopulmonary nature. The population at risk therefore implies one of weakened health, caused by ageing or illness. However, the association, although consistent, is small. Therefore the degree of stress induced by ambient PM is low or the susceptible population is small. It is suggested that this population consists mainly of individuals with severe, generalised homeostatic instability (Frank and Tankersley, 2002).

It is this homeostatic instability rather than the type or level of external stress, which promotes the fatal outcome. Homeostasis is defined as the organism's capacity to withstand stress and maintain a stable, relatively constant internal environment. It is most robust in young adulthood and declines with ageing and illness. As this decline in homeostasis from ageing or illness proceeds, the risk of dying increases. Studies have suggested that certain individuals may be at higher risk to adverse effects of PM (Dockery *et al.*, 1993; Schwartz, 1994). It is not surprising that the elderly with chronic cardiopulmonary diseases, those with pneumonias and asthmatics at any age (Neas *et al.*, 1995) appear to be at greater risk.

The increasing instability in homeostasis with ageing is evidenced by deleterious changes to the immune system, which are associated with a decreased ability to effectively handle oxidative stress. That elevated levels of cellular oxidative stress are present in aged experimental animals is indicated by elevation in tissue and circulating lipid peroxide levels as well as by oxidised proteins (Poynter and Daynes, 1998). It has been suggested that the imbalance and genetic variability of γ-GCS and proinflammatory gene expression in response to oxidative stress and inflammatory response may be a determinant of susceptibility to lung disease (Rahman and MacNee, 2000a).

4.4.5. Summary on mechanisms

Ambient PM toxicity studies have been intensified in recent years. The current limited data, however, have not yet resulted in sufficient evidence to convincingly indicate one or several mechanisms explaining PM health effects at ambient levels in susceptible people considered to be at increased risk. Most of the data indicate that PM fractions are able to induce inflammation and immunotoxicity in airways and lungs via oxidative stress or via a neurological mechanism by impairing respiratory and cardiac neurological functions. Rather than assigning the association between PM and mortality to the toxicity of PM itself, a more plausible explanation is that this association is the result of the organism's reduced capacity to withstand stress and maintain a stable, relatively constant internal environment. It is therefore plausible that the population at risk is largely defined in particular by receptive individuals with failing health, attributable to ageing or illness.

4.5. Summary and conclusions

Dosimetry indicates that the delivered PM dose is not only dependent on the ambient or exposure concentration but also on critical factors such as age, activity pattern and respiratory diseases affecting the morphology. This argues against the linear relationship approach as used in epidemiological studies. For example, coarse particle tracheobronchial and thoracic deposition fractions are significantly larger for children (aged 0–15 years) than for adults. Deposited aerosol mass rates and exposure doses in the tracheobronchial region per unit surface area decrease with progressing age. Deposited pulmonary mass rates and exposure doses of ultrafine and 2.5 µm particles per alveolus are lower for the age range of 2–3 years compared with adults and higher for children aged 8–14 years old compared with adults for 5 µm particles. The pulmonary exposure doses per unit surface area increase almost linearly between the ages of 20 and 80 years by approximately 30%. Deposited dose and mass rate increase with increasing physical exertion. This is partly due to switching from a nasal to an oral breathing pattern.

Tracheobronchial and thoracic deposition fractions are larger for humans than for rats at the same exposure level. Coefficients for the animal-human extrapolation of exposure dose need to be taken into account when extrapolating results of animal exposure studies to a human equivalent.

Results obtained in a variety of human and animal studies, as well as in *in vitro* studies using cell cultures, has improved our understanding of the plausibility of the epidemiological evidence for the pulmonary and cardiovascular health effects of particles. From a toxicological perspective, a causal relationship between PM mass concentrations and the various health effects at current ambient levels is very unlikely. This is illustrated by studies that have used highly toxic particles (polytetrafluoroethylene fumes); adverse health effects were found after exposures in the tens of $\mu g/m^3$ range.

Preliminary data on animal and human exposures to Concentrated Ambient Particles (CAPs) seem to indicate that minimal changes can be provoked in people with a distorted or weaker homeostasis. Health endpoints include HRV, HR, BP, nasal lavage, sputum and blood inflammatory, immunological and coagulation factors, as well as blood pressure mediators (endothelins, NO). Even then, the effects appear to be mild and reversible within a short period after exposure. The CAPs studies (either in humans or experimental animals) met with difficulties because of constantly changing (and thus uncontrollable) ambient PM levels and composition. This requires substantial replication of these studies and careful statistical analysis, relating the large number of exposure parameters with the health endpoints. This is one of the major reasons why, as yet, very little work on this topic has been published in peer reviewed literature. Although a systematic review of the CAPs studies has not yet been performed, signs of PM being able to cause adverse effects in both healthy and compromised animals and humans are emerging. The available evidence from studies of CAPs exposure in either animal or human subjects implies that PM mass is not evidently associated with adverse health effects. This suggests that other metrics may be more appropriate, e.g. chemical composition or physical properties, although

it has been suggested that a number of transition metals (V, Ni) contribute to the causal relationship between mass and health outcomes in regions where epidemiological and toxicological studies have been carried out (Utah Valley). A unique series of studies with Utah Valley PM have shown consistencies between epidemiology and toxicology. However, the critical issue persists – in toxicological studies no effects of exposure to PM at concentrations close to ambient levels have been found which can unequivocally be linked to serious health effects as observed in epidemiological studies. Even at exposure concentrations far above the average levels in the western hemisphere, health effects have not consistently been found that are likely to be caused by ambient PM mass levels.

Health effects may occur in particular at relatively high acidity, for instance in combination with acidic sulphates. In the Netherlands, annual average particle-associated metal concentrations are close to or far below 1 $\mu g/m^3$ (Visser *et al.*, 2001). At such levels it is unlikely that these metals are the causal fraction of ambient particles. In the Netherlands, the above-mentioned transition metals (V, Ni) are only present in the ng/m^3 range, with the exception of iron. Annual average concentrations of Fe_2O_3 in the Netherlands are in the range of 0.2–1.1 $\mu g/m^3$. As Fe(III) has a toxic potency for disturbing the redox environment of humans and animals, the possibility of this metal playing a role in human health impacts on the Dutch population cannot be ruled out. This is further supported by recent studies in Germany, in which both iron and nickel have been identified as key players in the production of Reactive Oxygen Species (ROS) in epithelial cells (Borm *et al.*, 2002). Based on the evaluation of pulmonary toxicity data and the redox potential of iron, it is also possible to speculate that this metal may be important for individuals, in particular those highly susceptible to oxidative stress. For healthy individuals, though, no toxicity can be expected. Susceptible groups include old people, persons with poor nutrient intake – in particular low protein intake, persons with (non-manifested) haemachromatosis.

The low acidity of Dutch ambient PM combined with the low pulmonary and cardiovascular toxicological potency of acidic aerosols leads to the conclusion that acidity is probably not the causal factor for PM-associated health effects in the Netherlands.

Apart from the clear associations between PM mass and the health effects in population studies, epdemiological studies also used sulphate as a proxy to correlate with the effects. Large fractions of Dutch ambient particles are ammonium nitrate and sulphate, together with sodium chloride. No toxicity has been identified for the pure components of these substances in rodent studies after exposure at an order of magnitude higher than ambient levels in the Netherlands. These components are soluble in water and normal concentrations in body fluids are at least an order of magnitude higher than those that can be reached from the absorbed dose by way of inhalation. This makes it highly unlikely that pure sodium chloride (sea salt), ammonium nitrate or ammonium sulphate (SIA) are the causal fractions of the PM-associated health effects in the Netherlands. However, there are some caveats in this type of evaluation. For example, it may be important to consider possible interactions between sea salt and SIA and ambient gases in developing conclusions as to

biologically effective levels of the former. Very little information is available to investigate this possibly joined action, nor is there sufficient information on the atmospheric chemistry to determine what are the actual concentrations of the possibly formed constituents to which the respiratory tract as the prime target organ may be exposed. Based on current toxicological evidence, a standard or limit value for sulphate would be higher than the current PM value on a mass base. Epidemiology and toxicology seem to make paradoxical statements with respect to sulphates and this issue needs further attention. Obviously, using sulphate as an indicator does not imply that sulphate itself will be the causal factor, but it is representative for a part of PM that induces the effects and is as yet unidentified. Unfortunately this has led to much confusion. In this context, the role of SO_2 as the precursor gas for sulphates should be considered as well.

Some studies have pointed out that traffic exhaust particles are able to enhance respiratory allergy, in particular rhinitis, but the relationship between more serious symptoms such as heart failure or sudden death have not yet been made.

Dutch endotoxin concentrations in ambient air are in the range of ng/m^3, whereas the human toxicity is manifested in the range of $\mu g/m^3$ exposures. Hence, it is unlikely that endotoxin is the causative fraction of the PM-associated health effects in the Netherlands.

A number of other factors that may relate to the causality have been identified (charge, radiation, size, number and surface area) but at present insufficient data are available to judge their role in the PM-associated health effects.

5. Standards for PM

5.1. Previous standards for PM

In the 1987 criteria document on fine PM in the Netherlands (Van der Meulen *et al.*, 1987) two standards for ambient PM_{10} were presented. As an annual average standard a level of 50–65 $\mu g/m^3$ of PM_{10} was proposed and the same criteria document suggested a daily average standard of 100–150 $\mu g/m^3$. At that time it was presumed that both standards would provide protection against lung function decrements and other health effects, though probably not with a margin of safety as they were based on effect thresholds found in epidemiological studies.

The actual standards the Dutch government eventually promulgated were 40 $\mu g/m^3$ as an annual average and 140 $\mu g/m^3$ as a daily average for PM_{10}. Both of the Dutch standards were slightly below the then prevailing PM_{10} standards in the USA of 50 $\mu g/m^3$ as an annual average and 150 $\mu g/m^3$ as a daily average.

These previous Dutch PM_{10} standards were never intended to offer adequate protection to persons who are allergic to some or all of the ambient PM or pollen, or to offer adequate protection against parts of the PM that might be carcinogenic or toxic, e.g. PAH, diesel soot and heavy metals. Specific carcinogenic substances and metals are covered by composition-specific standards in the Netherlands and other EU countries and not by the general PM standards. Germany, for instance, has a specific standard for diesel soot. In the Netherlands, pollen is not seen as a problem that can be regulated by the Ministry responsible for the environment.

Following publication of the results of the first epidemiological studies described in Section 3 in the early 1990s, it became clear that ambient PM concentrations below these previous standards may still be associated with mortality and hospital admissions, emergency or otherwise. These serious health effects were taken as a basis for all the research activities relating to the new standards for PM over the last decade. The other health effects mentioned were still considered to be covered as far as appropriate by the previous standards.

There are a number of time-series studies demonstrating an association of PM_{10} with mortality and morbidity in the Netherlands. The magnitude and the seriousness of these health effects as described in Section 3 is such that standard setting is warranted. In subsections 5.2 and 5.3 these time-series studies and their results are used to argue the choice of a particle size and an averaging time that might be appropriate for the mix of pollution and the susceptibility of our population.

5.2. Particle sizes for a PM standard

The present health limit in the Netherlands is based on PM_{10}, and Dutch air quality has been measured as PM_{10} in the National Air Quality Monitoring Network since 1992. No annual measurement series are currently available in the Netherlands for $PM_{2.5}$. This lack of data complicates a substantiated $PM_{2.5}$ standard for the Netherlands. PM_{10} and $PM_{2.5}$ are ‘containers’ of PM for a daily varying mix of different substances. Based on the Hill criteria (1965), epidemiological causality can

be inferred for PM or air pollution in general. The specific causal fractions for health effects and their sources still have to be identified. Both PM_{10} and $PM_{2.5}$ probably include material that does not contribute much to their toxicity, and this quantity of less toxic material and/or water 'contaminates' the current mass-based limits. This diminishes the effectiveness of health risk reduction by a crude reduction of the aerosol mass. On the other hand, this quantity of less toxic material leads to the conclusion that the remainder of the PM probably contains the causal factors and therefore becomes more important for risk management purposes. Knowing the sources of these causal factors would make an effective abatement strategy more feasible in terms of both cost and result.

The coarse fraction of PM_{10} (PM with diameters between 2.5 and 10 µm) in the Netherlands has some toxicity in *in vitro* systems, so it does seem prudent to keep this fraction of PM covered in a health-based limit for PM. If at some time in the future a totally different fraction or fractions of PM were to be discerned as more definitive culprit(s) and if the coarse fraction is cleared on the basis of valid and replicated scientific results, only then might it be discarded from this future limit. This is definitely not the case at present. It should be noted that non-anthropogenic PM is not concentrated solely in the coarse fraction with a cut-off of 2.5 µm. The majority of crustal material is indeed coarse, but sea salt is approximately evenly distributed over the fine and coarse fractions, while 80% of OC – part of which is non-anthropogenic – is in the fine mode. On the other hand, a quite considerable anthropogenic contribution of PM re-suspended by traffic and from tyre and road wear is predominantly found in the coarse fraction. Currently, there appears to be a tendency in the USA towards promulgating a separate standard for the coarse fraction of PM. It would be a gross oversimplification to state that the coarse fraction with a cut-off of 2.5 µm is non-anthropogenic and that the fine fraction is anthropogenic. The reality is more complex.

Losses of volatile materials mean that current automatic measurement values for $PM_{2.5}$ concentrations have relatively larger uncertainties than the automatic measurements of PM_{10}. More accurate measuring methods for $PM_{2.5}$ have not yet been validated for routine measurements. Because the losses of PM in the Netherlands are mainly associated with ammonium nitrate and semi-volatile organic matter and as both of these components are predominantly in the fine mode (with diameters < 2.5 µm), it may well be conjectured that the default loss factor of 1.3 for PM_{10} (Williams and Bruckmann, 2001) would be even larger for $PM_{2.5}$ if this size range were used as a PM metric.

Recent new scientific evidence indicates that other PM fractions much smaller than 2.5 µm in diameter (ultrafine (UF) particles with diameters of less than 0.1 µm) also appear to be associated with health effects, indicating that in the future other PM metrics and limits supplementary to PM_{10} or $PM_{2.5}$ may be appropriate. Before deciding on such a limit, though, more research on the epidemiology, toxicology and air quality of ultrafine PM is indicated. It is clear that simply switching from a PM_{10} limit to a smaller particle metric, e.g. $PM_{2.5}$, does not address the UF. In the Netherlands, UF are not well correlated with PM_{10} or $PM_{2.5}$, so abatement of PM in general will probably not be very effective for UF. Future research for a new standard should in any case include UF as well.

Choosing a PM metric smaller than PM_{10} as a standard would give rise to the question of whether this should be $PM_{2.5}$ or some other PM metric, e.g. PM_4 or PM_1. A number of arguments can be put forward for either of these alternatives.

On the basis of PM dose considerations it could be argued that a different (larger) metric like PM_4 would be preferable to $PM_{2.5}$. PM_4 is a metric that describes the respirable fraction of PM (ACGIH, 2001). If the main concern for human health effects were the total mass of PM deposited in the gas-exchange region of the lungs, a metric like PM_4 would be a sharper indicator of the deposited dose than a currently used metric like $PM_{2.5}$. ACGIH (2001) indicates that the most significant difference from previous definitions of the respirable fraction is that the median cut-point of a respirable particulate sampler has been increased from 3.5 µm to 4 µm in accordance with the International Organisation for Standardisation/European Standardisation Committee (ISO/CEN) protocol.

A completely different problem is that of the crustal contribution to current PM_{10} levels. This phenomenon peaks in southern Europe. The process of turbulent transport through the atmosphere in itself means that even some of the originally coarse crustal PM becomes fine and can be transported over considerable distances. A cut-off point at 2.5 µm does not really discriminate between these (foreign) aeolian particles and other parts of PM, in this case the anthropogenic part of the respirable PM fraction. If a differentiation of this kind between 'coarse' and 'fine' PM were necessary, because authorities wanted two separate standards for different sizes, a cut-off at 1µm would make more sense than one of 2.5 µm. Wind-blown crustal material generally does not end up in sizes below 1 µm in diameter. However, a certainly non-negligible part of this aeolian PM is in the respirable size range between 1 and 4 µm and consequently contaminates a standard with a cut-off of 2.5 µm. A recent study by Kegler *et al.* (2001) for the arid area of Arizona in the US confirms this and they conclude that the Phoenix data suggest that measurements for PM_1 may provide a more useful exposure surrogate than measurements of $PM_{2.5}$.

Nevertheless, there is a general feeling that other measures of the smaller size ranges or possibly for specific source categories of PM might also be useful in supplementing or replacing PM_{10} in the future. At the moment, however, it is still too early to decide whether this should be a measure like $PM_{2.5}$ or PM_1. There are a number of arguments in favour of $PM_{2.5}$, the most important being that there is currently much more information available (mostly from the US though) on $PM_{2.5}$ than on PM_1. However, arguments of practical applicability speak more in favour of putting the future cut-off at 1 µm, because then the crustal contribution and the contribution of sea-salt particles largely disappears from the sampled PM. For southern European countries affected by Saharan dust, this would be an argument to consider. Sea salt emerges as a major contributor to average Dutch PM_{10} levels. Using a cut-off of 1 µm would prevent sampling of the majority of the sea salt mass and contaminate compliance measurements. Before a new standard is promulgated, it should therefore be ensured that reliable measurement methods are available for this fraction which can be used for routine measurements in order to avoid problems similar to those we now have with our measurements of PM_{10} and the assessment of compliance with the current standards.

5.3. Averaging periods for a PM standard

The averaging periods for a PM standard should be relevant to the characterisation of the population health risk. This statement may seem obvious, but will prove essential later for a substantiated choice of averaging period.

First of all, it has to be established whether it is essential for reasons of health risk management to have two different averaging times: one with a daily and one with an annual period. If this is not deemed essential, the second question of whether multiple averaging times would be convenient for policy reasons or for reasons of risk management arises. As to the first question, 'Are two or more averaging times essential for health risk management?', the short answer is: no, probably not. (This point will be elaborated below.) The previous position paper by Seifert *et al.* (1997) indicated that a daily average and an annual average standard can both be used to reduce health risks to a similar level in a population and that both standards are chosen to be mutually balanced and to a certain extent interchangeable. There is currently no new scientific information that challenges this 1997 position. So, similarly to the choice of cut-off size for PM, either both averaging periods or one of the two would suffice. A number of arguments will be presented, partly of a more practical nature, that ultimately come down more in favour of the longer of the two averaging periods. The choice, however, is up to the policymakers.

In the position paper (Seifert *et al.*, 1997) it is stated that the annual limit and daily limit 'should be balanced, i.e. the relation between those two limit values should reflect a "normal European situation" in the sense that control levels are more or less equally necessary to meet both limit values'. In the Netherlands this is not the case for the 2005 limit values: the EU daily standards are exceeded, while at the same time the EU annual average values are met. Days with levels of 50 $\mu g/m^3$ and higher mostly occur in the Netherlands when winds are southeasterly and there is less rain than usual. The new information collected during the last few years indicates that the chosen levels of 40 $\mu g/m^3$ as an annual average and 50 $\mu g/m^3$ as a daily average with 35 exceedances per year are not equivalent in the Netherlands. In the original EU position paper, the equivalence of both standards was a point of departure. Figure 2.15 shows that an annual average PM_{10} value of 40 $\mu g/m^3$ is equivalent to a daily value of 50 $\mu g/m^3$ with 80 allowed exceedances per year under average Dutch atmospheric conditions.

As has been made clear in Section 2 and will be elaborated in Section 6 on the current and future levels of PM in the Netherlands, an annual value of 40 $\mu g/m^3$ can probably be met in 2005. A further examination of the few grid cells where levels close to 40 $\mu g/m^3$ or exceedances are modelled is expected to clear away the remaining question marks. However, we will not be able to comply with a daily standard of 50 $\mu g/m^3$ with 35 permitted exceedances per year everywhere in the Netherlands in 2005.

Because the EU annual average and daily standards are not equivalent in the Netherlands, the Dutch government is advised to propose that the EU should bring both standards into line and promulgate two equivalent PM standards as envisaged in the original EU position paper. A daily PM_{10} standard of 50 $\mu g/m^3$ with 80 allowed exceedances per year would be equivalent to an annual average of 40 $\mu g/m^3$.

However, other, probably more practical daily average values with a different number of exceedances are also equivalent to an annual average level of 40 µg/m^3, as will be elaborated below. This value might even be a more sensible choice.

The measurement uncertainty in current PM measurements owing to the loss of semi-volatile material also renders the original choice of a value of 50 µg/m^3 debatable. In the 1997 EU position paper on PM_{10} (Seifert *et al.*, 1997) the majority of the working group advised a 50 µg/m^3 daily average concentration as a limit value for a 98-percentile. Besides the argument on 'change of slope' and the target for the Auto-Oil study, an important argument was the one put forward by the UK Expert Panel on Air Quality Standards that considered recommending a standard of 50 µg/m^3. This standard was based on a study in Birmingham (Wordley *et al.*, 1997), which showed one extra hospital admission daily for respiratory disorders with an increase in PM_{10} levels from 20 to 50 µg/m^3. What the 1997 position paper and the UK Expert Panel did not take into account, however, was the fact that these Birmingham air quality measurements were taken using a TEOM. In areas that are not explicitly arid (i.e. northwest Europe in general and the Netherlands in particular) and where semi-volatiles are abundant, TEOMs lose quite some material. The readings therefore need to be corrected for these losses. Recent 12-month intercomparisons of TEOMs as well as β-attenuation-type ('FAG') instruments in the Netherlands have shown that for average Dutch urban conditions the TEOM underestimates ambient PM concentrations by a factor of 1.9 compared with the FAG (using the standard correction factor of 1.3 for the FAG) (Voerman and van den Elshout, 2001 and Van den Elshout *et al.*, 2001). This would imply that TEOM readings need to be corrected by a factor of 1.9 in average urban conditions in the Netherlands. So a daily average of 95 µg/m^3 for the standard Dutch FAG would therefore be equivalent to a daily average of 50 µg/m^3 based on TEOM. Although the value of 1.9 for a correction factor certainly seems too high for general use in Europe (most correction factors presented by Williams and Bruckmann (2001) turned out to be considerably lower), the correction factor for TEOM in northwest Europe is certainly larger than unity. A more precise determination of it can be said to be essential for the process of European standard setting.

If for policy reasons a daily average standard is deemed necessary in addition to an annual average value, a higher value like a 98-percentile would be easier to work with than a value of 50 µg/m^3 with a large number of agreed exceedances. The concentrations of ambient air pollution, be it gases or PM, can generally be described by a log-normal distribution. Based on the EU annual average value of 40 µg/m^3 (as an arithmetic mean (AM)) with a certain geometric standard deviation (GSD), the resulting geometric mean (GM) can be calculated and a 98-percentile determined.

Table 2.5 presents the GSD for PM_{10} measurements for a number of recent years; the average GSD for the regional situation is 1.6–1.7. For the calculations presented in Figure 5.1 a value of 1.7 is taken, as this is the average GSD for the 7-year update of recent PM time-series studies in the Netherlands.

In Boleij *et al.* (1995) the following formula is presented for arriving at a GM (e^{μ}) on the basis of an AM and GSD (e^{σ}):

$$AM = \exp (\mu + \tfrac{1}{2} \sigma^2)$$

In this formula, $\mu = \ln(GM)$ and $\sigma = \ln(GSD)$.
Filling in the values of 40 for AM and 1.7 for GSD then produces a value of 34.75 as the GM, being equivalent to an annual average limit of 40 µg/m^3 in the Netherlands. The 98-percentile can then be calculated using the expression:

$$e^{(\mu + z*\sigma)} > \text{98-percentile}$$

The value of z is 2.055 when a 98-percentile is to be estimated (in the current case μ = 3.55 and σ = 0.53). This results in a calculated value of 103.6 µg/m^3. Allowing for a number of uncertainties in the actual PM measurements and other assumptions, this value of the 98-percentile is rounded off to 100 µg/m^3.

Figure 5.1 presents a scatter plot of the annual average PM_{10} levels in the Netherlands for three different years (1993, 1997 and 2000) for all the measuring sites in the NAQMN in relation to the 98-percentile for the annual average value at the site in question (blue diamonds). It also contains the theoretical relationship between the 98-percentile and the annual average concentrations for a GSD of 1.7 (pink squares).

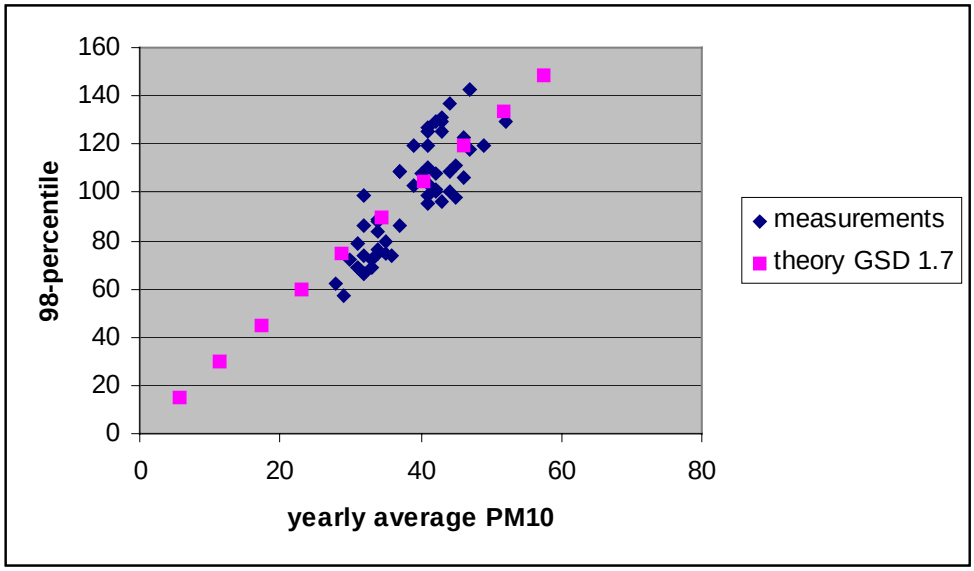


Figure 5.1 Annual average PM_{10} levels in the Netherlands for three different years (1993, 1997 and 2000) for all sites in NAQMN in relation to their 98-percentile.

It can be concluded from Figure 5.1 that the Dutch measurements confirm the above theoretical calculations of a 98-percentile of 100 µg/m^3, which is equivalent to an annual average value of 40 µg/m^3. The measurement uncertainty in the PM measurements can be seen in the large scatter around the theoretical line (connecting the pink squares in Figure 5.1). A 98-percentile of 100 µg/m^3 PM_{10} for a daily average is also equivalent to an annual average level of 40 µg/m^3 in the Netherlands. For reasons of risk management and practicability, a 98-percentile is preferred to a lower value of 50 µg/m^3 with a correspondingly higher number of allowed exceedances.

The working group that wrote the position paper recommended that the decisions taken on the basis of their advice should be reviewed within five years, assuming that more scientific information would then be available. Although the final word on PM cannot yet be spoken, there seems to be enough information to warrant revising a number of the previous positions.

When the annual and the daily average PM_{10} standards are equivalent they provide similar protection against health risks, as these are based on the current epidemiological RR and nearly linear concentration-response curves in the Netherlands. When the available concentration-response curves for PM are used, there are currently no health-based reasons that force us to have two different averaging times for risk management purposes. When the concentration-response curves level off slightly instead of being linear, the extrapolated total population risk at the higher PM levels, summed over a whole year, would be slightly better covered even with an annual average standard. When both standards can be said to be similar, it is up to the policymakers to decide which they prefer.

In the Netherlands, a large fraction (approximately 85%) of the *temporal, daily variations* in PM_{10} concentrations is caused by weather variations, whereas only 15% of the *variation* in annual average concentrations is influenced by meteorology. This indicates that local, regional and national authorities have limited opportunities for influencing daily average PM values. On the other hand, reducing annual average concentrations also results in a decrease in daily concentrations, although the pattern of daily variations remains unaltered. Furthermore, it must be remembered that the current measurement uncertainty in the automatic PM measurements becomes more manifest in daily concentrations than in annual average concentrations. The shorter the averaging time, the higher the chances of some of the measurements indicating exceedances. As a consequence of these measuring uncertainties, in a 'real world' situation with PM concentrations below, but close to, the standard, a number of daily PM measurements will indicate otherwise.
A third consideration regarding an annual average and a daily average standard is modelling. At present, deterministic dispersion models that can reliably model PM concentrations on a daily basis are not operational and have not been validated in the Netherlands. Model tools that have demonstrated they can do an adequate job with a reasonable level of precision in modelling annual average concentrations are currently operational. In the past, modelling tools have been very helpful and, as can be seen in this report, they can be called pivotal in estimating, predicting and evaluating the effectiveness of various abatement strategies.

For local (or regional) authorities the availability of a daily standard appears to be a helpful instrument for risk communication and advice to the public. As already mentioned, episode levels as such are not amenable to control by taking short-term emission reduction measures.

On the other hand, a 24-hour-average standard could easily be used for risk communication to the public. A more or less similar position regarding risk communication and 'alert' levels was previously taken for other components in EU daughter directives. For instance, an information system is currently in use in the Netherlands for ozone, PM_{10}, SO_2 and NO_2 and for pollen. However, the 24-hour standard of 50 $\mu g/m^3$ for PM_{10} is so low that this daily average is exceeded

somewhere in the Netherlands during six months of the year. A complicating factor with PM is that it is less clear how individual members of the public can actually protect themselves from the PM-associated health risks. Contrary to pollutants like ozone, SO_2 and pollen, indoor levels of PM_{10} and $PM_{2.5}$ are only slightly lower than ambient levels. Nevertheless, a reduction in physical activity can generally be recommended to reduce the amount of air inhaled and, hence, the amount of pollution inhaled on high pollution days. Also, avoiding participation in motorised traffic and avoiding major roads will help to reduce exposure to PM.

A value of 100 $\mu g/m^3$ with 7 exceedances per year is equivalent to an annual average value of 40 $\mu g/m^3$ in the Netherlands. If this value were chosen as the EU daily standard, 'alerting' the public would become more feasible. A complication with a risk communication scheme of this kind is that at present the prediction of exceedances above a certain level of PM_{10} on a daily basis is not well established. In the early 1990s, there were two types of risk communication or 'smog alert' in the Netherlands: 'summer smog' and 'winter smog'. The first one was triggered by a one-hour maximum ozone level of more than 240 $\mu g/m^3$ and the second by the sum of the daily PM_{10} and SO_2 concentrations exceeding a value of 450 $\mu g/m^3$. These alert levels for the general public were chosen because they were deemed to correspond to the threshold levels for health effects. This means that below these levels no health effects were presumed to occur and risk communication below these levels was therefore not considered necessary.

However, recent epidemiological research has demonstrated that there no longer seems to be a threshold for PM-related health effects, as health effects appear to exist at any level of PM. So, risk communication for non-threshold toxicants, indicating that any level of exposure carries a certain risk, would probably be more appropriate than one which specifically comes into operation when a certain PM_{10} threshold of 50 $\mu g/m^3$ or 100 $\mu g/m^3$ is exceeded.

Although the EU has proposed two standards for PM, there are several arguments that only one standard would suffice – annual mean concentrations being the best choice. However, for reasons of communication to the public and preventing exceedances above a certain threshold, daily standards may be appropriate.

5.4. Quantification of a standard

The principles on standard setting have been laid down in the policy document accepted by the Dutch parliament entitled 'Omgaan met risiko's' (Tweede Kamer, 1989, 21137/5), translated into English as 'Premises for Risk Management'. This document was completed before the current PM enigma emerged and its practical application to PM is rather complicated because of its far-reaching consequences. A dogmatic interpretation of these prevailing policy principles promulgated in 'Premises for Risk Management' would lead to a PM standard that is unrealistically low and even well below any natural background level in the Netherlands. (In order to comply with the principles in the policy document the annual average standard for PM has to be a factor 100 to 1000 lower than the current standard of 40 $\mu g/m^3$). Also, the uncertainties in the quantification of the PM-associated health risks described in this report make the results of such a dogmatic exercise highly questionable. Although the general principles in the policy document remain scientifically sound and are

currently still used successfully for setting standards for carcinogenic and other toxic substances, the Dutch Health Council (1995a) has recommended the commencement of a process of revaluation of the principles of standard setting for risks. So, unfortunately, for PM we will have to look elsewhere.

A possible way of proceeding on this complex issue has been described in two articles published forty years ago (Joosting and Zielhuis, 1962 and 1963). In these publications the authors describe the then prevailing American and Russian views concerning standard setting for ambient air pollution. The title of the publication captures the essence of both articles: pragmatism or dogmatism? The Russian view was that any detectable effect in an organism caused by exposure to a pollutant should not be accepted and as a consequence the standard should be set below that level. The American view at that time was described as a trade-off between the cost of the adverse health effects and the cost of abatement measures. In their articles the authors ended up taking a hygienic view, which is more or less a mix of both views. They recommend neither using solely economic arguments, nor dogmatically preventing any response in humans without even knowing if it is an adverse health effect.

In essence, this choice made in the 1960s still describes aspects of the current problems with PM standard setting. In the toxicological domain detectable effects and dose-response curves have been found in a host of *in vitro* and *in vivo* systems for different chemical and physical fractions of PM: LPS, transition metals, UF or crystalline material to name but a few. These are described in more detail in Section 4 on dosimetry and toxicology. The sources of these compounds and fractions are so diverse that virtually any source of PM or precursors seems to be relevant. A dogmatic application of these results would lead to unrealistically low standards, whose gains in health terms are impossible to predict. On the other hand, simply being pragmatic by making no more than the economic trade-off between the cost of health effects and the cost of abatement measures is just as unrealistic. In a careful process of weighing the pros and the cons, a balance needs to be found in which both economics and the ‘precautionary principle’ have a role to play. The ‘precautionary principle’ is enshrined in the Rio Declaration, which states that ‘where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures’.

It should be borne in mind that on a time-scale of decades the levels of PM have decreased considerably in the Netherlands (and elsewhere in northwest Europe). Part of this decrease is the result of policy measures taken by authorities, but part of it is also the consequence of autonomous economic progression in the last century, in which steam engines and the use of coal for domestic heating have been succeeded by electricity and natural gas. This trend also shows an unlinking of economic progress and PM air pollution in the Netherlands over the past decades. The general tendency continues towards a decrease in ambient PM levels in the Netherlands. Because we still do not know which sources are responsible for the PM-associated health effects, cost-effective abatement measures largely remain a shot in the dark. Therefore, the most cost-effective abatement policy at the moment will probably be to concentrate on the search for the causal factors. Obviously, not every part of PM is as relevant for health effects. Concentrating on the relevant sources will make effective risk reduction at lower cost more feasible. Naturally, these sources need to be identified first, so a specific research programme is required.

The fact that no threshold has as yet been discerned for PM_{10} implies that serious health consequences will still occur whatever the level of a standard. Strictly interpreting the concentration-response functions in Figure 3.1 in a causal manner indicates that a lower concentration also results in a smaller risk. However, the results of recent risk estimates in time-series studies presented in this report indicate that a decrease in PM levels does not necessarily lead to a subsequent decrease in estimated health effects.

In 1995, the expert report published by a Committee of the Health Council in the Netherlands concluded that 'it is up to the Government to decide what levels of reduction are socially acceptable and desirable and what emphasis should be placed on the health effects involved'. (Health Council, 1995b, p.66).

Since the Health Council report in 1995 no threshold has been found for PM-associated health effects. So, basically, there is no change in the current impossibility of setting a standard in the regular way to achieve a general level of safety from exposure to PM_{10} for the population. From a strictly scientific point of view this is where the advice would stop.

However, it could be argued that the current annual average standard of 40 $\mu g/m^3$ will still present problems in a number of places ('hot spots') in the Netherlands, especially when the currently used uniform correction factor of 1.3 turns out to be location- specific and assumes a value of say 1.45 in streets and 1.15 in regional situations (Visser *et al.*, 2001). The future situation could become even more complex if the correction factors also turn out to depend on the season (time). Given the seasonal dependency of ammonium nitrate levels in the Netherlands, this situation would be quite probable, though.

In the previous Sections a number of 'hot spots' are mentioned in regard to this annual average, based on the current correction factor of 1.3. One is that for a number of grids of 5 x 5 km^2 an annual average value in excess of 40 $\mu g/m^3$ is assessed for 2000 based on differential mapping and concentration modelling for a number of specific emission source categories, despite the fact that measured annual average values in the regional setting in the Netherlands were only 31 $\mu g/m^3$ in 2001. Some of the high values in a number of grids with this kind of assessment may be due to problems with the spatial attribution of a number of sources of coarse PM_{10} by the source category: storage and transshipment of material. For this category the modelling is still under scrutiny. A second 'hot spot' can be found in some urban street canyons. Despite the fact that the average value for the urban background was 33 $\mu g/m^3$ in 2001, TNO could still measure average values in a street canyon that locally do not rule out levels in excess of 40 mg/m^3 PM_{10}. This indicates that for a number of urban situations the combination of traffic and the local dispersion parameters, which are probably heavily influenced by the current urban built-up area and its geographical location, may result in unfavourable situations. This is a point of concern, which needs some extra attention in the near future.

For the 50 $\mu g/m^3$ daily PM_{10} average an assessment was made for 2010 in Section 2. This assessment took into account the influence of future abatement policies up to 2010. It came to the conclusion that even in that more favourable situation, with fewer

emissions in the Netherlands in 2010, the daily standard of 50 µg/m^3 would still be exceeded approximately 36–40 times a year. It also indicates that it will not be possible now or in the near future to comply with the value of 35 permitted exceedances of the daily average in 2005.

The current level of health risks associated with PM in the Netherlands is considerable and surpasses the maximum tolerable risk levels defined in the Dutch policy document 'Premises for Risk Management'. However vague and elusive the presented PM risk estimates sometimes seem to be, it should by any means be very clear that a maximum tolerable risk of one excess mortality per year per million inhabitants (which is the yardstick in the prevailing policy document 'Premises for Risk Management') is considerably exceeded by the PM-associated health effects. The Dutch policy document was, however, completed before the current PM enigma emerged and its practical application to PM is rather complicated because of its far-reaching consequences. An option of risk reduction by halving the crude PM_{10} standards in 2010 does not seem practically feasible, neither does it seem very sensible when the current knowledge of PM_{10} as a container is taken into account.

In conclusion, due to the continuing uncertainties concerning quantification of the health risks of PM, a PM standard in accordance with the prevailing Dutch policy document 'Premises for Risk Management' is not realistic. For this reason, the quantitative choice for a PM standard now shifts from the scientific domain to that of the policy domain. In spite of all the uncertainties and open questions concerning PM, the sheer magnitude of the estimated health effects causes it to remain a subject of serious concern and therefore a focal point of research and policy measures.

6. Reduction of PM levels

This chapter looks at the quantification of a major part of the risk chain, the part from sources to ambient PM levels. Technical options for further decreases in PM emissions and costs are presented and the question of the impacts on air quality of these emission reductions is considered. The final part of the risk chain – health effects and the associated risk reduction achieved by different control measures – will not be quantified in this chapter as we do not yet have enough information to make a reliable quantification. However, in the introduction to Section 6.1 some views are presented on a possible link between a reduction in PM levels and risk reduction in the general population.

Chapter 6 focuses on the available national abatement options for primary aerosol emissions. Additional options for controlling emissions of SIA precursor gases were not studied, neither were possible additional reductions or costs in foreign countries analysed. Throughout the chapter, the information presented concentrates on those fractions that are probably more health-relevant, such as primary PM and more particularly traffic-related pollution.

Section 6.2 presents future trends in emissions for the Netherlands and for Europe as a result of current legislation emission scenarios (CLE). Section 6.3 provides information on different abatement scenarios and costs. The abatement options mentioned briefly in Section 6.3 are worked out in more detail in Section 6.4. In Section 6.5 cost curves for PM_{10} and $PM_{2.5}$ emission abatement are compared. Section 6.6 presents the modelling results of the previously presented emissions and economic scenarios. Finally, Section 6.7 summarises all the information, and identifies and proposes cost-effective no-regret options for further PM risk reduction. Annex D presents some additional, more detailed information on PM emission reductions in the transport sector.

6.1. Reduction of PM and risk reduction

In the EU daughter directive the current annual average standard for PM_{10} for 2005 is 40 $\mu g/m^3$; the indicative annual average value for 2010 is 20 $\mu g/m^3$. Will a 50% reduction in ambient PM concentrations also result in a 50% risk reduction? However logical it may seem that a reduction in PM concentrations should also result in a similar reduction in health risks, on the basis of our current knowledge it remains a presumption. While there is such fundamental uncertainty about the causal factor(s) for the observed associations and while the mechanism(s) and sources remain unknown, we cannot present a reliable quantification.

It is one of the fundamental asymmetries in risk assessment sciences that relationships that can – and on the basis of the ‘precautionary principle’ should – be used by authorities to protect their population lead to different conclusions when risks are assessed than when risks have to be managed cost-effectively.

It is exactly this duality that fails to produce simple solutions, but that has to be explored in all its complexity to find some kind of optimal solution or, hopefully, a ‘no-regret’ option for PM reduction policies. Therefore, in this Section two extreme

hypotheses will be explored, more as a thought experiment, to assess their influence on policy options. The basic question underlying this exercise is whether a 50% reduction in ambient PM concentrations will also produce a 50% reduction in population health risks.

These two extreme hypotheses will be explored in more detail. Naturally, it would also be possible to formulate an infinite number of intermediate hypotheses for such an, in scientific terms, 'ill-posed' problem. However, these will not be elaborated here because it would take up too much space without really addressing the problems connected with PM reduction policies and their health impact.

The first hypothesis is that all population health effects are directly caused on a mass basis by the general mix of PM that forms PM_{10}, and that the concentration-response relationship of this mix is more or less linear and has no threshold. For the sake of argument, the policy consequences of such an extreme hypothesis will be explored. In this case, any reduction in annual average ambient PM levels would lead to similar reductions in health risks in the population. The optimum policy option then would be relatively straightforward, as the most cost-effective solution for reducing any PM mass would be the preferred choice. With this hypothesis it would make no difference if the amount of water or sea salt in the aerosol were reduced or if the levels of transition metals, particulate PAH, diesel soot or LPS were reduced in PM_{10}. The fact that such an extreme hypothesis would be highly improbable with our current knowledge should not preclude us from formulating it.

The other equally extreme hypothesis is that it is something else that causes the observed associations of PM and health effects. Because the association between this something else and the same meteorological influences that govern the dispersion of air pollution in the atmosphere is high, it associates closely with PM (and gaseous air pollution), though it is no part whatsoever of the measured PM levels. In epidemiological terms this phenomenon is known as residual confounding. In this case we are all looking the wrong way when we think of PM as the causal factor. With an extreme hypothesis of this kind, any reduction in ambient PM levels would lead to the counter intuitive result of an increase in population health risks. The answer to the question posed at the beginning of this Section concerning the influence of a 50% reduction in PM levels would then be that this reduction by half would result in a doubling of the PM-associated health risks (RR) in the population. The size of the true causal factor would not have been changed and the number of health effects it had would remain the same, while the levels of PM to which the health effects are spuriously ascribed would have been reduced by half, leading to a doubling of the RR per unit of PM.
Such an extreme hypothesis at the other end of the spectrum, also highly improbable with our current knowledge, should nevertheless be explored.

The truth, as so often, probably lies somewhere in the middle. However, where exactly is unfortunately impossible to say with any scientific certainty. On the basis of our current knowledge part of the PM mix appears to be somewhat less toxic, at least in the concentrations generally found in the ambient environment in the Netherlands. It seems less probable that the part of PM that is formed by water, sea salt and, possibly, non-crystalline crustal material or SIA is able to cause the observed PM-associated health effects in the Netherlands, the more so because no plausible

mechanisms could be found in laboratory animals or a number of human clinical studies for exposure to, environmentally speaking, extremely high concentrations of these pure substances. Bringing down the levels of the less toxic part of PM will therefore probably not have any measurable influence on health effects in the population, although such PM abatement measures might, of course, have other effects like better visibility or other secondary positive effects.

The lack of a clear causality therefore interferes with an effective general PM abatement strategy. However, focusing on the fractions that are probably more health-relevant, e.g. traffic-related PM, may at least enable some no-regret options to be identified, as has been done in the remainder of this chapter. Apart from this, the scientific community should continue to look for the vital information that is needed to tie sources causally to PM-associated health effects. A major effort will be necessary to reveal this vital information. When these sources have been found, a more cost-effective risk reduction strategy can be followed.

6.2. PM emissions trends – Current Legislation Scenario

6.2.1. In the Netherlands

Future emissions of PM_{10} and $PM_{2.5}$ under current policies have been projected in the Fifth Dutch Environmental Outlook (RIVM, 2000a). Emission projections were performed for PM_{10}. Projections for $PM_{2.5}$ were derived from PM_{10} using the size profiles presented in subsection 2.5.1.1. (see Table 2.12). For industry, the presented size fractions are the result of calculations made source-by-source at a more detailed level (TNO, 2002). At this detailed level of individual emission sources, $PM_{2.5}$ fractions were assumed to be constant over time. However, at the national level or at the level of main-source sectors, the growth pattern for $PM_{2.5}$ emissions could differ from PM_{10} due to differences in source contributions.

Projections include the latest knowledge on primary emissions in the Netherlands (see subsection 2.5.1.1.) and are based on socio-economic developments presented in long-term scenarios developed by the Netherlands Bureau for Economic Policy Analysis (CPB, 1997). Growth is high in the Global Competition scenario (3.3% per year) and low in the European Coordination scenario (2.7% per year). The projections consider all national and international policy measures decided upon before 1 January 2000. An overview of these current legislation measures is presented in Table 6.3. Base year for the projections was 1997 for industrial and agricultural sources, and 1995 for other sources. The results are presented in Tables 6.1 and 6.2. A graphical presentation of future emission trends for PM_{10} and $PM_{2.5}$ is presented in Figures 6.1 and 6.2.

Projections for PM_{10} presented in this study differ from the results of a recently published study on greenhouse emission trends to the year 2010 (RIVM, 2002). This greenhouse gases study was intended to prepare an update of Dutch emissions for greenhouse gases up to the year 2010 previously reported in the Dutch Environmental Outlook (RIVM, 2000a). The greenhouse gas emissions study incorporates new insights into emission levels of greenhouse gases only and socio-economic and energy-use developments. However, the current results of this study also incorporate new emission projections for PM_{10}, VOC, SO_2 and NO_x as far as

energy-related emissions are concerned. For non-energy-related emissions, no new estimates were prepared.

For PM_{10}, emissions in the current study and the greenhouse study differ by 15 ktonnes. This difference is not explained by newer and different insights into energy projection in the greenhouse gases study. Changes in transport forecasts cause only minor differences in PM_{10} emission levels. The difference between the two studies has to do with the completeness of the inventory. The greenhouse gases report does not include PM_{10} emissions from agriculture (9 ktonnes), from the ventilation air of industrial buildings (3 ktonnes) or transport emissions from wear of road, tyres and brakes (4 ktonnes).

This section mainly focuses on national emission trends and results are discussed in subsection 6.2.1.1. Only some international shipping emissions trends were included: maritime shipping emissions in Dutch ports and international inland shipping in the Netherlands. Future emission trends on the local urban scale are mentioned briefly in subsection 6.2.1.2.

Table 6.1 PM_{10} emissions for the EC and GC scenario.

	1998	2005	2010		2020		2030		*share in*
			EC	GC	EC	GC	EC	GC	*2010 emissions*
million kg									
transport	**19,3**	**15,9**	**11,4**	**11,7**	**11,5**	**12,5**	**13,0**	**14,7**	**27%**
combustion	*16,3*	*12,2*	*7,2*	*7,4*	*6,5*	*7,0*	*7,2*	*8,0*	*17%*
process	*3,0*	*3,7*	*4,1*	*4,3*	*5,0*	*5,5*	*5,8*	*6,7*	*10%*
industry	**16,1**	**13,6**	**13,2**	**13,5**	**13,1**	**15,2**	**13,5**	**15,7**	**32%**
combustion	*3,7*	*2,5*	*2,2*	*2,4*	*2,0*	*3,5*	*2,0*	*3,5*	*5%*
process, other	*9,6*	*8,4*	*8,3*	*8,4*	*8,4*	*9,0*	*8,8*	*9,5*	*20%*
process, building venting	*2,7*	*2,7*	*2,7*	*2,7*	*2,7*	*2,7*	*2,7*	*2,7*	*7%*
consumers	**3,9**	**3,5**	**3,4**	**3,5**	**3,3**	**3,4**	**3,4**	**3,5**	**8%**
combustion	*2,2*	*1,8*	*1,7*	*1,8*	*1,6*	*1,7*	*1,7*	*1,8*	*4%*
process	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*4%*
agriculture	**9,7**	**8,6**	**8,8**	**8,8**	**9,4**	**9,4**	**10,0**	**10,0**	**21%**
combustion	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0%*
process	*9,7*	*8,5*	*8,8*	*8,8*	*9,3*	*9,3*	*10,0*	*10,0*	*21%*
commercial and institutional sector	**2,5**	**2,6**	**2,7**	**2,7**	**2,7**	**2,7**	**2,6**	**2,5**	**6%**
combustion	*0,1*	*0,5*	*0,5*	*0,5*	*0,5*	*0,5*	*0,5*	*0,4*	*1%*
process	*2,5*	*2,1*	*2,2*	*2,2*	*2,2*	*2,2*	*2,1*	*2,0*	*5%*
construction	**1,2**	**1,3**	**1,5**	**1,5**	**1,9**	**2,0**	**2,3**	**2,5**	**4%**
combustion	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	**0%**
process	*1,1*	*1,2*	*1,4*	*1,5*	*1,8*	*2,0*	*2,3*	*2,4*	**3%**
waste treatment and disposal	**0,1**	**0,1**	**0,1**	**0,1**	**0,1**	**0,1**	**0,1**	**0,1**	**0%**
combustion	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0%*
process	*0,0*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0%*
energy sector	**0,6**	**0,6**	**0,5**	**0,7**	**0,4**	**0,3**	**0,4**	**0,3**	**1%**
combustion	*0,4*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0%*
process	*0,2*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0%*
TOTAL	**53,4**	**46,3**	**41,6**	**42,5**	**42,5**	**45,7**	**45,3**	**49,3**	**100%**
combustion	*22,8*	*17,8*	*12,3*	*12,9*	*11,2*	*13,2*	*12,0*	*14,2*	*30%*
process	*30,6*	*28,5*	*29,2*	*29,6*	*31,3*	*32,5*	*33,4*	*35,1*	*70%*

[b] Part of international shipping is included, i.e. international maritime transport in Dutch ports and international shipping on Dutch inland waterways.

6.2.1.1. National scale

a. PM_{10}

Without control measures, economic growth would cause PM_{10} emissions to increase, but thanks to these measures a different trend can be observed. Over the last twenty years, PM_{10} emissions have been reduced from about 115 ktonnes in 1980 to 55 ktonnes in 1998, i.e. a halving of emissions. This downward trend will continue in the years up to 2010, although the pace of reduction will be slower. From 1998 to 2010 emissions are anticipated to decrease by about 20% from 54 to 42–43 ktonnes. Transport emissions will be reduced most, as is illustrated in Figure 6.1. For mobile sources a decrease in PM_{10} emissions is anticipated from about 19 to 11–12 ktonnes, i.e. a reduction by about 40%. Emissions from industry, consumers and agriculture will also lessen somewhat. Emissions from other sectors, e.g. construction and storage and handling companies, are expected to increase slightly.

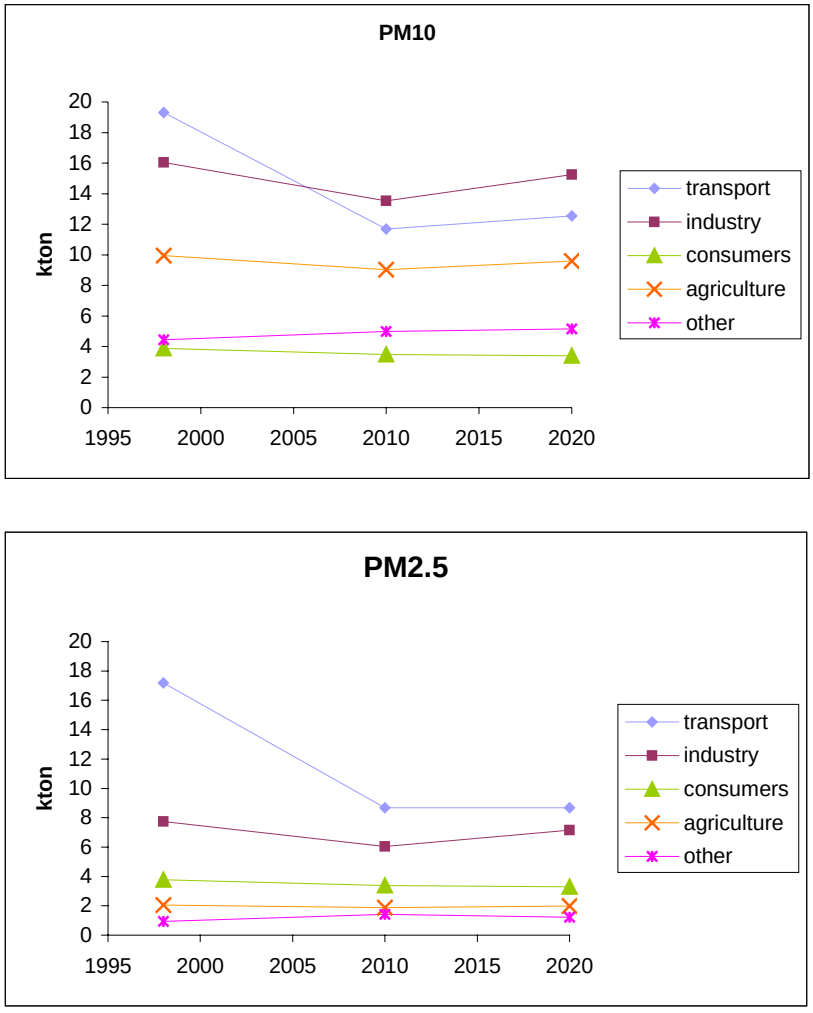


Figure 6.1 National emissions of PM_{10} and $PM_{2.5}$ for the GC scenario.

Estimates show that in 2010 exhaust emissions from road vehicles will no longer be the major source of transport emissions for PM_{10}. The EU has paid special attention to the control of these diffuse combustion emissions. In 2010, tailpipe emissions from road vehicles are anticipated to be only 2.5 ktonnes PM_{10}. About 4.3 ktonnes will be emitted due to wear of tyres, brakes and the road. Non-road mobile sources are anticipated to emit 4.9 ktonnes, of which 2.8 ktonnes is accounted for by inland and seagoing ships and 1.6 ktonnes by off-road mobile machines.

Emission reductions for the transport sector are the result of the continued tightening of existing EU emission regulations for exhaust emissions from diesel-powered road vehicles (Euro3/2000 and Euro4/2005), and of EU regulations for off-road mobile machinery (phase1/1998 and phase2/2000–2003). PM_{10} emissions from seagoing ships in Dutch ports will be reduced as a result of the cut in the sulphur content of heavy fuel oil foreseen in 2003. This measure is included in the baseline scenarios, but it should be realised that the underlying agreement is not yet in force. Besides exhaust emissions, road vehicles also emit PM_{10} due to wear of tyres, road surfaces and brakes. These emissions are uncontrolled and will increase in the future with the volume of transport. An increase in wear emissions by about 40% is anticipated from 1998 until 2010. Emissions from inland shipping are also expected to increase in the coming years despite the coming into force in 2003 of phase 1 emission limits agreed by the Central Commission for Rhine Navigation. These limits are only a first step and will not be effective in reducing actual PM_{10} emissions in the Netherlands because most ship engines already emit less PM than the new limit prescribes.

Emissions from industry will decrease from about 16 ktonnes in 1998 to about 13 ktonnes in 2010. The control of emissions from the sintering process in the ferrous industry and the shift from residual oil to gas for underfiring in refineries explains the larger part of this decline. Abatement of the sintering process has already been implemented in practice. The control of underfiring emissions from refineries has been agreed upon by industry with the competent authorities in the Netherlands. It is actually a spillover effect from measures to reduce emissions for SO_2. Smaller emission reductions come from the further control of process emissions in the fertiliser, glass and dairy industries through a combination of process-integrated and end-of-pipe measures.

Cleaner residential stoves enforced by the national certification schemes for new wood stoves, and other related measures directed at residential stoves and fireplaces, will cause a reduction in emissions by consumers from about 3.9 in 1998 to 3.4–3.5 ktonnes in 2010. This trend towards lower emission levels is reinforced by the current trend of increasing sales of artificial fireplaces and stoves with gas-fired or electrically lit logs. Presented trends do not account for a possible increase in so-called heavy wood users, which might be expected due to rising gas and oil prices.

Emissions from agriculture are anticipated to fall from about 10 to 9 ktonnes due to a decline in livestock numbers. However, it is not at all certain whether this materialised as a result of the EU-imposed development towards animal-friendly but PM-unfriendly animal housing systems, a trend which has not been taken into account here. The decrease presented therefore gives a rather optimistic view regarding future emissions from agriculture. Emissions from sectors other than industry, transport, consumers and agriculture will increase from about 4.4 ktonnes to 4.7–5.0 ktonnes.

The chemical composition of PM varies in accordance with the different emission sources. Apart from PM_{10}, black smoke (BS) and carbonaceous PM (EC/OC) are also frequently used as indicators for health-related effects. The emissions of PM_{carb} and black smoke (BS) were not estimated in this study. However, combustion-related emissions, which are closely linked to black smoke and PM_{carb}, were distinguished in the study from other non-energy-related process emissions as they are known. Corresponding future trends for both emission types are presented in Figure 6.2. From 1998 to 2010 combustion-related emissions of PM_{10} are anticipated to decrease by about 40% from 23 to 13 ktonnes, whereas process emissions will stabilise at a level of approximately 30 ktonnes. The large reduction in combustion emissions is explained by the already agreed further control of transport emissions.

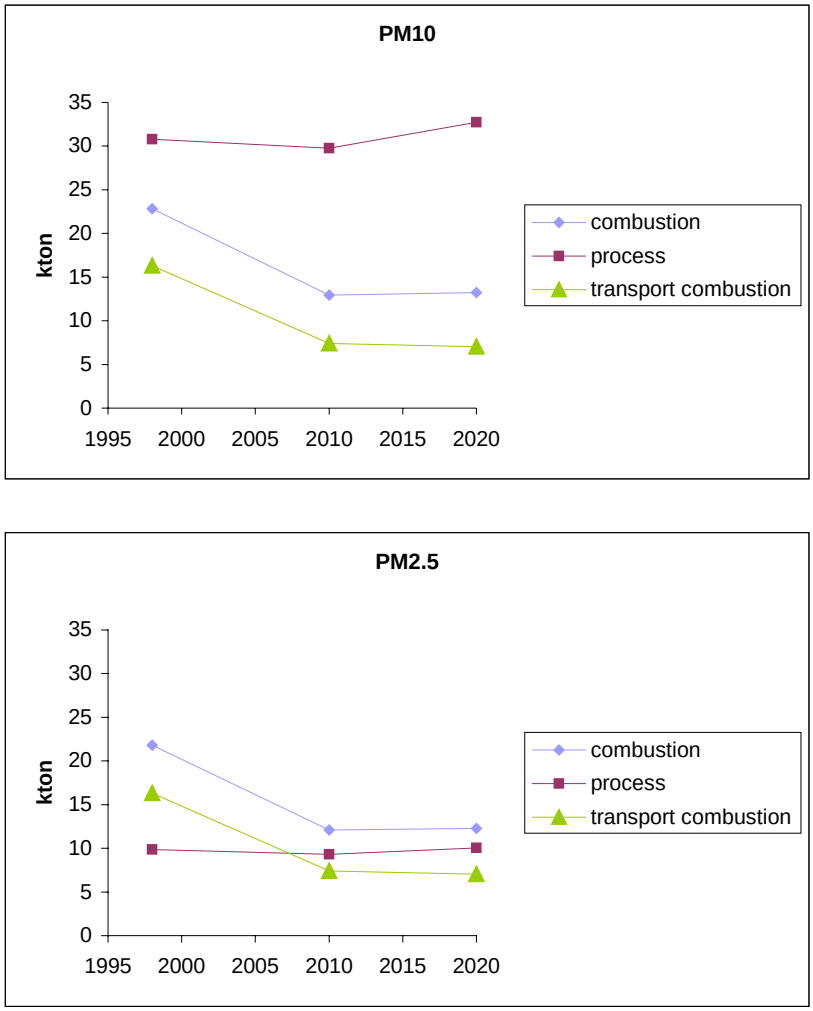


Figure 6.2 Combustion and process emissions of PM_{10} and $PM_{2.5}$ for the GC scenario.

b. $PM_{2.5}$

The anticipated emission reduction in the coming years will be larger for $PM_{2.5}$ than for PM_{10}. From 1998 to 2010, emissions of $PM_{2.5}$ are expected to decrease by about

30% compared with 20% for PM_{10}. The larger decline anticipated for $PM_{2.5}$ is explained by the fact that exhaust emissions from transport, which are all in the $PM_{2.5}$ range, contribute more and other sources less to total national emissions. Combined with the anticipated further control of exhaust emissions from vehicles in the next few years and the stabilisation of emissions from other sources, this explains the relatively larger decrease for $PM_{2.5}$ than for PM_{10}.

The results presented in Figure 6.2 show that $PM_{2.5}$ emissions are more closely linked to carbon-related combustion processes and less to non-energy-related mechanical processes. Currently, about 70% of $PM_{2.5}$ emissions are combustion-related, compared with 40% for PM_{10}. Due to the control of combustion-related transport emissions, the $PM_{2.5}$ share will decrease in the coming years. In 2010, about 60% of $PM_{2.5}$ emissions will be combustion-related and only 30% of PM_{10} emissions.

Table 6.2 $PM_{2.5}$ emissions for the EC and GC scenario.

	1998	2005	2010		2020		2030		*share in*
			EC	GC	EC	GC	EC	GC	*2010 emissions*
million kg									
transport	**17,2**	**13,3**	**8,5**	**8,7**	**8,0**	**8,7**	**8,9**	**10,0**	**41%**
combustion	*16,3*	*12,2*	*7,2*	*7,4*	*6,5*	*7,0*	*7,2*	*8,0*	*35%*
process	*0,9*	*1,1*	*1,2*	*1,3*	*1,5*	*1,6*	*1,7*	*2,0*	*6%*
industry	**7,7**	**6,1**	**5,8**	**6,0**	**5,7**	**7,2**	**5,8**	**7,4**	**28%**
combustion	*2,9*	*2,0*	*1,7*	*1,9*	*1,6*	*2,8*	*1,6*	*2,8*	*8%*
process, other	*4,0*	*3,3*	*3,3*	*3,4*	*3,3*	*3,6*	*3,4*	*3,8*	*16%*
process, building venting	*0,8*	*0,8*	*0,8*	*0,8*	*0,8*	*0,8*	*0,8*	*0,8*	*4%*
consumers	**3,8**	**3,4**	**3,3**	**3,4**	**3,3**	**3,3**	**3,3**	**3,4**	**16%**
combustion	*2,1*	*1,7*	*1,6*	*1,7*	*1,6*	*1,6*	*1,6*	*1,7*	*8%*
process	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*1,7*	*8%*
agriculture	**2,0**	**1,8**	**1,8**	**1,8**	**1,9**	**1,9**	**2,1**	**2,1**	**9%**
combustion	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0%*
process	*1,9*	*1,7*	*1,8*	*1,8*	*1,9*	*1,9*	*2,0*	*2,0*	*8%*
commercial and institutional sector	**0,3**	**0,7**	**0,7**	**0,7**	**0,7**	**0,7**	**0,7**	**0,6**	**3%**
combustion	*0,1*	*0,4*	*0,5*	*0,5*	*0,5*	*0,5*	*0,4*	*0,4*	*2%*
process	*0,2*	*0,2*	*0,2*	*0,2*	*0,2*	*0,2*	*0,2*	*0,2*	*1%*
construction	**0,2**	**0,2**	**0,2**	**0,2**	**0,2**	**0,3**	**0,3**	**0,3**	**1%**
combustion	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	*0,1*	**0%**
process	*0,1*	*0,1*	*0,1*	*0,1*	*0,2*	*0,2*	*0,2*	*0,2*	**1%**
waste treatment and disposal	**0,0**	**0,0**	**0,0**	**0,0**	**0,0**	**0,0**	**0,0**	**0,0**	**0%**
combustion	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0%*
process	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0%*
energy sector	**0,4**	**0,4**	**0,4**	**0,5**	**0,3**	**0,3**	**0,2**	**0,2**	**2%**
combustion	*0,3*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0%*
process	*0,1*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0,0*	*0%*
TOTAL	**31,7**	**25,9**	**20,7**	**21,4**	**20,1**	**22,3**	**21,4**	**24,0**	**100%**
combustion	*21,8*	*16,9*	*11,6*	*12,1*	*10,5*	*12,3*	*11,3*	*13,3*	*56%*
process	*9,8*	*9,0*	*9,1*	*9,3*	*9,6*	*10,0*	*10,1*	*10,7*	*44%*

[b] Part of international shipping is included, i.e. international maritime transport in Dutch ports and international shipping on Dutch inland waterways.

6.2.1.2. Urban scale

Central urban emissions are different from national averages. Sources such as the energy sector, industry, shipping and off-road machinery are usually concentrated on the periphery of urban areas or localised in rural areas in the Netherlands. Exhaust emissions from road vehicles will be reduced considerably in urban centres in the next few years, while domestic wood-burning and emissions from wear of tyres, brakes and road surfaces will become more dominant sources on the urban scale.

Table 6.3 Control measures assumed in the agreed measures emission scenario. All measures decided upon before 1 January 2000 have been taken into account.

In general

- Firm climate measures published in National Climate Policy Implementation Plan Part 1 have been included; measures for which implementation in 2010 is uncertain have been excluded.

Transport

- Diesel passenger cars and light duty vehicles (GVW < 1305 kg):
  Euro3/2000 (0.05 g/km and Euro4/2005 (0.025 g/km)
- Diesel light duty vehicles (GVW 1305–1760 kg):
  2001 (0.07 g/km) and 2006 (0.04 g/km)
- Diesel light duty vehicles (GVW > 1760 kg):
  2001 (0.10 g/km) and 2006 (0.06 g/km)
- Diesel heavy duty vehicles and buses:
  Euro3/2000 (0.13–0.21 g/kWh), Euro4/2005 and Euro 5/2008 (0.02–0.03 g/kWh)
- Mobile machines:
  Phase 1/1998 (0.54–0.85 g/kWh) and Phase 2/2000–2003 (0.2–0.8 g/kWh)
- Inland shipping:
- Phase 1/2001 Central Commission for Rhine Navigation (0.54–0.85 g/kWh)
- Inland shipping:
  Maximum sulphur content for gas-oil of 0.1% from 2008
- Seagoing ships[a]:
  maximum sulphur content for heavy fuel oil of 1.5%

Industry

- Oil refineries: shift (about 85%) from residual oil to gas for underfiring in refineries.
- Other sectors: measures announced by companies in their environmental planning reports (BMP) published every four years. Compared with the base year 1998, further measures are expected to be implemented in the following sectors: basic metals production, production of fertilisers and other chemical products, production of food and animal feed.

Consumers

- Testing of new stoves mandatory from 1997: CO concentration < 0.4 vol% for stand-alone wood stoves and < 0.5 vol% for built-in wood stoves
- Discouragement of the construction of chimneys for fireplaces in new homes
- Promotion of use of dry wood

Other sectors

No further control compared with the base year is anticipated.

[a] This agreement is not yet in force

6.2.2. Other European countries

Information on current and future PM_{10} emissions in other European countries in general and in neighbouring countries more in particular is necessary for the modelling of future PM_{10} air concentrations. The CEPMEIP European PM_{10} emission inventory for 1995 was used as a starting point (TNO, 2001). However, this study does not present emissions for future years. IIASA also prepares such projections but at the time the model runs were being prepared, these were not consistent with CEPMEIP 1995 data. To remain in line with CEPMEIP, RIVM therefore prepared its own emission projections for EU countries, taking CEPMEIP 1995 data as its starting point. Emissions for non-EU countries were based on the Fifth Dutch Environmental Outlook and are reported in Van Wee *et al.* (2001). The basis for these projections was an older version of the CEPMEIP inventory, namely for the year 1990.

For EU countries and for most categories the values from the CEPMEIP inventory for 1995 (TNO, 2001) were used as the starting point for projecting emissions for future years. PM_{10} emissions for EU countries were extrapolated from 1995 using:

- for non-transport combustion-related emissions (in the energy sector, industry and residential, commercial and institutional sector): emission trends per sector reported in the integrated assessment study conducted jointly by the European Commission and the EEA (RIVM, 2000b). This study presents several scenarios, one of which is called the baseline or BL scenario. The BL is based on a business-as-usual socio-economic scenario, assuming continued implementation of existing EU and ECE policies as far as decided on before August 1997. The BL only reflects emission standards laid down in EU directives and ECE protocols; more stringent national policies for PM emission control were not taken into account in this study.
- for industrial process emissions: country-specific trends in physical production (RIVM, 2000b).
- for construction and storage and handling: emission-trends for the Netherlands as calculated in the Fifth Dutch Environmental Outlook (RIVM, 2000a).
- for non-exhaust wear emissions from road transport, for other mobile sources and machinery, and for agriculture: country-specific emission trends as calculated in the Fifth Dutch Environmental Outlook (RIVM, 2000a).
- for other activities: emissions were held constant at the 1995 level.

For EU countries exhaust emissions from transport and emissions from municipal waste incineration in 2010 were taken from the results of the Fifth Dutch Environmental Outlook (RIVM, 2000a). The transport exhaust emissions were calculated on the basis of projected 2010 emission factors for the Dutch vehicle fleet, taking into account EU emission limits and forecast energy use per vehicle category as estimated in the aforementioned joint integrated assessment study (RIVM, 2000b).

6.2.3. Precursor emission trends

The modelling of future ambient PM_{10} concentrations requires information on emissions of the precursor gases involved in SIA formation, i.e. SO_2, NO_x, NH_3 and VOC.

National emission scenarios for these pollutants were based on calculations made for the Fifth Dutch Environmental Outlook. The results were reported in a background document focusing on emissions (Van Wee *et al.*, 2001). Information on the effects and costs of additional national control measures for these pollutants can be found in a report published by the Environment Ministry (VROM, 2000).

Current legislation emission scenarios for European countries were also based on results from the Fifth Dutch Environmental Outlook, i.e. the indicative emission ceilings proposed by countries during the latest negotiations on the Gothenburg protocol. The results of negotiations at the 31st meeting of the ECE Working Group on Strategies held from 26 August to 3 September were used as input for the atmospheric modelling calculations (UN-ECE, 1999). These provisional emission ceilings are similar to the final ceilings agreed on 1 December 1999 in Gothenburg, except for SO_2 and NO_x in Belgium and NO_x and NH_3 in Sweden. For both countries the final ceilings are somewhat lower.

The emission ceilings for SO_2, NO_x and NH_3 are based on a 5-year international research programme, which produced a great number of possible scenarios for the International Institute of Applied Systems Analysis. The set finally approved fits in well with the proposed IIASA scenario of agreed policies, based on the contribution of countries to the development of energy use and the expected development in production of the agriculture and industry sectors (IIASA, 1999a). A special implementation and compliance committee will closely monitor enforcement of the new protocol. It is therefore likely that agreed emission targets will be achieved.

6.3. PM emission abatement scenarios

This Section explores the technical possibilities for further reduction in emissions of particulate matter in the Netherlands in addition to agreed policies. It also presents emission scenarios indicating the range of emission reductions that could be achieved in the near future using available technology.

Subsection 6.3.1. lists technical control options and presents the maximum technically feasible reduction package assuming the full penetration of technology ($MFR_{ultimate}$). Next, subsection 6.3.2. puts forward an abatement package which results in an additional decrease in emissions of about 25% at a marginal cost of up to 55 euro/kg PM_{10}. In subsection 6.3.3. two other variants have been elaborated for the transport sector. The results for the different abatement packages are summarised in Table 6.5.

6.3.1. PM_{10} - Maximum technically Feasible Reduction (MFR)

Subsection 6.3.1. presents the ultimate technical potential for further emission reduction without looking at cost and assuming the full penetration of measures. The corresponding emission package is called the Maximum Feasible emission Reduction package MFR_{ult}. It shows whether the available technology is a constraint for achieving further reduction. Emission reductions and costs are presented per measure, per sector and for the Netherlands as a whole.

Maximum feasible reductions have been calculated using projected activity levels for the year 2010. It should be realised that these reductions are not quite realistic for that year, because technology is replaced gradually and replacement will never be complete in 2010 unless these technologies are taken out of service before the end of their normal life spans.

Control options for the transport, agriculture and consumers sectors were studied by RIVM (Smeets *et al.*, 2002). TNO studied applicable technical abatement options for industry, the energy sector and construction (TNO, 2002). Control options for the transport sector are discussed in detail in Annex D.

Table 6.4 lists the technical control options and shows estimated reductions and costs for the year 2010. Results show that available technology is not a constraint for further reduction of PM_{10}. The control options studied have the ultimate potential to reduce national emissions by about 60% compared with the 2010 current legislation emission level, i.e. from about 43 ktonnes to 16 ktonnes. The largest reduction potential, 9 ktonnes, is found in industry, followed by agriculture and transport with 7 and 6 ktonnes respectively. The cost is extremely high, i.e. about 6000 million euro per year.

There are two major reasons for this cost. First of all, the Dutch NeR and BEES guidelines (see Section 2.5.1.1.) impose strict controls on the treatment of exhaust waste gas streams with high emissions in industry and in the energy and waste sectors. Further control options produce only slight emission reductions because technology is state-of-the-art. Secondly, the abatement of other, less controlled sources is not as cost effective, because these emissions come from many sources with relatively low emissions (road vehicles, wood stoves, animal houses, construction sites), each requiring separate and costly abatement technologies. A detailed description of cost-effective technical control options is presented in Section 6.4.

6.3.2. PM_{10} – '$2010_{quart\ red}$' emission reduction

The ultimate PM_{10} reduction potential of 26 ktonnes presented in subsection 6.4.1. needs to be clearly distinguished from reductions that may be feasible in the future when cost considerations are also taken into account. This subsection shows further reductions that are feasible taking into account more realistic cost criteria. Based on the cost curve for industrial sources, a cut-off point of 55 euro/kg reduction emerged. An abatement package was therefore developed by selecting all the technical measures against this criterion of cost-effectiveness. Figure 6.3 shows that from about

55–60 euro/kg, the cost of measures in industry increases rapidly. Up to 55–60 euro/kg the cost curve appears relatively flat.

Table 6.4 PM_{10} emission reductions and cost of individual measures and package of measures for the maximum feasible reduction package and for '$2010_{quart\ red}$' variant. Results are presented for the year 2010, with the GC scenario as reference and assuming full penetration of selected measures.

sector	control measure	$MFR_{ultimate}$ emission reduction	$MFR_{ultimate}$ cost	25% cost-effective emission reduction	25% cost-effective cost	$MFR_{ultimate}$ range of marginal costs for distinct measures	average marginal cost MFR
		kton	*million euro per year*	*kton*	*million euro per year*	*euro/kg*	*euro/kg*
inland shipping	Euro-5 HDV technology (particulate traps)	1,8	10	1,8	10	4	4
mobile machines	Euro-5 HDV technology (particulate traps)	1,4	504			360	360
seagoing ships	particulate traps	0,7	*n.e.*	0,7	*n.e.*	*4*	4
seagoing ships	lowering S-content residual oil from 1.5 to 0.5 mass percent	0,1	*n.e.*	0,1	*n.e.*	*80*	80
LD diesel veh. and diesel p. cars	Euro-5 HDV technology (particulate traps)	1,2	3600			3000	3000
diesel road vehicles	emulsified diesel fuel	1,3	780			600	600
HD diesel vehicles	100% Euro-5 HDV technology (particulate traps)	0,4	120			300	300
HD diesel vehicles	LPG/CNG-fuelled vehicles with particulate traps	0,1	*n.e.*			*n.e.*	*n.e.*
passenger cars	road pricing (2 eurocent/km)	< 0.1	0				0
total transport		***5,8***	***5014***	***2,5***	***10***		***865***
refineries	Best Available abatement Techniques	2,0	75	2,0	75	24-45	38
basic metals	Best Available abatement Techniques	0,9	133			2-506	
food	Best Available abatement Techniques	2,2	136	1,8	51	7-746	29
chemicals	Best Available abatement Techniques	1,3	137	0,6	10	6-335	18
building materials	Best Available abatement Techniques	1,7	161	0,9	38	11-340	41
metal working	Best Available abatement Techniques	0,4	79			35-362	185
paper	Best Available abatement Techniques	0,4	55			148	148
rubber & plastics	Best Available abatement Techniques	0,0	33			539-889	763
textile	Best Available abatement Techniques	0,1	33			280	280
wood	Best Available abatement Techniques	0,4	107			255	255
total industry		***9,4***	***949***	***5,2***	***174***		***100***
total storage and handling companies		***2,0***	***25***	***2,0***	***25***	***13***	***13***
fireplaces	prohibiting the construction of chimneys for fireplaces in houses	0,3	*n.e.*			*n.e.*	*n.e.*
stoves	100% certified stoves (Dutch CO standard: 0.4/0.5 vol%)	0,2	*n.e.*			*n.e.*	*n.e.*
stoves	tightening Dutch emission standards (50% extra PM10 reduction)	0,4	*n.e.*			*n.e.*	*n.e.*
stoves	tightening emission standard (DIN+-standard 0.12 vol% CO)	*n.e.*	*n.e.*			*n.e.*	*n.e.*
total consumers		***0,5***	***n.e.***				***n.e.***
pigs: animal housing systems	filtering ventilation air (low energy scrubbers)	2,5	*n.e.*			*n.e.*	*n.e.*
poultry: animal housing systems	filtering ventilation air (low energy scrubbers)	4,8	*n.e.*			*n.e.*	*n.e.*
pigs: animal housing systems	spraying in the house, e.g. with plant oil	0,7	*n.e.*			*n.e.*	*n.e.*
poultry: animal housing systems	spraying in the house, e.g. with plant oil	1,3	*n.e.*			*n.e.*	*n.e.*
total agriculture		***7***	***n.e.***				***n.e.***
construction sites	Best Available abatement Techniques	1,3	240			190	190
total construction		***1,3***	***240***				***190***
storage and handling coal	Best Available abatement Techniques	0,1	0,1			2	2
combustion	100% shift from coal to gas	0,1	*n.e.*			*n.e.*	*n.e.*
total energy sector		***0,2***	***n.e.***	***0,1***	***0,1***		***n.e.***
waste combustion	Best Available abatement Techniques	0,0	2,0			182	182
total waste treatment and disposal		***0,0***	***2***				***182***
Total		***26,2***	***6230***	***9,8***	***209***		***238***

It should be realised that although the '$2010_{quart.\ red}$' selects only the cheaper measures, it still assumes full penetration of the selected measures.

Implementing all technical measures up to a marginal cost of 55 euro/kg reduction will reduce emissions of PM_{10} in 2010 by about 10 ktonnes, i.e. 25% at an annual cost of about 210 million euro. This is in the same order of magnitude as the cost of extra climate change measures (on top of already agreed measures) to achieve the Dutch target for greenhouse gases as agreed in the Kyoto protocol[1]. The annual cost amounts to about 10 million euro for the shipping sector, 175 million euro for industry and 25 million euro for specialist storage and handling companies not located within industry.

Transport emissions would be reduced by about 2.5 ktonnes, i.e. 30%. This reduction is fully explained by additional measures taken in the shipping sector. The marginal cost of abating shipping emissions is only 4 euro/kg PM_{10} reduction.

Emissions from industrial activities (including storage and handling activities in specialist non-industrial companies) would be reduced by about 7.3 ktonnes, i.e. 45%. Cost-effective extra reductions are mainly located in the classical high-emission industries and in storage and handling companies. Cost-effective reductions can be found in storage and handling companies (2 ktonnes), refineries (2 ktonnes), food industry (1.8 ktonnes), building materials (0.9 ktonnes) and the chemical industry (0.6 ktonnes). Measures for the basic metals industry are also cost-effective, but these have already been included in the current legislation scenario.

6.3.3. PM_{10} – Transport abatement scenarios

Cost-effective measures up to 55 euro/kg are included in the '$2010_{quart\ red}$' variant. This package still assumes full penetration of all these measures in 2010. However, the implementation of quite a number of measures is gradual, e.g. the introduction of new emission standards for new motor vehicles. The turnover time of the vehicle fleet and installations determines how long it will take before all vehicles and installations have been replaced. As already mentioned, this illustrates that assuming full penetration of technology in the year 2010 is overoptimistic.

An attempt was therefore made to estimate the Maximum technically Feasible emission Reductions (MFR) that could be implemented by 2010 assuming standard turnover rates of technology. This exercise was only performed for the transport sector. The results are presented as the MFR_{real} package. A distinction was drawn between reductions that could be achieved by national actions only and what might be possible if the EU also took extra steps.

Further EU steps supported by appropriate national action would reduce PM_{10} emissions by at most 2.0–2.6 ktonnes, i.e. 15–20% compared with the current legislation scenario. Almost all transport measures demand a European approach. The maximum reduction of 2.0–2.6 ktonnes requires Euro4/5 emission limits for inland shipping, passenger cars, light duty vehicles and mobile machinery. In addition, an agreement is needed on the retrofitting of particulate traps on all seagoing ships sailing under European colours. Conventional diesel should be replaced by a mixture

[1] Annual reduction of 25 Mtonnes CO_2 equivalent (= estimate for gap between target and emissions according to agreed policies) at an assumed average price of 10 euro per avoided tonne CO_2 equivalent will cost 250 million euro per year.

of diesel fuel and water (Lubrizol).

Table 6.5 PM_{10} emissions and abatement cost for different packages according to the Global Competition scenario.

	PM_{10} emissions (ktonnes)				
Sector	**CLE – Current Legislation**	**$2010_{quart.\ red.}$**	**MFR_{ult}**	**$MFR_{realistic}$ – National policy action**	**$MFR_{realistic}$ – National and EU policy action**
Transport	11.7	9.2	5.9	10.9	9.1
Road exhaust	*2.5*	*2.5*	*0.6*	*2.1*	*1.1*
Non-road exhaust[b]	*4.9*	*2.4*	*1.0*	*4.5*	*3.7*
Wear	*4.3*	*4.3*	*4.3*	*4.3*	*4.3*
Industry	14	8	4		
Commerc. and institut.	3	1	1		
Agriculture	9	9	2		
Consumers	4	4	3		
Other	2	2	1		
Total	**43**	**33**	**16**		

	Cost compared with Current Legislation (million euro/yr)				
Sector	**CLE – Current Legislation**	**$2010_{quart.\ red.}$**	**MFR_{ult}**	**$MFR_{realistic}$ – National policy action**	**$MFR_{realistic}$ – National and EU policy action**
Transport		10[a]	5000	120	1000
Road exhaust			4500	120	900
Non-road exhaust[b]		10[a]	500	2	100
Wear			n.e.	n.e.	n.e.
Industry		175	950		
Commerc. and institut.		25	25		
Agriculture			n.e.		
Consumers			n.e.		
Other		0.1	250		
Total		**210**	**6000**		

[a] Excluding costs for technical measures on seagoing ships, which were not estimated.
[b] Part of international shipping is included, i.e. international maritime transport in Dutch ports and international shipping on Dutch inland waterways.

Although most action requires a coordinated EU approach, there is also room for further national action. Encouraging the replacement of old engines used in inland shipping, encouraging the retrofit of particulate traps on pre-Euro4 heavy duty vehicles or encouraging the early penetration of Euro4/5 heavy duty vehicles all constitute action that could be taken by national authorities. These national initiatives could reduce PM_{10} emissions in 2010 by at most 1.0 ktonnes.

In summary, an additional reduction in emissions of about 1.0–2.6 ktonnes on top of agreed measures (i.e. 10–20% of transport emissions) seems realistic by the year 2010. A 1-ktonne reduction could be achieved with additional national policy action. A further reduction of up to 2.6 ktonnes requires fast, concerted action by the EU based on best available technologies.

6.4. PM_{10} – Abatement options in detail

6.4.1. Industry

For industry, the Maximum technically Feasible Reduction potential (MFR_{ult}) comes to about 9.5 ktonnes, i.e. 70% of the agreed measures reference level in 2010 and 60% of the 1998 level. The cost-effectiveness of additional abatement options ranges from 2.3 to about 900 euro/kg PM_{10} reduction. Measures costing less than 2.3 euro/kg have already been taken in the Netherlands under requirements formulated in the Dutch emission guidelines (NeR) and combustion plant emission requirements (BEES).

NeR states a value of 2.3 euro/kg as an indicative reference point for the acceptability of costs for reducing total particulate matter (TSP). Because PM_{10} is only a part of TSP, the value needs to be adjusted for this component. For PM_{10} the value of 2.3 euro is the lower limit when assuming 100% of TSP emissions are made up of PM_{10}. For exhaust flows with 50% $PM_{10,}$ for example, the corresponding PM_{10} value is 4.6 euro/kg.
When considering options for further abatement in industry, a distinction should be drawn between 1) diffuse emissions from storage and handling activities, 2) ventilation emissions from industrial buildings, and 3) process and combustion emissions through stacks.

1) Focusing on diffuse emissions from storage and handling activities, the NeR contains specific instructions for emission prevention. However, many of these measures are not yet general practice in industry. Further reduction of PM_{10} emissions can therefore be achieved through stricter implementation of the NeR guidelines for clean storage and handling practices.

2) Ventilation emissions from industrial buildings constitute 20% of PM_{10} emissions in industry in 2010 and are mostly uncontrolled because of the large air flows combined with low dust concentrations (0.5–5 mg/m^3). The cost-effectiveness of filtering these large air flows is low. Better cost-effectiveness can be achieved by combining source evacuation on emission sources and subsequent standard purification measures.

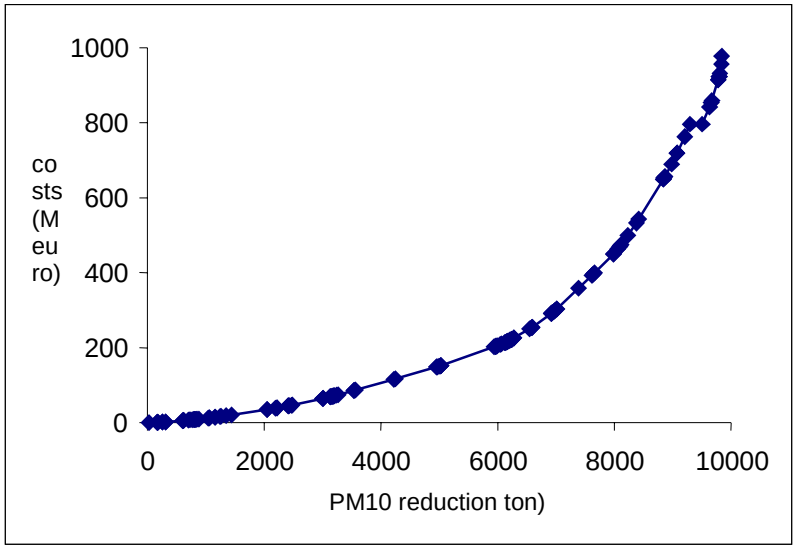
3) Process and combustion emissions through stacks can be reduced by applying the most advanced dust-cleaning methods. Basically, only high-energy scrubbers, electrostatic precipitators and fabric filters are applicable. Looking at the situation in 2010, it is possible to distinguish three abatement cases: (1) abatement of emissions where no abatement techniques are used, i.e. processes with small gas flows and low concentrations, resulting in limited cost effectiveness, (2) the optimisation of installed abatement techniques, and (3) the application of additional reduction techniques besides those in place. The efficiency of these three options ranges from 75–90%. Process-integrated measures were not taken into account in this study.

The marginal cost curve for PM_{10} reduction by industrial source is presented in Figure 6.3, with the current legislation GC emission level as reference. The slope of the cost curve is relatively slight up to a reduction of about 5 ktonnes, i.e. approximately 40% of the agreed measures reference level. Up to this point, reductions can be reached at

marginal costs of less than 55 euro/kg PM_{10}. These cost-efficient measures are all found in the high-emission industries, i.e. refineries, chemicals, food and building materials. Measures for the basic metals industry are also cost-effective, but these have already been included as agreed policies. In the range up to 5 ktonnes of cost-effective reduction almost no options are found for other sectors of industry, e.g. wood, paper, textile, rubber and plastics. The total extra cost of PM_{10} reduction up to 5 ktonnes in industry is approximately 200 million euro per year. More ambitious reductions are accompanied by a relatively sharp increase in marginal cost. An additional reduction of 3–8 ktonnes more than doubles the total cost to 450 million euro per year.

To mark the significance of the cost figures presented for sectors of industry, the cost of further PM_{10} reduction has been expressed as a percentage of the Added Value for different industrial sectors in 1998 (TNO, 2002). The Added Value for an industrial sector is the value that is added to the resource materials by transforming a product. This has been done for the 5-ktonne reduction scenario, i.e. taking into account all measures with a marginal cost of 60 euro/kg. In this case, associated costs for the five high-emission sectors are in the range of about 0.2–2% of Added Value. The burden for all of industry is 0.5% of Added Value. In the case of reductions higher than 5 ktonnes, the burden for individual sectors will increase sharply.

The cost curves presented in Figure 6.3 assume that all measures are selected and implemented on the basis of cost-effectiveness, resulting in a least-cost reduction strategy for PM_{10} emissions from industrial sources. Current Dutch policy on PM_{10} emission reduction is firmly based on generic emission standards (mg PM_{10}/m^3 waste gas). Tightening of these generic standards will not be the most cost-effective option for realising additional reductions in individual situations. At PM_{10} reduction rates of 30–65%, the cost of generic emission concentration standards is approximately twice as high as that of a cost-effective strategy (TNO, 2002). This is illustrated in Figure 6.4. Generic emission standards result in higher costs since all processes, including the ones with small gas flows and high unit costs, are obliged to install expensive filters. Possibilities for achieving cost-effectiveness are the use of market-based instruments such as emission taxes or tradable permits, making agreements with sectors, or by changing NeR requirements for generic emission standards in such a way that companies are required to take measures below a specific marginal-cost level (euro/kg PM reduction).



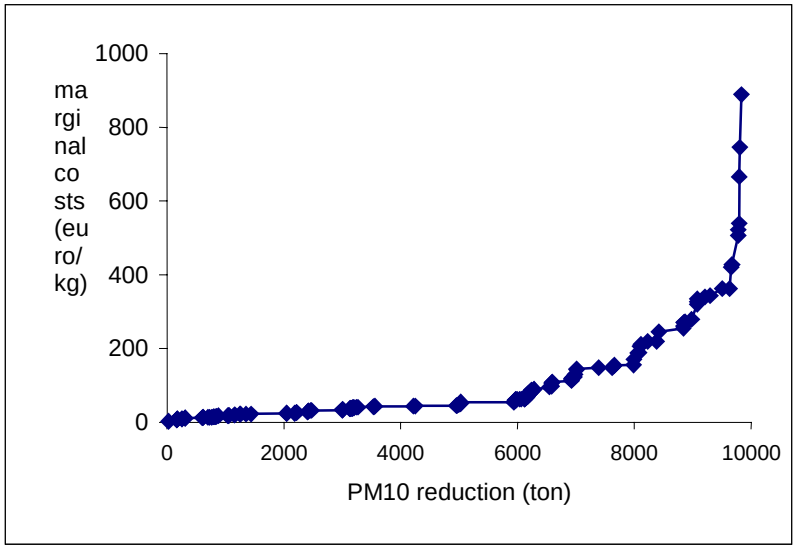


Figure 6.3 Total annual reduction costs (M EUR, a) and marginal costs of measures (euro/kg, b) as a function of the total cumulative reduction by Dutch industrial sources in 2010 for a cost-effective PM_{10} reduction strategy with GC agreed measures emissions as reference.

6.4.2. Transport

For transport, the ultimate Maximum potential for Further Reduction (MFR_{ult}) reaches about 6 ktonnes PM_{10}, i.e. 50% of the reference level in 2010 and 30% of the 1998 level. The cost-effectiveness of additional abatement options ranges from 4 to about 3000 euro/kg PM_{10} reduction.

Further PM reductions in the transport sector are technically feasible for inland and seagoing ships, for mobile machines as well as for road vehicles.

The most cost-efficient option is the control of PM emissions from inland and seagoing ships. Current international regulations for these sources are limited, so in 2010 these sources will still be controlled to a lesser extent than other economic sectors. With current legislation, anticipated future emission limits for PM_{10} (kg/kWh) from ship engines and mobile machinery (used in industry, agriculture and the construction sector) are still high compared with the Euro4/5 diesel technology that has been agreed for heavy duty vehicles. For inland shipping, emission limits agreed by the Central Commission for Rhine Navigation are 25–40 times higher than Euro5 limits, depending on the engine size. Technically, there are no restrictions for applying Euro4/5 Heavy Duty Vehicles (HDV) technology, which requires particulate traps, in the shipping sector. However, this demands coordinated action within international bodies such as the Central Commission for Rhine Navigation and the International Maritime Organisation. To achieve this, the EU and member states need to play an active part. The marginal cost of these measures is 4 euro/kg reduction. The cost of implementing Euro4/5 technology on all *inland ships* is about 10 million euro per year.

Another option in the transport sector is the further control of emissions from relatively uncontrolled mobile machines, although cost-effectiveness is less favourable than for ships. EU emission regulations for engines used in mobile machinery are more stringent than those for inland ships, but not as strict as Euro4/5 standards for heavy duty road vehicles. Depending on the size class of the engine, emission limits for mobile machines are 10–40 times higher than Euro5 limits. Euro4/5 HDV diesel technology, which includes particulate traps, could also be used for mobile machinery. The total annual extra cost of introducing this technology for mobile machinery is approximately 500 million euro.

Emissions from road vehicles can be reduced by applying advanced Euro5 diesel technology, which has to be developed for heavy duty vehicles and will also be applied to passenger cars and light duty vehicles, or by the use of emulsified diesel fuel, i.e. a mixture of diesel fuel and water.

Road pricing for passenger cars (2 eurocent/km for passenger cars and 16 eurocent/km for heavy duty vehicles) produces only minor PM_{10} reductions. At these price levels, road pricing is not considered capable of reducing PM_{10} emissions significantly.

6.4.3. Agriculture

The reduction potential for the agricultural sector comes to about 7 ktonnes, i.e. 80% of the reference level in 2010. For this sector, only measures for reducing emissions from animal housing systems were studied.

Emissions from animal housing systems can be reduced by applying a filtering system to the ventilation air. The cost-effectiveness of this measure is limited, however, because ventilation flows are large and dust concentrations are relatively low. No quantitative estimates were made. Spraying a small quantity of plant oil in the animal house is an alternative abatement option. Dutch research showed dust reductions of up to 25% in pig houses (Roelofs and Binnendijk, 2001). Takai *et al.* (1998) reported reductions of up to 50–90% for this abatement option. Besides removing dust from

the ventilation air, this option also reduces PM concentrations inside the animal houses and so is also beneficial from the point of view of farmer and animal health.

6.4.4. Consumers

The cost and effects of measures for reducing emissions from residential sources were not studied in detail for this report. Full penetration of certified stoves, an agreed measure that will still not be fully realised in 2010, will reduce emissions by 0.2 ktonnes, i.e. about 5% of the current legislation level in 2010.

Currently, the obligatory certification of new wood stoves acts as the most prominent measure for controlling PM_{10} emissions from residential combustion. This certification is aimed at reducing CO levels. The reduction of PM_{10} is a spin-off. To control emissions from fireplaces the government has issued recommendations to municipalities expressing disapproval of the construction of chimneys for fireplaces. These agreed measures for stoves and fireplaces could be further extended, e.g. tighter CO emission standards or specific standards for PM_{10} could be implemented for stoves, and the construction of new chimneys in houses could be prohibited. Another measure with a potentially large impact is the promotion of technical innovations, e.g. heat exchangers to improve the energy efficiency of stoves. Other measures include promoting the use of well-dried wood and supplying the consumer with information regarding optimum size/capacity and correct instructions for use. Promoting the use of well-dried wood is expected to have only a minor impact because most of the wood burnt in houses originates from informal trade flows out of reach of any certification scheme for well-dried wood.

It needs to be realised that additional abatement options directed at residential wood combustion require further attention because these measures are promising options in relation to health-relevant combustion-related emissions, and even more so if one focuses on the urban and street level. A complete ban on wood burning in houses would reduce national combustion-related PM_{10} emissions by about 10%.

6.4.5. Construction

Emissions of PM_{10} from construction sites can be reduced by about 1.2 ktonnes, i.e. by 80%. The reduction potential in this sector is larger than that of any of the studied measures in the industrial sector. However, the cost-effectiveness of this measure is low due to the diffuse nature of the emissions.

6.4.6. Storage and handling companies

Emissions of PM_{10} from specific non-industrial companies specialising in materials handling can be reduced by about 2.0 ktonnes. The cost-effectiveness of this measure is comparable to extra options in high-emission industries. Materials handling abatement options in industry are incorporated under the sector industry.

6.5. PM_{10} and $PM_{2.5}$ – Cost curves for industrial sources

6.5.1. Comparison of cost curves

Cost curves for PM_{10} emission reductions in industry were presented in the previous section. Subsections 6.5.1. and 6.5.2. will extend this analysis to include $PM_{2.5}$. Note that Figures 6.4 and 6.5 take as reference a frozen technology case (TNO, 2002) which does not account for the effects on emissions of the already agreed measures. For this reason, PM_{10} emissions in 2010 are about 5 ktonnes higher than the current legislation emission level, which was the basis for the PM_{10} cost curves presented in Figure 6.3.

An analysis will be presented of the impact on overall industrial PM reductions and abatement costs of the use either of PM_{10} or of $PM_{2.5}$ as a basis for designing reduction policies. The cost curve for $PM_{2.5}$ was developed on the assumption that PM_{10} and $PM_{2.5}$ emissions are removed equally efficiently by abatement technologies, since all additional technologies studied have a reduction efficiency for $PM_{2.5}$ of more than 95% of the achieved PM_{10} reduction (TNO, 2002).

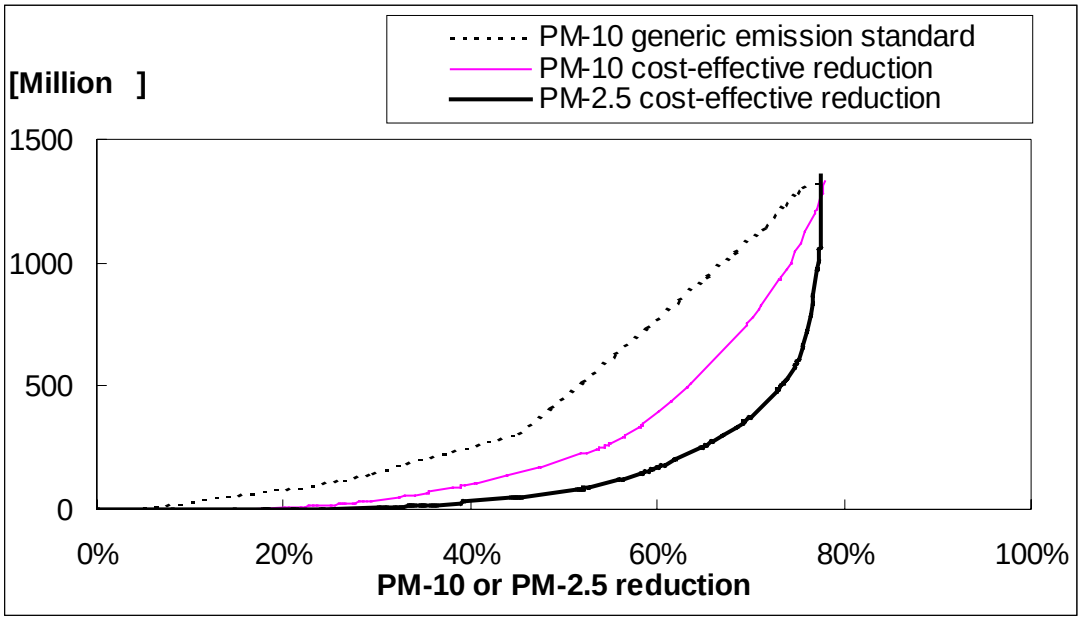


Figure 6.4 Total annual reduction costs as a function of the total reduction percentage (with respect to the 'frozen technology' reference scenario) achieved by Dutch industrial sources in 2010 in three cases, viz. a cost-effective $PM_{2.5}$ reduction strategy, a cost-effective PM_{10} reduction strategy and the application of a general PM_{10} concentration standard (indicated in the graph).

The presented cost curves for industrial sources for PM_{10} and $PM_{2.5}$ clearly illustrate that reductions of $PM_{2.5}$ at rates of 30–65% (compared with the frozen technology case) are possible at approximately half the cost of reducing PM_{10}. This phenomenon

arises because $PM_{2.5}$ size fractions are not the same for all industrial sources. The cost of the $PM_{2.5}$ reduction case is lower because a cost-effective $PM_{2.5}$ reduction strategy focuses first on sources with high $PM_{2.5}$ fractions and relatively low gas flows. These sources might be different from the sources selected when the focus is on cost-effective PM_{10} reduction. At the maximum feasible reduction rates of almost 80%, cases for PM_{10} and $PM_{2.5}$ involve similar cost, as all options available in the database were applied.

6.5.2. PM_{10} or $PM_{2.5}$ as basis for reduction policies

Besides the aspect of cost, what will be the consequence of choosing PM_{10} or $PM_{2.5}$ as a basis for reduction policies?

PM_{10} includes all $PM_{2.5}$, while $PM_{2.5}$ excludes all larger particles within PM_{10} (2.5 µm < d < 10 µm). Reducing PM_{10} also results in a reduction in $PM_{2.5}$, depending on the $PM_{2.5}$ fraction. Reducing $PM_{2.5}$ will also lead to a reduction in larger particles in most cases in industry, because in almost every case the $PM_{2.5}$ fraction is lower than 1.

In a cost-effective PM_{10} reduction strategy, the first attractive PM_{10} reductions contain high fractions of $PM_{2.5}$, resulting also in relatively high reduction shares for $PM_{2.5}$. This is illustrated in Figure 6.5, where the first 50% PM_{10} reduction leads to a 60% $PM_{2.5}$ reduction. It is a matter of coincidence that sources with cost-effective PM_{10} reductions also have relatively high fractions of $PM_{2.5}$. The combination of the physical properties of large-scale production processes and the economic aspects of the abatement options applied to these processes cause this phenomenon. It is probably a general phenomenon that also occurs in other European countries, although its importance depends on the underlying processes and local sector structures.

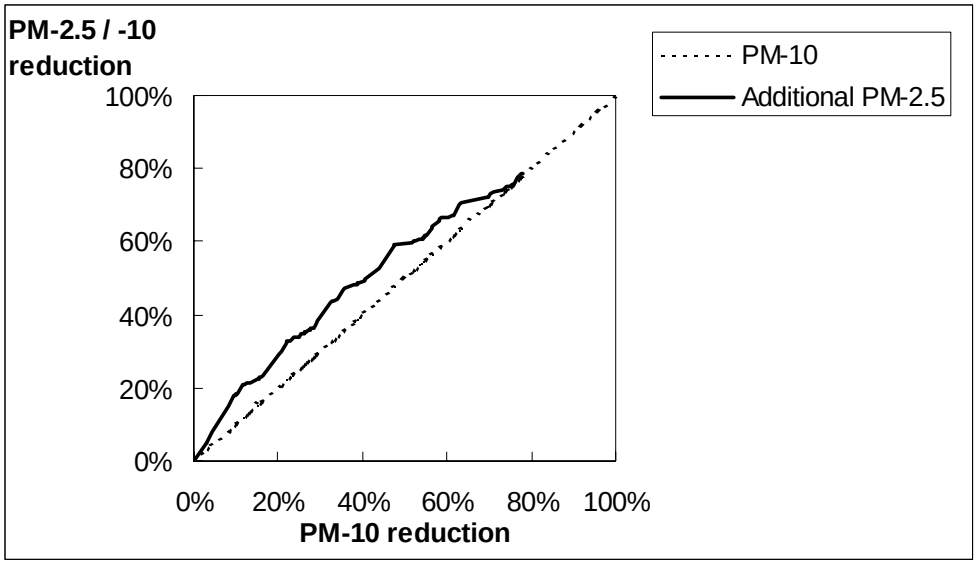


Figure 6.5 Additional $PM_{2.5}$ emission reduction as a result of cost-effective PM_{10} reduction policies presented as a function of the level of PM_{10} reduction for Dutch industrial sources.

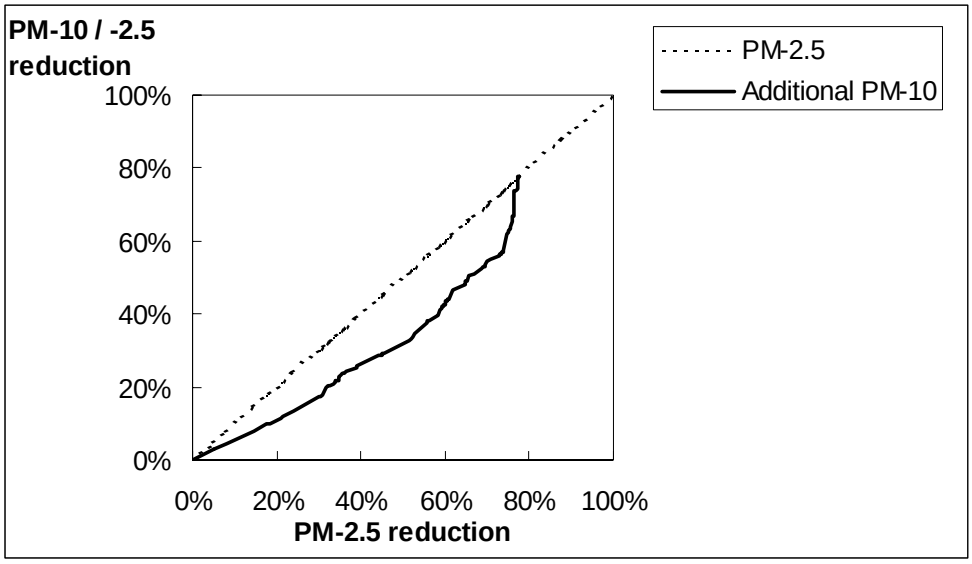


Figure 6.6 Additional PM_{10} emission reduction as a result of cost-effective $PM_{2.5}$ reduction policies presented as a function of the level of $PM_{2.5}$ reduction for Dutch industrial sources.

Conversely, $PM_{2.5}$ reduction always leads to PM_{10} reduction as well, since most industrial sources in the Netherlands have size fractions lower than 1. However, a cost-effective strategy for $PM_{2.5}$ focuses on the sources with high $PM_{2.5}$ fractions, resulting in relatively low additional PM_{10} reductions. In Figure 6.6, the first 50% $PM_{2.5}$ reduction leads only to a 30% PM_{10} reduction.

The conclusion is that a reduction policy in terms of PM_{10} is a general approach with complete coverage of all particles, which is also – or at least as – effective for $PM_{2.5}$, though it is more expensive than a similar cost-effective strategy focusing on $PM_{2.5}$. A cost-effective reduction policy formulated in terms of $PM_{2.5}$ is approximately 50% cheaper but results in much lower reductions of PM_{10}.

6.6. Modelling future levels of PM

6.6.1. Introduction

This Section presents the results of air quality modelling of the PM_{10} emission packages described in the previous paragraphs. Table 6.6 summarises the national totals of PM_{10} emissions used. The 2005-CLE (current legislation) variant is computed by linear interpolation between 1995 en 2010-CLE emissions. This does not account for the differentiated speed of penetration of abatement measures, which occurs in reality. Note that for all scenario calculations Dutch emissions of SIA precursors in 2010, as well as foreign emissions of primary aerosol and acidifying compounds, were kept constant. This assumption may be somewhat crude, because it does not facilitate the investigation of the potential effect of measures taken on a European scale on air quality in the Netherlands. Our assumptions, however, do allow us to explore the effectiveness of additional national abatement measures devoted to primary PM. The contribution of emissions from transport is addressed to further illustrate the potential impact of measures on concentration levels both in the rural and

urban environment. The air quality results were checked against the framework of the 2005 and 2010 limit values for PM_{10}.

Table 6.6 Summary of Dutch national total PM_{10} emissions (ktonnes) used in air quality calculations for a number of packages.

Scenario	PM_{10}
1980	116
1995	60
2005-CLE	46
2010-CLE	43
2010-$MFR_{real\ transport}$	40
$2010_{quart.\ red.}$	33
2010-MFR_{ult}	16

6.6.2. Results of current policies in 2010

The computations were performed using the OPS and EUROS/LOTOS models. These were introduced in subsection 2.5.3 along with the results of calculations for 1995. The country-averaged results of the PM_{10} concentrations for the 2010-CLE scenario are presented in Table 6.7. Remember that the transport sector includes road transport, ships and other mobile sources.

Table 6.7 Annually averaged primary and secondary inorganic concentrations of PM_{10} averaged over the Netherlands. Calculated for the year 2010-CLE.

Dutch sources	Primary PM_{10} ($\mu g/m^3$)	NH_x ($\mu g/m^3$)	NO_y ($\mu g/m^3$)	SO_x ($\mu g/m^3$)	Summed concentration ($\mu g/m^3$)
Industry	0.2	0.0	0.1	0.0	0.4
Energy	0.0	0.0	0.0	0.0	0.1
Transport	0.8	0.0	0.5	0.0	1.3
Agriculture	0.5	0.5	0.0	0.0	1.1
Others	0.5	0.1	0.1	0.0	0.6
Sum	**2.0**	**0.7**	**0.7**	**0.1**	**3.5**
Other countries					
Industry	0.7	0.0	0.0	0.0	0.8
Energy	0.3	0.0	0.4	1.0	1.8
Transport	0.9	0.0	1.1	0.3	2.3
Agriculture	0.1	1.0	0.0	0.0	1.1
Others	0.6	0.0	0.4	0.6	1.7
Sum	**2.7**	**1.0**	**2.0**	**1.9**	**7.6**
All sources					
Sum	**4.8**	**1.6**	**2.7**	**2.0**	**11.1**

The total sum of PM_{10} modelled in 2010 is 11.1 $\mu g/m^3$, which is composed of 3.5 $\mu g/m^3$ from national sources and 7.6 mg/m^3 from other countries. Approximately 5 $\mu g/m^3$ of the annual average consists of modelled primary particles and most of this results from emissions in other countries. About 6.5 $\mu g/m^3$ of PM_{10} consists of SIA, of which the contribution of nitrate is largest. Irrespective of the origin and composition

of the aerosol, the transport sector is the main contributor (3.6 $\mu g/m^3$) to the concentrations of PM_{10}.

Figure 6.7 presents the spatial pattern of PM_{10} modelled in 2010 using current policies. The map shows an increasing concentration gradient from the north to the south of the country. Some local maximums are observed in the Rotterdam area in the west and in the province of Limburg in the south. We added Figure 2.33 to this map in order to address the non-modelled part of PM_{10}. The result is shown in Figure 6.8. The average concentration in this map is 29.1 $\mu g/m^3$, of which 11.1 $\mu g/m^3$ is modelled and 18 $\mu g/m^3$ is the outcome of the difference map.

In Figure 6.8 about 0.5% of the surface of the Netherlands (i.e. 7 cells of 5 x 5 km^2) exceeds the 40 $\mu g/m^3$ annual limit value. The maximum concentration modelled is 43 $\mu g/m^3$. Most of these cells are located in industrial grids where the storage and handling of materials takes place. We are currently checking our emission inventories for the correctness of the storage and handling category. Initial indications from these checks result in lower emissions and probably lower concentrations in the above 7 cells. This activity will be reported later this year in a separate update of this report.

Apart from the storage and handling issue, the model indicates that the 2005 annual limit value of 40 $\mu g/m^3$ will most likely be met in the Netherlands in 2010. Interpolation to 2005-CLE results in the same conclusion: the annual limit value is likely to be realised. Nonetheless, our model calculations do not rule out the possibility of exceedances occurring *within* grid cells of 25 km^2. The daughter directive states that a spatial representativeness 'of several square kilometres' should be used to describe the urban-background situation. Although not very specific, the phrase 'several square kilometres' sounds somewhat less than 25 km^2, but it is in the same order of magnitude of the model resolution. For traffic locations a spatial representativeness of 'no less than 200 m^2' is aimed at. The OPS model does not resolve the latter size and this aspect needs further study. It should, however, be noted that particulate matter has a typical large-scale character and that concentration gradients at the level of 200 m^2 even near sources of traffic are much weaker than in the case of other shorter-lived traffic-related components like NO or UF.

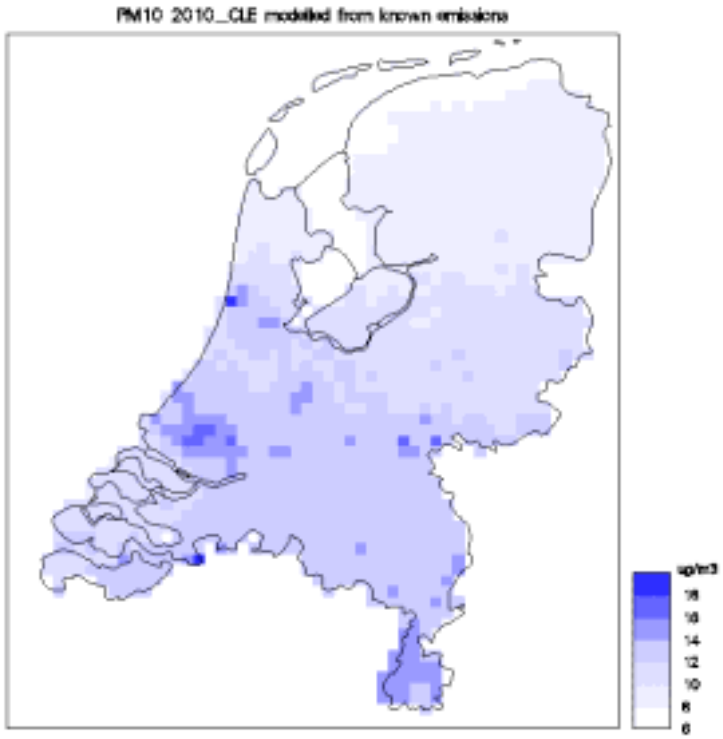


Figure 6.7 Modelled concentration field of PM_{10} using the 2010-CLE scenario. This field does not include the assessment of the non-modelled part of PM_{10}.

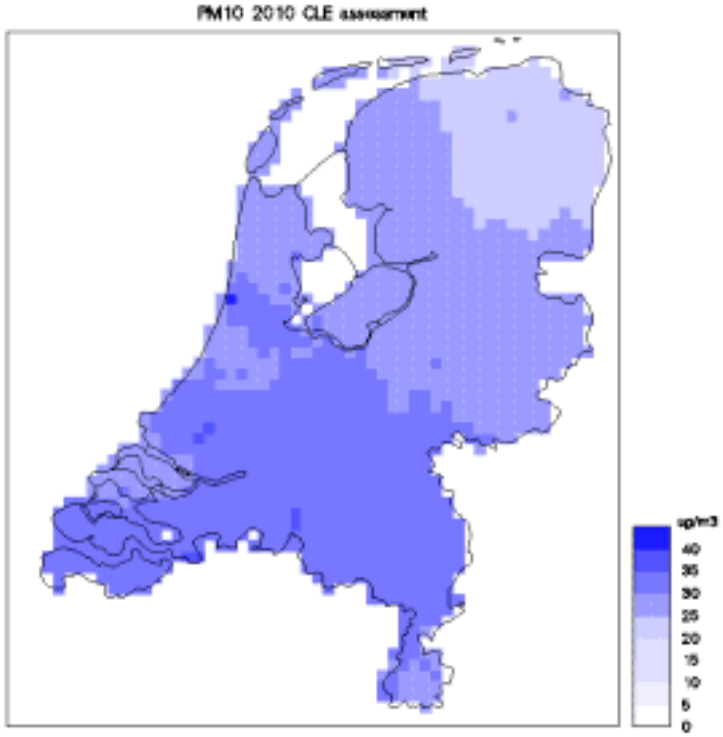


Figure 6.8 Assessment of the PM_{10} concentration field in 2010 using current legislation assumptions. This field includes the non-modelled part of PM_{10}.

In order to evaluate whether or not the 24-hour limit value for the protection of human health will be met in 2010, we used the annually averaged concentration of 30 $\mu g/m^3$. According to Figure 2.15, this annual average corresponds to about 35 days on which

the 24-hour value of 50 μg/m^3 is exceeded. Our 2010 assessment of current policies reveals that on 40% of the surface area of the Netherlands 30 μg/m^3 as an annual average is exceeded. This leads to the conclusion that the EU 24-hour limit value for 2005 will not be met in 2010. Linear interpolation to 2005 also shows it is improbable that the EU 24-hour standard will be met.

The 2010 indicative limit values will not be met either. Note that the assessment of the non-modelled part adds up to 18 μg/m^3. Under current assumptions this would leave only 2 μg/m^3 from anthropogenic sources to achieve compliance with an indicative standard of 20 μg/m^3.

A second source of information on the issue of realisation of the 2005 EU daily limit value in 2010 was produced by the EUROS/LOTOS models (see subsection 2.5.3.2) A 'difference–concentration-time' pattern from ten sites in the NAQMN was used to address the temporal behaviour of the non-modelled part of PM_{10}. The already presented Figure 2.29 confirms the above conclusions from the OPS model.

6.6.3. Modelled concentrations 1995 and 2010-CLE

A comparison with 1995 results shows that for the Netherlands the annual concentrations of modelled PM_{10} will decrease on average by 5.4 μg/m^3 from 16.5 to 11.1 μg/m^3. Of this decrease, 1.2 μg/m^3 is a result of reductions of primary PM_{10}, the greater part of which (0.9 μg/m^3) is due to abatement in the Netherlands, mainly in the transport sector.

With respect to 1995, international acidification policy reduces SIA concentrations by 4.1 μg/m^3. So, 75% of the downward trend between 1995 and 2010 is the outcome of reductions in emissions of the precursor gases SO_2, NO_x and NH_3. In total, nitrate concentrations decrease by almost 2 μg/m^3, followed by 1.6 μg/m^3 for sulphate. In particular, the abatement of acidifying species in foreign countries has an impact on concentrations in the Netherlands. Almost 75% of the modelled decrease of SIA emanates from acidification policies in these countries. This emphasises the importance of international collaboration in abating PM as a 'container' concept. Owing to the reduction in SIA, the aerosol becomes relatively 'blacker' and will be linked more closely to emissions from combustion processes.

A relatively constant fraction of 30–35% of the modelled PM_{10} stems from Dutch emissions in both 1995 and 2010-CLE despite the major reductions of SIA following cuts in foreign countries. Figure 6.10 illustrates the changes in the modelled composition of PM_{10} as an average over the Netherlands. The 18 μg/m^3 'not modelled' bar represents the average of the difference map explained in subsection 2.6.3. The decrease of SIA from emissions in foreign countries is clearly visible in Figure 6.9.

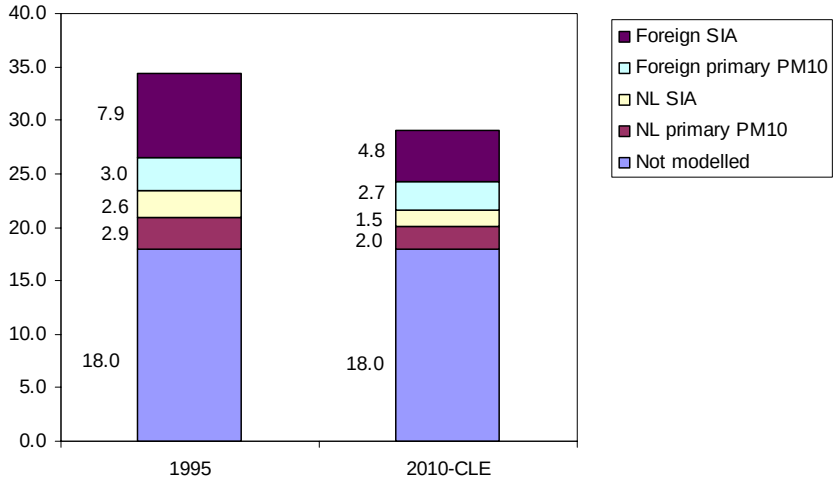


Figure 6.9 Changes in the composition of modelled PM_{10} between 1995 and 2010-CLE.

The variability of the modelled composition of PM_{10} at two typical locations is illustrated in Figure 6.10. Valthermond is a rural location in the northeast of the country and Rotterdam-Schiedamsevest is an urban background location in the densely populated west, situated close to the world's largest port. Both are currently station locations in the NAQMN. Obviously, the foreign contributions to SIA and primary PM_{10} are quite similar at these sites. However, the input from Dutch sources, in particular of primary PM_{10}, is much larger at the urban site than at the rural site. This figure shows the maximum potential of abatement measures on a national scale. To what extent the Dutch contribution to urban PM_{10} can be reduced will be addressed in the section on additional policy options (6.6.4.).

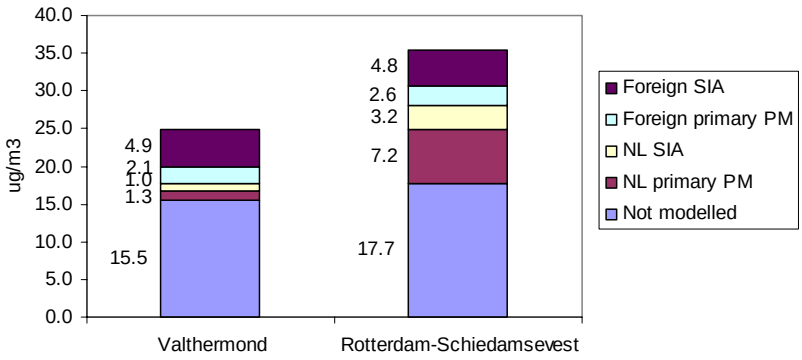


Figure 6.10 Modelled composition of PM_{10} at two locations in the Netherlands in 2010-CLE. Valthermond is a rural location in the northeast of the country, whereas Rotterdam-Schiedamsevest represents a polluted urban site.

A comparison with calculations for 1995 (not presented) leads to the conclusion that the Dutch contribution to PM_{10} at the more rural and more densely populated urban sites will be reduced by 30–35% between 1995 and 2010. The absolute reduction from Dutch sources in this period is modelled at almost 5 $\mu g/m^3$ at Rotterdam-Schiedamsevest. The decline in Dutch primary PM_{10} is assessed at 3.3 $\mu g/m^3$.

6.6.4. Effects of additional policy measures

The effects of three extra policy packages on air quality in the Netherlands were examined in addition to the current legislation scenario. The technical details of these packages are explained in Sections 6.3 and 6.4. The abatement packages studied focus on primary PM in the Netherlands only. Emissions of primary PM in foreign countries and precursors of SIA in the Netherlands and abroad were kept constant. Table 6.8 summarises the results of these calculations.

Table 6.8 Country-averaged results of air quality calculations using the 2010 scenario variants. In order to facilitate the comparison with calculations from past years, data from 1980 and 1995 are listed as well. The non-modelled fraction of 18 μg/m^3 is not shown in this table

Year	Modelled average PM_{10} concentration (μg/m^3)	Modelled average concentration of primary PM_{10} (μg/m^3)	Primary PM_{10} concentration index compared with 2010-CLE (%)	Primary PM_{10} concentration index from Dutch sources compared with 2010-CLE (%)	Additional costs per year compared with 2010 CLE (M)
1980	29.7	11.4	240	225	
1995	16.5	6.0	125	145	
2005-CLE	12.9	5.2	108	115	
2010-CLE	11.1	4.8	100	100	
$2010_{quart\ red}$	10.8	4.5	95	90	210[a]
$2010\text{-}MFR_{real\ transp}$	11.0	4.6	97	95	1000[a]
$2010\text{-}MFR_{ult}$	10.0	3.6	76	45	6000[a]

CLE = Current Legislation decided on before 1 January 2000 according to the Global Competition C scenario

$2010_{quart\ red}$ = CLE with an extra reduction of approx. a quarter, marginal costs of up to 55 euro/kg PM_{10} reduction.

MFR_{ult} = ultimate Maximum technically Feasible Reductions

[a] Excluding costs for technical measures in seagoing ships

Table 6.8 shows that the three abatement packages in addition to current policies are expected to lead to a decrease of at most 1 μg/m^3 averaged over the Netherlands even when exceptional cost is incurred. The primary PM_{10} concentration index reveals a larger downward effect. When Dutch and foreign sources are considered, this index can be reduced to 76% compared with current legislation when the MFR_{ult} variant is applied, and to 95–97% with the other two scenarios. The primary contribution from Dutch sources is expected to be more than halved in the MFR_{ult} variant, whereas the '$2010_{quart\ red}$' variant cuts this parameter by 10%.

In Figure 6.11 the spatial pattern of PM_{10} from the 2010 MFR_{ult} package is shown. The conclusions that can be drawn from this map are similar to those from the 2010-CLE field with respect to the limit values for particulate matter. Compliance with the annual limit value of 40 μg/m^3 is expected everywhere in the Netherlands. No cells exceed 40 μg/m^3. Linear interpolation to the 2005 MFR_{ult} field shows 1 cell where a value of 42 μg/m^3 is calculated. The 24-hour limit value (50 μg/m^3 with a maximum

of 35 exceedances) will not be met. The area where this limit value is exceeded is, however, smaller than in the 2010 current policies scenario. Realisation of the 2010 indicative limit values is out of sight even in this expensive package.

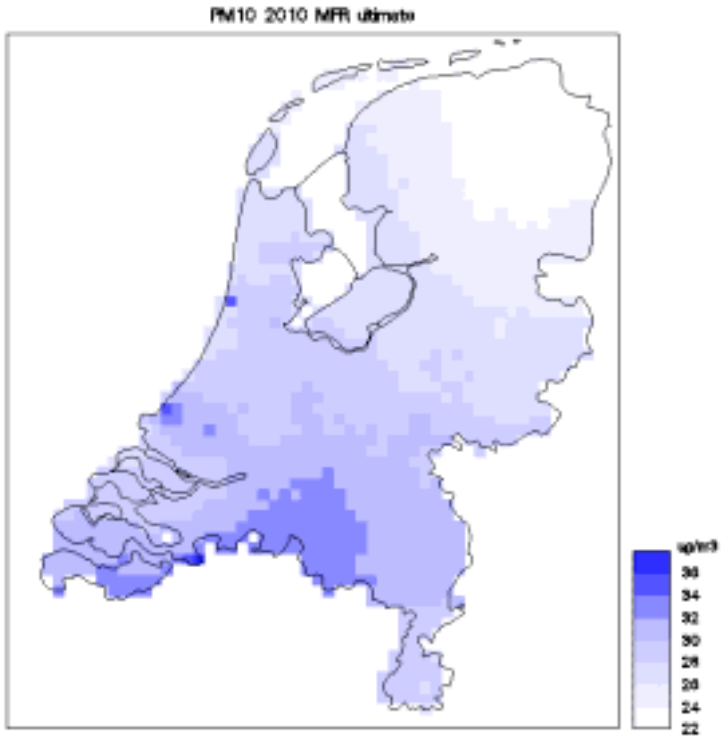


Figure 6.11 Annually averaged PM_{10} concentration using the 2010 $MFR_{ultimate}$ emission abatement package

6.6.5. Spatial effects of additional policy measures on primary PM_{10}

In many places the local benefits of the additional abatement calculations are somewhat larger than the data given in Table 6.8. This is illustrated for the Rotterdam-Schiedamsevest location in Figure 6.12. This site is affected by local emissions of primary PM_{10} from transport and industrial sources. The Figure shows that the concentration of primary PM_{10} decreases by 0.5–3.5 µg/m^3, depending on the package. In particular, the 'MFR $_{real\ transp}$' and '$2010_{quart\ red}$' variants are efficient in reducing primary PM_{10} at this location relative to their total emission reductions. This is because these packages put the emphasis on emission reductions in the transport (both packages) and industry ('$2010_{quart\ red}$' variant only) sectors. At this location the '$2010_{quart\ red}$' variant produces the largest reduction per euro spent.

Table 6.9 summarises the same issue for 6 agglomerations in the Netherlands. This classification of agglomerations is used in the EU framework and daughter directive to report air quality to the European Commission. The table shows that in the densely populated areas in the Netherlands concentrations of primary PM_{10} are expected to decrease by 7–17% under the '$2010_{quart\ red}$' assumptions compared with current policies.

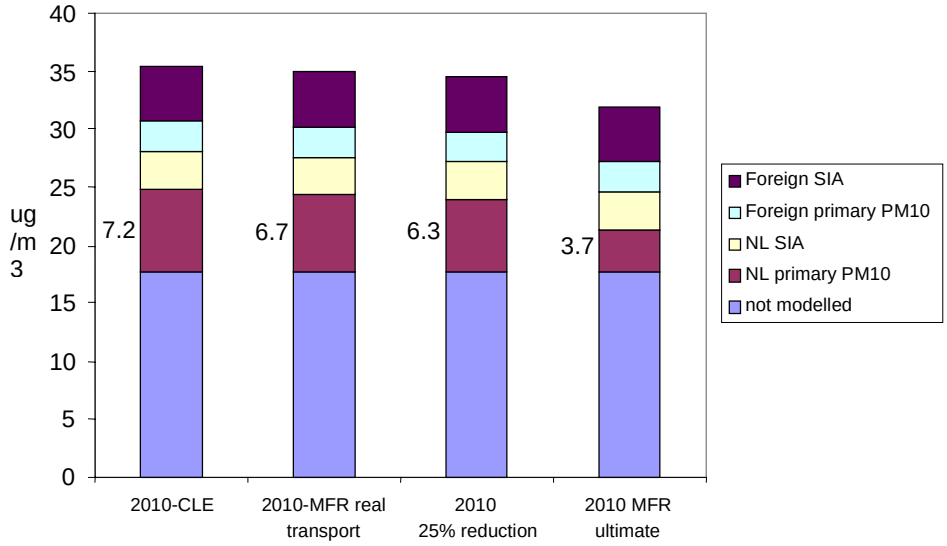


Figure 6.12 Origin of PM_{10} modelled at the Rotterdam-Schiedamsevest location in four 2010 emission variants.

Table 6.9 The modelled concentration of primary PM_{10} from Dutch emissions in 6 agglomerations in the Netherlands under current policies and '$2010_{quart\ red}$' variants.

	Amsterdam-Haarlem	Rotterdam-Dordrecht	The Hague-Leiden	Utrecht	Eindhoven	Heerlen-Kerkrade
2010-CLE	4.5	4.9	3.9	3.9	2.7	2.1
$2010_{quart\ red}$	3.7	4.1	3.4	3.5	2.5	2.0
% reduction	17	17	13	11	8	7

Table 6.10 presents the primary contribution from Dutch sources of *transport* in six agglomerations. In addition to Table 6.9, the data reveal that the transport sector can contribute up to 50–55% of total primary PM_{10} of Dutch origin. Table 6.10 also shows that the cost-effective '$2010_{quart\ red}$' package can reduce the transport fraction by another 10–25% in addition to current policies. Note that the transport policy measures assumed in the '$2010_{quart\ red}$' variant are concentrated in the shipping sector (Table 6.4).

Table 6.10 The modelled concentration of primary PM_{10} from Dutch transport emissions in 6 agglomerations in the Netherlands in the current policies and '$2010_{quart\ red}$' reduction variants.

	Amsterdam-Haarlem	Rotterdam-Dordrecht	The Hague-Leiden	Utrecht	Eindhoven	Heerlen-Kerkrade
2010-CLE	1.4	2.7	1.5	1.8	1.0	0.7
$2010_{quart\ red}$	1.2	2.1	1.3	1.6	0.9	0.6
% reduction	16	23	17	14	13	12

Although the causal factors and biological mechanisms behind the PM-associated health effects are unclear and although the issue of PM_{10} is a typically large-scale phenomenon, we have shown that abatement of primary PM_{10} emissions from Dutch sources can help to achieve significant reductions in primary PM_{10} in Dutch agglomerations. In particular, relatively cheap measures in the shipping sector bring

about a meaningful reduction in primary aerosol in densely populated areas. Since these diesel soot reductions fall in the category of 'prime suspects' in the health effects domain, these measures can be labelled as no-regret options.

6.7. Summary and conclusions

Under current policies, emissions of PM_{10} will diminish by 20% between 1998 and 2010. Emission reductions in other health-related PM indicators will improve even more; emissions of $PM_{2.5}$ will decrease by 30%, total combustion-related emissions by 45% and transport-related exhaust emissions by 55%. Note that $PM_{2.5}$ emissions are related more to combustion processes and less to mechanical processes than general PM_{10} emissions.

The successful abatement of the classical PM sources, e.g. point sources in the energy sector and in industry, waste incineration and traffic exhaust, makes other currently less controlled sources more important. These less controlled sources can be found in industry (point sources with low emission flows, building ventilation, storage and handling), transport (ships, mobile machines, wear of tyres, road and brakes), agriculture (animal housing systems, tilling and harvesting), consumers (wood stoves and fireplaces), construction (construction sites) and the commercial and institutional sector (storage and handling of materials).

Current knowledge on health effects presented in other chapters shows that despite decreasing emissions and despite the expected future compliance with the PM_{10} annual standard of 40 $\mu g/m^3$, serious adverse PM-associated health effects can still be expected in the year 2010. This warrants supplementary source-oriented abatement action by the Dutch government and other interested parties, such as the EU. However, a problem with identifying and proposing cost-effective solutions for the further abatement of PM_{10} emissions is that probably not all PM_{10} fractions are equally health-relevant. Current knowledge on health effects leads to the conclusion that PM_{10} has to be seen as a 'container', for which no fractions can be completely ruled out, but for which some fractions (probably more primary and combustion-related) appear to be more health-relevant than others (like sea salt, SIA and, possibly, non-crystalline crustal material).

Because of the health relevance of primary emissions and because the PM_{10} annual standard will probably be met in the Netherlands, the technical study on abatement measures focused on *primary* PM_{10} emissions. No study was made of the technical possibilities for and cost of further reducing precursor gases of secondary aerosols, i.e. SO_2, NO_x, NH_3 and VOC. For atmospheric modelling up to the year 2010 it was assumed that the Netherlands would comply with the national emission ceilings as agreed within the EU.

The results of the study of PM emission control show there is no technological constraint for further reducing PM_{10} emissions in the Netherlands on top of currently agreed measures. Applying best available technology will reduce emissions by about 60%. However, the cost is about 6000 million euro per year. A more realistic cost-effective reduction of 10 ktonnes, i.e. a proportional reduction of 25%, can be achieved at a cost of 210 million euro per year. These cost-effective reductions can be achieved in industry (6.0 ktonnes), the shipping sector (2.5 ktonnes) and storage and

handling companies (2.0 ktonnes). Cost-effective measures in industry are found at refineries (2 ktonnes), in the food industry (1.8 ktonnes), building materials (0.9 ktonnes) and chemical industry (0.6 ktonnes). Measures for the basic metals industry are also cost-effective, but these have already been included under current policies. Measures in other industrial sectors not counted among this group of five high-emission industries are relatively more expensive.

The analysis of what is needed in terms of abatement measures to achieve a further cost-effective reduction of *primary* PM_{10} emissions produces:
- strong support for the reduction of the S-content of residual oil used by seagoing ships;
- strong support for the application of technical measures (optimised engines, particulate traps) that will reduce uncontrolled emissions from diesel engines on inland and seagoing ships;
- strong support for the implementation of additional technical measures in the already controlled classical high-emission industries, and in companies specialising in materials handling.

The first abatement option requires Annex VI of the MARPOL convention to come into force (from 3% to 1.5%). In a next step, international agreements need to be made on a further reduction (from 1.5% to 0.5%). The second option demands stringent Stage-2 PM emission limits for inland ships (e.g. Euro4/5 limits for Heavy Duty Vehicles), while PM emission limits for seagoing ships should also be agreed. With respect to the third option for industrial sources, two conclusions can be drawn. In the first place, reinforcing generic national PM emission standards will not be the most cost-effective option for achieving reductions in industry. Other policy instruments are needed to achieve this kind of reduction. Secondly, a cost-effective reduction policy for PM_{10} will result in a probably even larger reduction of $PM_{2.5}$ emissions.

The concentration of PM_{10} decreased by about 13 $\mu g/m^3$ between 1980 and 1995 and is expected to decrease by a further 5.5 $\mu g/m^3$ between 1995 and 2010.
In this decade, European acidification policy will be particularly efficient in abating PM_{10}. We forecast that three-quarters of the decrease will be linked to reductions in emissions of SO_2, NO_x and NH_3, predominantly in countries other than the Netherlands. The future aerosol will be more combustion-related and will become 'blacker' as a result of the reduction in SIA (the 'white' salty part). The transport sector will be the main contributor to ambient concentrations of PM_{10} in 2010.

The annual limit value of 40 $\mu g/m^3$ will most likely be realised in 2005 in the Netherlands, whereas the daily standard of 50 $\mu g/m^3$ with 35 permitted exceedances will not be met. A few exceedances of 40 $\mu g/m^3$ continue to be modelled in our 2010 assessment. Most of these are related to the 'storage and handling' emission category. We are currently conducting a detailed check of our emission inventory with respect to this issue. Exceedances may also continue to occur on a smaller spatial scale that is not resolved by the model. The daughter directive gives indications of the spatial representativeness of measurement sites for urban background and traffic locations.

Concentrations of primary PM_{10} are a typically large-scale phenomenon, similar to SIA, for which the importance of European collaboration in abating emissions should

be emphasised. Averaged over the Netherlands, about 55% of our primary PM_{10} will originate from foreign countries in 2010. Additional Dutch mitigation measures will therefore only marginally reduce (1 $\mu g/m^3$) the PM_{10} concentration *averaged over the country*. However, primary PM_{10} is expected to decrease by 5–25%, depending on the package chosen. In densely populated areas close to sources of industry and transport, somewhat larger concentration decreases of primary PM_{10} are computed. The realistic '$2010_{quart\ red}$' package focuses on emissions from industry, shipping, and storage and handling. This package leads to a reduction of more than 10% in primary PM_{10} stemming from Dutch sources in the city of Rotterdam. The contribution from Dutch *transport* emissions is reduced somewhat further: 10–25% in Dutch agglomerations. It is interesting to note that the measures directed at *transport* in the '$2010_{quart\ red}$' package are devoted solely to the shipping sector. The cost of implementing these measures is estimated at about 10 million euro per year. These findings underpin the fact that cost-effective measures in the Netherlands will be useful in significantly reducing a PM fraction (i.e. diesel soot) that is suspected of being of relevance to health effects.

Proposed abatement measures for bringing down *primary* PM_{10} emissions are fully justified if the aim is to bring down PM_{10} concentration levels independent of the underlying fractions that are thus controlled. However, considering the lack of knowledge concerning the causal factors and the biological mechanisms behind the PM-associated health effects, the question arises of whether these additional measures can be justified on the precautionary principle. However, costly mistakes should be avoided as far as possible. In this context it is worth remembering that scientific knowledge indicates that, in general, primary combustion-related emissions, and more in particular traffic-related diesel soot emissions, are probably health-relevant. In this respect, one cost-effective option deserves particular attention, i.e. the abatement of uncontrolled diesel-engine emissions from inland and seagoing ships.

All in all, this leads to the conclusion that, although the Netherlands is likely to meet the EU annual PM standard of 40 $\mu g/m^3$, there are arguments supporting vigorous additional national and international source-based policy action aimed at the abatement of uncontrolled emissions from inland and seagoing ships.

7. Conclusions

7.1. Summarised Conclusions

Epidemiological studies present worldwide evidence for particulate matter (PM) associated health effects in the general population. Dutch observations are in line with the international scientific literature. In spite of the ongoing scientific debate and prevailing uncertainties concerning the quantification of acute and chronic health effects, the overall conclusion is that PM-associated health effects are so extensive and serious that further action is warranted.

Epidemiological studies could not identify a threshold for exposure levels related to PM health effects. This precludes regular standard setting, with a No Observed Adverse Effect Level (NOAEL) and safety factors. It implies that for any PM standard a certain level of impact on health will have to be accepted.

Because there is no threshold, adverse health effects are less effectively avoided by reducing episodic high concentrations than by reducing annual average concentrations (which will reduce the magnitude of occasional peak concentrations as well).

Overall, the health effects are consistently associated with PM_{10} and $PM_{2.5}$, in spite of the local differences in air quality, sources and the proportion of the susceptible sub-population. However, there seems to be heterogeneity between locations within the various epidemiological time-series studies for PM-associated health effects. This heterogeneity is manifested in differences in the size of the effects and may probably be influenced by local ambient and population-related circumstances. As for the future, a gradually ageing population and an increasing proportion of asthmatics or people with circulatory problems will proportionally enlarge the potentially susceptible sub-population.

Support is emerging to supplement the current PM standard with other (smaller sized or source-related) indicators than PM_{10}. There is currently a lack of reliable information on ambient levels of these smaller sized or source-related fractions which is representative of the situation in the Netherlands. At the moment the available toxicological and epidemiological evidence is insufficient for regulating ultrafine (UF) particulate concentrations, though this is another field requiring further research as the potential health implications of UF may be considerable. It is recommended that PM_{10} be retained as a standard for the time being as it covers the effects of both fine and coarse particles. In view of the emerging evidence implicating fine particles in health effects, it is recommended that a standard be developed for fine PM or a source-related fraction as well.

The levels of PM_{10} measured by a stationary site monitor seem to be representative of the personal exposure of the general public to ambient PM_{10}. Accurate measurement of PM is complicated and the automatic PM_{10} monitoring network corrects for losses of semi-volatile material in the Netherlands by using a factor of 1.3, as an approximation. The accuracy of PM measurements should be increased, as semi-volatile ammonium nitrate is a principal component of PM in the Netherlands. More information on the specific chemical composition and size distribution of PM

representative for typical situations in the Netherlands is needed to test relevant hypotheses concerning health effects, source contributions and possibly atmospheric influences.

Current policies will reduce emissions of PM_{10} by about 20% from 1998 to 2010. The fraction of PM_{10} that is combustion-related and suspected of being health-relevant will show an even larger reduction of 40%, based on projections of energy use, transportation developments and performance of new technology under real-world conditions. Dutch emissions of $PM_{2.5}$ will decrease by about 30%. Traffic is an important source of carbonaceous PM (which can be broken down into Elemental Carbon (EC) and Organic Carbon (OC), the mixture of which comprises diesel soot) and ultrafines, which are emitted at breathing height, close to a large part of the population in the Netherlands.

Whether a reduction in PM levels leads to a proportional reduction in health effects is still uncertain. PM is a complex mixture with fractions that are to a greater or lesser extent health-relevant. Changes in the composition of this mixture might change the health impact. So, the most cost-effective policy will be to reduce that part of PM that causes the health problems. Unfortunately, there are currently only suggestions for the causal fractions as they have not yet been identified.

Certain fractions of ambient PM probably do not cause significant health effects. These include particle-bound water and probably sea salt particles. A number of epidemiological studies suggest that the crustal fraction is less health-relevant than combustion-related fractions. Toxicological studies with pure ammonium sulphate and nitrate (Secondary Inorganic Aerosol, SIA) have not established any overt toxicity of these components, even at concentrations considerably above ambient levels. By contrast, epidemiological studies continue to find strong associations between adverse health effects and secondary aerosol components such as sulphates and nitrates. This divergence of results has not yet been resolved.

The EU's original position paper envisaged promulgating two equivalent PM standards. However, the current EU annual average standard (40 $\mu g/m^3$) and daily standard (50 $\mu g/m^3$, 35 exceedances) for 2005 are not equivalent in the Netherlands. With **80** permitted exceedances per year, a daily average of 50 $\mu g/m^3$ would be equivalent to an annual average of 40 $\mu g/m^3$, like a daily standard of 100 $\mu g/m^3$ PM_{10} with 7 exceedances. For practical reasons a standard with a value of 100 $\mu g/m^3$ and 7 exceedances is preferred to a value of 50 $\mu g/m^3$ that may be exceeded on 80 days. In general, the public is able to comprehend a standard with a small number of exceedances better. Although the EU has proposed two standards for PM, there are several arguments that only one standard would suffice – annual mean concentrations being the best choice. However, for reasons of communication to the public and preventing exceedances above a certain threshold, daily standards may be appropriate. Whether or not two averaging times are needed for an EU PM standard is a policy decision.

Annual averaged values in the Netherlands obtained by modelling are consistent with measurements of PM_{10} here. Compliance with the annual average value of 40 $\mu g/m^3$ seems feasible for PM_{10} in the Netherlands, although local exceedances at 'hot spots' cannot be ruled out. However, compliance with the daily average value of 50 $\mu g/m^3$

with 35 permitted exceedances is probably not feasible in 2005. Because of the relatively large contribution of foreign PM in a small country like the Netherlands, combined with our substantial natural background levels caused by sea salt, crustal and other natural material, a daily level of 50 $\mu g/m^3$ will easily be exceeded. Expectations are that there will still be 36–40 exceedances per year of the EU daily standard of 50 $\mu g/m^3$ even after all planned abatement measures (Current Legislation of Emissions, CLE) have been taken in 2010. Compliance with the indicative annual average value of 20 $\mu g/m^3$ for PM_{10} and compliance with the indicative daily average value of 50 $\mu g/m^3$ with 7 permitted exceedances in 2010 in the Netherlands is not possible. If abatement measures are implemented in neighbouring countries (as may be expected), the number of exceedances will decrease.

The ultimate potential for reducing primary PM_{10} emissions (on top of currently agreed measures, CLE) could be 60% in the Netherlands. This abatement package is called 'MFR_{ult}': ultimate Maximum Feasible Reduction. The cost of achieving the 'MFR_{ult}' is about 6000 million euro per year. The 'MFR_{ult}' reduction of 60% in primary PM_{10} emissions in the Netherlands will result in a 1.1 $\mu g/m^3$ lower PM_{10} concentration averaged over the country. An emission reduction up to a cost-efficiency of 55 euro/kg PM_{10} will lead to a reduction by a quarter (abatement package '$2010_{quart\ red}$'). This can be achieved at a cost of 210 million euro per year and will result on average in a 0.3 $\mu g/m^3$ lower PM_{10} concentration. From the absolute value of the PM_{10} levels one can conclude that, averaged on a national level, these reductions seem rather small. Locally, however, higher reductions in PM_{10} levels of 1–5.5 $\mu g/m^3$ are modelled in the '$2010_{quart\ red}$' abatement package. The maximum reductions will be achieved in Rotterdam, which is densely populated. It is interesting to note that the measures directed at *transport* in the '$2010_{quart\ red}$' abatement package focus on the shipping sector only. When concentrating on probably more health-relevant fractions of PM like traffic-related diesel soot, modelled reductions are relatively higher even. The presented abatement packages ('$2010_{quart\ red}$' and 'MFR_{ult}') correspond to a decrease of 20% and 50% respectively in average traffic-related diesel soot concentration levels of Dutch origin. These effects would increase even further if similar reduction technologies were to be applied to traffic in foreign countries also.

Supplementary PM abatement can be based on the precautionary principle. Further source-oriented actions could focus on the more cost-effective reductions of the total PM_{10} aerosol mass, or could first of all focus on those PM fractions that are expected to be more health-relevant. This last option is preferred. These fractions are probably transport-related (diesel soot) and, more generally, combustion-related primary PM emissions. In this respect, the abatement of uncontrolled shipping emissions has been identified as one of the more cost-effective control options. The abatement of other combustion-related sources such as industrial combustion, wood burning in fireplaces and off-road machinery is also possible, but is less cost-effective. Additionally, climate change mitigation strategies may reduce combustion-related PM emissions.

A substantial part of the PM_{10} levels in the Netherlands cannot be influenced by policy measures, as natural sources are responsible for their ambient concentrations. Because future abatement measures will further reduce the anthropogenic fraction, the contribution of the natural fraction will increase proportionally. More insight into the chemical composition (specific tracers) and contribution of different sources to the currently 'non-modelled' and generally natural part of PM_{10} is necessary to find out

how much of the current PM levels may eventually be influenced by abatement measures.

PM air quality will improve in the future. Despite the air quality, it could be conjectured that the health impact associated with PM will nevertheless become more pronounced. In the Netherlands the gradual ageing of the population and other demographic developments could lead to a more than proportionate rise in the susceptible sub-groups. However speculative the previous remark, continuing vigilance seems required for this only partially understood problem of PM.

7.2. Current answers to the questions

The following is an attempt to answer the five questions from the Environment Ministry (VROM) presented in the introduction. These answers are called current because although they represent the 'state of the art', they do not yet present a scientifically satisfying answer. They are based on the currently available and sometimes scarce or contradicting information. A number of previous problems have been solved in the past few years, while sometimes new enigmas have arisen.

1. How do the various indicators of PM compare as relevant for the causation of health effects?

The various indicators of PM (PM_{10}, $PM_{2.5}$ or BS) are all indicators of the observed health effects. All of the above indicators can be used as a proxy or a surrogate for the currently unknown causal factor. In epidemiological research finer particles seem to be more relevant for health effects than coarser particles. Whether the various indicators compare differently with regard to the causation of health effects in the Netherlands is a question that cannot yet be answered.

2. What is the relationship between concentrations of ambient PM and health effects in order to make a substantiated choice of PM standard?

Epidemiological studies could not identify a threshold for exposure levels related to PM health effects. This precludes regular standard setting with a No Observed Adverse Effect Level (NOAEL) and safety factors. It implies that for any PM standard a certain level of impact on health will have to be accepted.

Because there is no threshold, adverse health effects are less effectively avoided by reducing episodic high concentrations than by reducing annual average concentrations (which will reduce the magnitude of occasional peak concentrations as well).

Overall, the health effects are consistently associated with PM_{10} and $PM_{2.5}$, in spite of the local differences in air quality, sources and the proportion of the susceptible sub-population. However, there seems to be heterogeneity between locations within the various epidemiological time-series studies for PM-associated health effects. This heterogeneity is manifested in differences in the size of the effects and may probably be influenced by local ambient and population-related circumstances. As for the future, a gradually ageing population and an increasing proportion of asthmatics or

people with circulatory problems will proportionally enlarge the potentially susceptible sub-population.

Support is emerging to supplement the current PM standard with other (smaller sized or source-related) indicators than PM_{10}. There is currently a lack of reliable information on ambient levels of these smaller sized or source-related fractions which is representative of the situation in the Netherlands. At the moment the available toxicological and epidemiological evidence is insufficient for regulating ultrafine (UF) particulate concentrations, though this is another field requiring further research as the potential health implications of UF may be considerable. It is recommended that PM_{10} be retained as a standard for the time being as it covers effects of both fine and coarse particles. In view of the emerging evidence implicating fine particles in health effects, it is recommended that a standard be developed for fine PM or a source-related fraction as well.

3. What are the actual PM concentrations in the Netherlands and how big are the contributions of the different source categories to these concentrations?

In 2001 the annual average measured PM_{10} concentration at the regional sites in the Netherlands was 31 $\mu g/m^3$. Only a few urban and industrial grids of the 1700 (5 x 5 km^2) grids are estimated to have long-term average PM_{10} concentrations in excess of 40 $\mu g/m^3$. The long-term PM situation seems favourable, but the 24-hour averages are a different story. The EU daily standard of 50 $\mu g/m^3$ is still exceeded in most of the Netherlands quite a number of times above the limit of 35 exceedances fixed by the EU for 2005. It is clear that the EU daily and annual standards for PM_{10} are not equivalent in the Netherlands. It seems that although the annual average standard of 40 $\mu g/m^3$ PM_{10} can probably be met in 2005, compliance with the daily standard is a totally different question.

The contribution to PM_{10} levels of various natural and anthropogenic sources have been described in detail in a report by Visser *et al.* (2001). In order to present a broad picture, one of their figures is presented as Figure 7.1. This figure indicates which part of the PM_{10} concentrations at six different locations in the Netherlands can be influenced most by abatement measures within the Netherlands (orange) or by European (blue) abatement measures. The natural sources are shown in green.

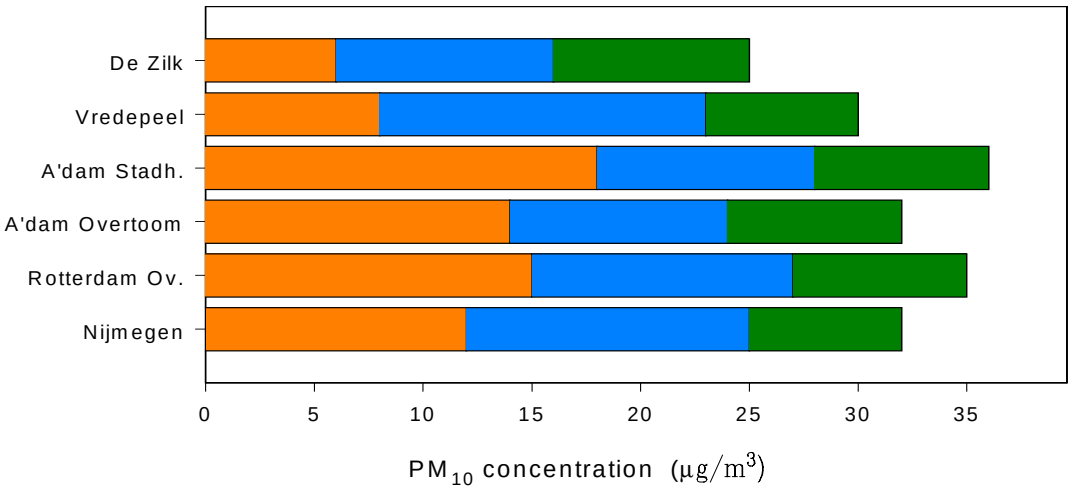


Figure 7.1 Maximum part of PM_{10} that can potentially be influenced by abatement measures in the Netherlands (orange) and the rest of Europe (blue), and contribution by natural sources (green). Data from Visser et al., 2001.

Figure 7.1 is based on a study for which the sampling was carried out in 1999. During 2000 and 2001 TNO and ECN took supplementary measurements in the NAP project, which confirmed this picture. ECN established by measurements taken at Cabauw (in the centre of the Netherlands) at heights of 20 and 200 m that the annual average foreign contribution of PM_{10} to Dutch levels is in the range of 7–17 µg/m^3. This ties in nicely with a modelled long-term contribution of 10 µg/m^3 as the magnitude of the foreign contribution, based on known emissions. Recent measurements in the Rotterdam area taken by TNO indicate that the average local contribution of a highway was 1.7 µg/m^3 EC. The measured average urban background of EC in Rotterdam was 1.4 µg/m^3. Reasonably similar urban background measurements taken by RIVM in a previous year came to 1.6 µg/m^3, indicating very good agreement. Tunnel measurements in Amsterdam have shown that the magnitude of the contribution of OC by traffic is similar to that of EC. At an urban background site the average EC/OC level would therefore be approximately 3 µg/m^3. This value ties in nicely with the long-term modelled primary traffic concentration of 3.5 µg/m^3 in the corresponding urban background grid. For highways, tunnel measurements taken in Rotterdam by TNO established a high contribution to the particle numbers, and an emission factor of 10^{14} UF particles per vehicle kilometre is reported. Annual average PM_{10} concentrations in specific street canyons in a major city show that, locally, levels in excess of 40 µg/m^3 cannot be excluded. This indicates that there may still be local ‘hot spots’ of PM_{10}.

4. Which indicator of PM is preferable if, besides health relevance, risk management considerations are also taken into account?

In epidemiological research finer particles seem to be more relevant for health effects than coarser particles. Whether the various indicators compare differently with regard to the causation of health effects is a question that cannot yet be answered. The sometimes reported associations between UF and health effects warrant further research in order to arrive at a clearer picture. In the Netherlands the correlation between UF and other PM metrics seems poor, which may suggest that an additional separate effect of UF may exist, with possibly other mechanisms of action. However, current toxicological, human-clinical and epidemiological information on UF is insufficient to base a standard on.

Nevertheless, arguments are emerging that standards in the smaller ranges of PM, but not as small as UF, might be useful. A number of arguments support a $PM_{2.5}$ standard as there are numerous scientific papers in which associations between $PM_{2.5}$ and health effects have been reported. Arguments of applicability call more for putting the future cut-off at 1 µm, because the crustal fraction will then be eliminated from the PM mixture. For southern European countries affected by Saharan dust this would be an argument to consider. However, less information is available on PM_1 at present.

In the future, other PM indicators will probably be appropriate to supplement or to replace the current PM standard. It is recommended that a size or source-related standard for fine, accumulation mode particles be developed using a cut-off in the 1–2.5 µm range in view of the large body of evidence that has accumulated concerning adverse health effects of fine PM. This standard should supplement rather than replace the PM_{10} standard in view of the indications of adverse effects of the coarse PM fraction. It is therefore recommended that PM_{10} also be retained as a standard for the time being. Research is needed to establish whether other standards also need to be developed, e.g. for the ultrafine particle fraction.

Apart from crustal material and SIA, the toxicological database on PM health effects is not yet at a stage where it can provide clear answers regarding causal factors, whether using a physical (surface area, charge, radiation), chemical (e.g. transition metals, organic substances), or biological (viruses, moulds, spores, bacteria or products of bacteria, e.g. endotoxins) entity. Quite a large amount of specific toxicological evidence seems to correspond with the results of epidemiological studies. An understanding of what in this complex mixture determines its toxicity would be of assistance in PM monitoring and control.

5. What is the quantification of the total source-risk chain now and in the foreseeable future?

Current policies will reduce emissions of PM_{10} by about 20% from 1998 to 2010. The fraction of PM_{10} that is combustion-related and suspected of being health-relevant will show an even larger reduction of 40%, based on projections of energy use, transportation developments and performance of new technology under real-world conditions. Dutch emissions of $PM_{2.5}$ will decrease by about 30%. Traffic is an important source of carbonaceous PM (which can be broken down into Elemental

Carbon (EC) and Organic Carbon (OC), the mixture of which comprises diesel soot) and ultrafines, which are emitted at breathing height, close to a large part of the population in the Netherlands.

Whether a reduction in PM levels leads to a proportional reduction in health effects is still uncertain. PM is a complex mixture with fractions that are to a greater or lesser extent health-relevant. Changes in the composition of this mixture might change the health impact. So, the most cost-effective policy will be to reduce that part of PM that causes the health problems. Unfortunately, there are currently only suggestions for the causal fractions as they have not yet been identified.

Certain fractions of ambient PM probably do not cause significant health effects. These include particle-bound water and probably sea salt particles. A number of epidemiological studies suggest that the crustal fraction is less health-relevant than combustion-related fractions. Toxicological studies with pure ammonium sulphate and nitrate (Secondary Inorganic Aerosol, SIA) have not established any overt toxicity of these components, even at concentrations considerably above ambient levels. By contrast, epidemiological studies continue to find strong associations between adverse health effects and secondary aerosol components such as sulphates and nitrates. This divergence of results has not yet been resolved.

Supplementary PM abatement can be based on the precautionary principle. Further source-oriented actions could focus on the more cost-effective reductions of the total PM_{10} aerosol mass, or could first of all focus on those PM fractions that are expected to be more health-relevant. This last option is preferred. These fractions are probably transport-related (diesel soot) and, more generally, combustion-related primary PM emissions. In this respect, the abatement of uncontrolled shipping emissions has been identified as one of the more cost-effective control options. The abatement of other combustion-related sources such as industrial combustion, wood burning in fireplaces and off-road machinery is also possible, but is less cost-effective. Additionally, climate change mitigation strategies may reduce combustion-related PM emissions.

References Section 1

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Acronyms

AA	ascorbic acid
ACE	angiotensin converting enzyme
ACEA	Association des Constructeurs Europeéns d' Automobiles
ACGIH	American Conference of Governmental Industrial Hygienists
ACS	American Cancer Society
ADI	Acceptable Daily Intake
AEA	Atomic energy agency
AFPC	ambient fine particle concentrator
AHSMOG	Adventist Health Study on Smog
ALI	acute lung injury
ANF	atrial natriuretic factor
ANN	artificial neural network
AP-1	activator protein-1
APA	accumulating potential of atmosphere; see Visser and Römer, 1999
APHEA2	Air Pollution and Health a European Approach 2
ARDS	acute respiratory distress syndrome
ARE	anitioxidant response element
AQCD	air quality criteria document, US-EPA, 1996 and 2001
ATP	adenosine triphosphate
AZ	Arizona
AWMA	Air Waste Management Association
BAL(F)	Bronchoalveolar lavage (fluid)
BAT	best available technology
BEAS-2B	human bronchial epithelial cells
BEES	Dutch acronym for ordinance on emission standards from furnaces
BL	base line scenario
BMP	industrial environmental planning reports
BN	Brown Norway
BP	Blood Pressure
BRD	Federal Republic of Germany
BrdU	Bromo deoxyuridine
BEES	Dutch acronym for ordinance on emission standards from furnaces
BRET	reference document on BAT
BS	black smoke, non reflective dark particulate matter, associated with the smoke stain method (BS number)
C	concentration
CA	California
CAP	concentrated ambient particles (diameters > 0.15µm), concentrated with AFPC
CARB	California air resources board
CBS	central bureau of statistics in NL
CC16	Clara Cell protein
CCDM	co-ordination committee for the monitoring of emissions from target sectors in NL
CCR	central commission on Rhine navigation
Cdyn	Dynamic compliance
CDP	Concentrated Diesel partciles

CEN	European association for standardisation
CIIT	Centers for Health Research
CEN	European association for standardisation
CI	confidence interval, mostly meaning the 95% CI
CNG	condensed natural gas
CNS	central nervous system
CoH	Coefficient of Haze, a measure of air pollution
COPD	chronic obstructive pulmonary disease (often taken to mean chronic bronchitis and emphysema)
COX	cyclooxygenase
CPB	central planning bureau in NL
d	day
DCMR	local environmental agency for the industrial Rijnmond area near Rotterdam
DDR	German Democratic Republic
DEF	deferoxamine
DEP	diesel exhaust particulate
DETR	department of environment transport and regions in the UK
DI	engine with direct injection
DMTU	dimethylthiourea
DNA	deoxyribonucleotide acid
DRG	dorsal root ganglia
EC	elemental carbon
EC	European Commission
EC	European co-ordination, a scenario presented in CPB, 1997
ECH-93	particulate matter sample from Ottawa
ECN	Energy Centre in the Netherlands in Petten
EE	Eastern Europe
EEA	European Environmental Agency in Copenhagen
ELF	epithelial lining fluid
EMEP	European monitoring and evaluation programme
ER	emergency room visits
ERK	extracellular regulated kinase
ERM	environmental resources management
ESP	electrostatic precipitation
ET-1	endothelin-1
EU	European Union
Euro2, -3, -4, -5	different European emission levels for traffic coming into force at various moments in time
EUROS	short term deterministic dispersion model; see Jacobs and Van Pul, 1996
F344N	specific strain of rats
FAG	Standard automatic monitor in NAQMN, based on the principle of beta-attenuation
FEV	Forced Expiratory Volume
FF	fabric filter
FVC	Forced Vital Capacity
GAM	generalised additive model, widely used in epidemiology with Poisson regression
GC	Global Competition scenario, see CPB, 1997
GCM	global climate model

GCS	glutamylcesteine synthetase
GDR	German Democratic Republic
GM	geometric mean
GP	General Practitioner
GSD	geometric standard deviation
GSH	glutathion
GSSG	oxidized glutathione (2 GSH)
h(r)	hour
HD	heavy duty vehicles
HDM	house dust mite
HDV	heavy duty vehicle
HES	high energy scrubber
HGV	high gearing vehicle
hPa	hecto Pascal
HR	heart rate
HRV	heart rate variability
HSC	Harvard six cities study, see Dockery et al., 1993
HVS	high volume sampler
ICAM	intercellular adhesion molecule
ICD-9	International classification of diseases, version 9
ICRP	international commission on radiological protection
IDI	engine with indirect injection
IIASA	International Institute of Aplied Systems Analysis in Austria
IGE	immunoglobuline E
IKK	IκB kinase
IκB	inhibitory kappa B
IL	interleukin
IL-6, -8	interleukine-6, -8
Ill	Illinois
IMO	international maritime organisation
IRAS	Institute for Risk Assessment Studies of University Utrecht in NL
ISEE	International Society for Environmental Epidemiology
ISO	International Standardisation Organisation
JNK	c-jun N terminal kinase
KEMA	Dutch organisation for the testing of electronic materials
KFG	klein filter gerät, a German reference instrument for PM measurement
KNMI	royal Dutch meteorological institute
L	liter (dm^3)
LA	Los Angeles
LC50	lethal concentration for 50% of animals
LD	low duty vehicles
LDV	light duty vehicle
LES	low energy scrubber
LGV	low gearing vehicles
LPG	liquefied petroleum gas
lpm	litres per minute
LPS	lipopolysaccharide
LRTAP	long range transboundary air pollution
LUMC	Medical Centre of Leiden University in NL
LUW	Agricultural University Wageningen

LVS	low volume sampling instruments in NAQMN for SIA
M	marine
MAP	mitogen activated protein
MAP-kinase	mitogen activated protein kinase
MB	environmental balance of RIVM
MCT	Monocrotaline
MEP	department of TNO for environment and energy
min	minute
MIP-2	macrophage inflammatory protein, rat variant of human IL-8
ML	mixing layer
MMAD	mass median aerodynamic diameter
mov av	moving average
MPPDEP	multiple path particle deposition model
MR	multiple regression
mRNA	messenger RNA
MV	environmental outlook of RIVM
N	number of ..
NA	North America
NAAQS	national ambient air quality standard
NADH	reduced nicotnamide adenine dinnucleotide
NADPH	reduced nicotinamide adenine dinucleotide phosphate
NAP	Netherlands Aerosol Programme
NAQMN	national air quality monitoring network of RIVM
NeR	Netherlands emission Regulations
NF-At	proinflammatory transcription factor
NF-κB	nuclear factor-kappa B
NGI	neurogenic inflammation
NIK	NF-κB-inducing kinase
NLCS	Netherlands Cohort Study on Diet and Cancer
NMMAPS	National Morbidity Moratality and Air Pollution Study
NMP	national environmental policy programme in the Netherlands
NO	nitric oxide
NOAEL	no observed adverse effect level, a level used in standard setting
NOS	nitric oxide synthase
OA	ovalbumin
$O_2^{\bullet-}$	superoxide anion radical
OC	Organic carbon, organic carbon compounds also containing other elements as N, O, H
Oct/OAP	proinflammatory transcription factor
$OH^{\bullet-}$	hydroxyl radical
OMEGAM	semi-public service for environmental sampling in Amsterdam
OPS	long term deterministic dispersion model to describe air pollution concentrations; see Van Jaarsveld, 1995
OR	odds ratio
PAH	compounds composed of polycyclic aromatic hydrocarbons
PEACE	study on pollution effects on asthmatic children in Europe,;see Roemer et al 1998
PEF	peak expiratory flow rate
PER	pollution emission register in NL
pH	level of acidity

PM	particulate matter, that usually remains airborne
PM_{10}	PM with a 50% cut off at an aerodynamic diameter of 10 µm, this particle fraction is usually seen as the inhalable fraction
$PM_{2.5}$	PM with a 50% cut off at 2.5 µm aerodynamic diameter
PMNs	polymorphyonuclear leukocytes
PSA	partculate secondary aerosol
PSD	particle size distribution
QUARG	Quality of urban air research group
R	correlation coefficient
RIVM	national institute for public health and the environment in NL
RL	reservoir layer in EUROS
RL	pulmonary resistance
RNA	ribonucleic acid
RNS	reactive nitrogen species
ROFA	residual oil fly ash
ROS	reactive oxygen species
RR	relative risk
RTA	regression tree analysis
SA	South America
SAPALDIA	Swiss Study on Air Pollution and Lung Disease in Adults
SE	standard error
SCARPOL	Swiss Study on Childhood Allergy and Respiratory Symptoms with Respect to Air Pollution, Climate and Pollen
SHR	spontaneous hypertensive rat
SIA	secondary inorganic aerosol, the ammonium salts of sulphate and nitrate and formed in the atmosphere from precursor gases
SIRS	systemic inflammatory response syndrome
SJAC	steam jet aerosol collector; see Slanina 2001
SL	surface layer in EUROS
SOA	secondary organic aerosols, formed in the atmosphere from precursor gases
SOD	superoxide dismutase
SW	south west
T	duration of exposure
T1/2	mean atmospheric residence time
TDI	turbo diesel injection
TEOM	tapered element oscillating microbalance for continuos measurement of PM
Th1, Th2	specific T-helper cells
TL	top layer in EUROS
TLV	threshold limit value
TNFa	tumour necrosis factor alpha
TNO	Netherlands organisation for applied scientific research
TOC	total organic carbon
TSP	total suspended particulate
UA	uric acid
UBA	Umwelt Bundesambt, environmental authority in Germany
UF	ultra fine PM, with diameters < 0.1 µm (~ $PM_{0.1}$)
UK	United Kingdom
UN/ECE	United Nations Economic Commission for Europe

URT	Upper Respiratory tract
US	United States of America
USC	University of Southern California
US-EPA	Environmental Protection Agency in the US
USSR	Union of Socialist Soviet Republics
UV	ultra violet
VA	Veterans Association
VAPS	versatile air pollution sampler
Vd	dry deposition velocity
VR1	vanilloid receptor
VROM	Ministry of housing, physical planning and environment in NL
WHO	World Health Organisation
WRAC	wide range aerosol classifier, CEN reference instrument for PM
Zi	mixing layer height

Annex A. Main Results of update time-series study on mortality

Table A1 Monitoring points in the Netherlands and different regions used in the time-series update in 1992–1998.
TABLE 1: Type and location of the selected air pollution monitoring site and pollutant concentration data used.

Type	Region	Town	Idnr	Traffic category	Particulate matter			Gases				
					PM10	Black Smoke	secondary aerosol *	ammonia	carbon monoxide	nitrogen oxides	sulphur dioxide	ozone
Rural	South	Vredepeel	131	low	x	x	x	x		x	x	x
		Wijnandsrade	133	low	x	x				x	x	X
		Biest Houtakker	230	low	x	x			x	x	x	X
		Huijbergen	235	low			x	x		x	x	X
	West	Philippine	318	low	x	x				x	x	X
		Westmaas	437	-	x	x				x	x	X
		De Zilk	444	-	x	x	x	x		x		
		Bilthoven	627	low			x		x	x	x	
		Zegveld	633	low				x		x	x	X
	East	Eibergen	722	low	x	x		x		x	x	X
		Wageningen	724	low	x					x	x	X
		Wekerom	738	low				x				
	North	Witteveen	928	low	x	x	x	x		x	x	X
		Kollumerwaard	934	low			x		x	x	x	X
		Wieringerwerf **	538	low	x	x		x		x	x	X
Urban	West	The Hague	404	high	x					x	x	X
		Rotterdam	418	high	x				x	x	x	
		Dordrecht	441	high	x				x	x		X
		Amsterdam	520	high	x				x	x	x	X
		Utrecht	640	high					x	x		X

*) Secondary aerosol: ammonium (NH4+), chloride (Cl-), nitrate (NO3-), and sulphate (SO4=).

**) Located in the upper northern part of the West region, the Wieringerwerf site was considered to be representative for the North region.

Table A2 24-hour average air pollution concentrations in µg/m^3 in the Netherlands for the entire time (a,b,c,d) during the old period 1992–94 and the new period 1995–1998 (e).

a. entire Netherlands

	mean	Min	max	perc_50	perc_95	perc_98
PM_{10}	40.6	8.7	251.7	34.1	81.9	101.2
BS (ZWR)	11.1	0.4	116.3	7.8	31.7	40.9
so2	7.7	0.6	60.8	6.1	19.4	26.9
o3	49.6	1.2	189.3	47.3	112.4	133.4
NO_2	29.7	6.7	93.3	28.0	50.8	57.3
NO	13.1	0	224.4	5.4	51.8	75.7
NH3	9.2	0	53.2	7.8	20.6	25.3
CO (x10)	50.7	17.2	313.4	44.0	95.5	112.4
SIA	12.4	2.2	95.9	10.1	27.4	37.4
NO3	4.6	0.4	28.9	3.7	10.8	14.3
SO4	4.1	0.4	48.2	3.1	10.3	13.6
Flue	4.8	0	57.0	2	18	
Temp avg	9.9	-13	26	10	20	
Pressure avg	1016	981	1043	1016	1031	
Rel. Hum.	83.3	38	100.0	85	97	

b. four largest cities or traffic high

	mean	Min	max	perc_50	perc_95	perc_98
PM_{10}	42.2	6.0	324.5	35.8	85.1	107.0
BS (ZWR)						
so2	11.9	0.7	80.7	10.3	27.7	36.3
o3	41.3	0.1	200.6	36.2	104.0	127.5
NO_2	42.0	10.1	128.4	40.8	67	75.8
NO	23.4	0	413	11.4	86.8	124.8
NH3						
CO (x10)	58.7	20	409.6	51.0	111	134

c. Netherlands minus four largest cities

	mean	Min	max	perc_50	perc_95	perc_98
PM_{10}	39.8	7.6	246.1	33.3	81.3	97.9
BS (ZWR)	11.1	0.4	116.3	7.8	31.7	40.9
so2	6.7	0.5	61.8	5.1	17.2	24.4
o3	52.3	0.9	193.8	50.8	115.1	137.3
NO_2	25.3	5.1	86.6	23.2	45.4	51.8
NO	9.5	0	177.7	3.4	40.8	60.1
NH3	9.2	0.0	53.2	7.8	20.6	25.3
CO (x10)	40.2	13.7	226.7	35.3	74.5	92.7

d. traffic-low region

	mean	Min	max	perc_50	perc_95	perc_98
PM_{10}	40.0	7.2	227.1	33.8	80.8	100.2
BS (ZWR)	11.1	0.4	113.3	8.0	31.7	41.4
so2	6.6	0.3	62.3	5.0	17.3	24.3
o3	52.5	1.0	192.5	50.8	115.1	137.3
NO_2	24.9	4.7	86.5	22.9	44.7	50.8
NO	9.0	-0.4	168.9	3.3	40.1	57.2
NH3	10.1	0.0	58.7	8.7	22.7	28.7
CO (x10)	40.2	13.7	226.7	35.3	74.5	92.7
SIA	12.4	1.9	96.0	10.1	27.4	37.6
NO3	28.9	0.4	4.6	3.7	10.8	14.7
SO4	4.1	0.4	48.2	3.2	10.3	13.8

e. entire Netherlands old (1992–1994) and new period (1995–1998)

	period	N	Mean	max	min	perc_98	perc_95	perc_50
BS	All years	2557	11.06	116.33	0.44	40.89	31.74	7.77
	Old	1096	11.23	116.33	0.56	42.14	32.47	7.78
	New	1461	10.93	68.56	0.44	40.78	30.81	7.67
PM_{10}	All years	2554	40.59	251.67	8.67	101.21	81.91	34.06
	Old	1093	41.34	251.67	8.67	108.62	86.64	34.31
	New	1461	40.03	150.82	12.30	95.75	78.71	34.03
SO_2	All years	2557	7.65	60.75	0.56	26.94	19.38	6.06
	All years	1096	9.43	49.04	1.41	29.06	21.50	7.88
	New	1461	6.32	60.75	0.56	23.94	14.52	4.89
O_3	All years	2557	49.55	189.30	1.18	133.43	112.42	47.30
	Old	1096	49.56	185.83	1.18	140.94	118.21	45.63
	New	1461	49.54	189.30	1.26	131.34	107.44	48.59
NO_2	All years	2557	29.70	93.31	6.73	57.29	50.76	28.03
	Old	1096	30.25	93.31	6.73	58.46	51.84	28.34
	New	1461	29.29	92.69	7.03	56.80	49.93	27.47
NO	All years	2557	13.14	224.42	-0.06	75.74	51.82	5.41
	Old	1096	14.51	224.42	0.51	88.43	52.93	6.33
	New	1461	12.11	162.11	-0.06	70.16	51.19	4.93
NH	All years	2339	9.15	53.20	0.00	25.29	20.57	7.84
	Old	881	9.01	52.65	0.56	25.76	18.91	7.86
	New	1458	9.24	53.20	0.00	25.01	21.10	7.81
CO	All years	2557	50.66	313.35	17.20	112.44	95.54	44.00

*10	Old	1096	53.60	313.35	22.40	123.20	101.23	46.22
	New	1461	48.46	176.08	17.20	104.40	90.88	42.60
SIA	All years	2308	12.4	96.0	2.2	37.4	27.4	10.1
Total								
	Old	847	14.7	96.0	2.8	48.9	35.5	11.7
	New	1461	11.1	53.3	2.2	29.6	23.8	9.2
NO_3	All years	2556	4.6	28.9	0.4	14.3	10.8	3.7
	Old	1095	5.0	28.9	0.5	16.3	12.8	4.0
	New	1461	4.2	25.1	0.4	12.4	9.6	3.5
SO_4	All years	2553	4.1	48.2	0.4	13.6	10.3	3.2
	Old	1092	5.0	48.2	0.6	16.6	12.3	3.9
	New	1461	3.5	18.6	0.4	11.1	8.6	2.7

Table A3 Association between air pollution and **total mortality** in different regions of the Netherlands. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders.

	NL			Cities			Non-urban			Traffic high			Traffic low		
Pollutant															
PM10lag 1	1,036	1,025	1,047	1,018	0,995	1,043	1,037	1,025	1,049	1,020	1,001	1,039	1,040	1,019	1,061
PM10 avg	1,050	1,038	1,062	1,044	1,016	1,072	1,048	1,035	1,062	1,028	1,007	1,050	1,061	1,038	1,084
BS lag 1	1,040	1,028	1,053				1,040	1,026	1,053				1,039	1,015	1,063
BS avg	1,063	1,050	1,077				1,063	1,049	1,078				1,073	1,048	1,099
NH3 lag 1	1,023	1,010	1,037				1,023	1,008	1,037				1,019	0,997	1,041
NH3 avg	1,007	0,995	1,018				1,006	0,994	1,019				1,005	0,986	1,025
CO lag 1	1,048	1,031	1,065	1,035	1,003	1,069	1,061	1,037	1,084	1,028	1,003	1,053	1,062	1,022	1,103
CO avg	1,065	1,044	1,086	1,053	1,011	1,096	1,086	1,058	1,114	1,026	0,994	1,058	1,121	1,073	1,172
NO lag 1	1,018	1,006	1,030	1,012	0,994	1,030	1,020	1,003	1,036	1,011	0,997	1,025	1,003	0,973	1,034
NO avg	1,025	1,009	1,042	1,017	0,992	1,041	1,029	1,008	1,051	1,006	0,988	1,025	1,040	1,000	1,080
NO2 lag 1	1,035	1,024	1,046	1,042	1,020	1,064	1,030	1,018	1,043	1,028	1,011	1,045	1,046	1,023	1,069
NO2 avg	1,048	1,038	1,058	1,037	1,016	1,058	1,048	1,037	1,059	1,020	1,004	1,036	1,075	1,055	1,095
SO2 lag 1	1,042	1,020	1,065	1,031	0,997	1,067	1,037	1,012	1,062	1,036	1,009	1,063	1,039	0,996	1,083
SO2 avg	1,133	1,107	1,159	1,081	1,037	1,127	1,144	1,116	1,173	1,060	1,027	1,095	1,141	1,093	1,191
O3 lag 1	1,041	1,024	1,059	1,024	0,985	1,065	1,041	1,023	1,059	1,038	1,007	1,070	1,039	1,008	1,072
O3 avg	1,028	1,009	1,047	1,036	0,991	1,083	1,026	1,006	1,046	1,026	0,992	1,062	1,015	0,982	1,050
Sec lag 1	1,039	1,027	1,052				1,041	1,027	1,055				1,047	1,023	1,071
Sec avg	1,035	1,021	1,048				1,033	1,019	1,048				1,037	1,012	1,062
NO3 lag 1	1,030	1,020	1,040				1,030	1,019	1,041				1,040	1,021	1,060
NO3 avg	1,025	1,015	1,036				1,024	1,013	1,035				1,038	1,019	1,057
SO4 lag 1	1,028	1,018	1,038				1,030	1,019	1,041				1,041	1,022	1,060
SO4 avg	1,027	1,017	1,037				1,027	1,016	1,038				1,036	1,018	1,055

For all pollutants, RR were higher in the summer than in the winter season (Table A4), except for NO_3, and statistically significant RR were above unity, except for **O_3**, in the winter period. For ozone, significant associations were only found for the summer period.

Table A4 Association between air pollution and **total mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Season-specific analysis.

	Summer			Winter		
	RR	RR-lo	RR-hi	RR	RR-lo	RR-hi
Pollutant						
PM10 lag 1	1,086	1,061	1,112	1,029	1,015	1,043
PM10 avg	1,129	1,098	1,161	1,041	1,026	1,056
BS lag 1	1,173	1,123	1,224	1,030	1,016	1,044
BS avg	1,333	1,266	1,404	1,048	1,034	1,063
NH3 lag 1	1,034	1,009	1,059	1,027	1,009	1,045
NH3 avg	1,037	1,014	1,061	1,000	0,984	1,015
CO lag 1	1,172	1,113	1,233	1,033	1,014	1,052
CO avg	1,316	1,231	1,406	1,047	1,023	1,070
NO lag 1	1,073	0,996	1,156	1,015	1,002	1,028
NO avg	1,283	1,155	1,425	1,018	1,001	1,035
NO2 lag 1	1,063	1,041	1,086	1,023	1,008	1,037
NO2 avg	1,108	1,084	1,133	1,033	1,020	1,047
SO2 lag 1	1,109	1,042	1,179	1,029	1,005	1,053
SO2 avg	1,251	1,162	1,347	1,111	1,085	1,139
O3 lag 1	1,067	1,045	1,089	1,008	0,979	1,037
O3 avg	1,089	1,064	1,114	0,966	0,937	0,996
Sec lag 1	1,052	1,025	1,079	1,039	1,023	1,055
Sec avg	1,067	1,036	1,099	1,027	1,011	1,044
NO3 lag 1	1,035	0,999	1,073	1,041	1,027	1,056
NO3 avg	1,033	0,996	1,070	1,032	1,019	1,046
SO4 lag 1	1,058	1,006	1,112	1,023	1,011	1,035
SO4 avg	1,061	1,008	1,117	1,021	1,010	1,033

In two-pollutant models (Table A5) the independent effect of ozone remained, irrespective of the co-pollutant added to the model. Except when BS was added as co-pollutant, the association between PM_{10} and total mortality remained statistically significant in all two-pollutant models. NO_2 and **SO_2** also remained statistically significant in all two-pollutant models. Most of the models with CO, NH_3 or NO as the co-pollutant resulted in significant negative associations for CO, NH_3 or NO.

Table A5 Association between air pollution and **total mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Two-pollutant models, weekly average concentrations, for **O_3** lag 1.

Model	Pollut-ant	RR	RR-lo	RR-hi
PM10+BS	PM10	0,995	0,971	1,019
	BS	1,071	1,043	1,099
PM10+NH3	PM10	1,061	1,045	1,077
	NH3	0,980	0,966	0,993
PM10+CO	PM10	1,046	1,028	1,064
	CO	1,009	0,979	1,039
PM10+NO	PM10	1,057	1,043	1,072
	NO	0,983	0,965	1,002
PM10+NO2	PM10	1,018	1,002	1,034
	NO2	1,037	1,024	1,051
PM10+SO2	PM10	1,016	1,000	1,032
	SO2	1,112	1,079	1,146
PM10+O3	PM10	1,054	1,042	1,067
	O3	1,052	1,035	1,070
PM10+sec	PM10	1,071	1,047	1,096
	sec	0,978	0,955	1,001
PM10+NO3	PM10	1,066	1,047	1,086
	NO3	0,983	0,967	0,998
PM10+SO4	PM10	1,065	1,042	1,088
	SO4	0,984	0,967	1,002
BS+NH3	BS	1,079	1,062	1,096
	NH3	0,978	0,966	0,991
BS+CO	BS	1,099	1,072	1,127
	CO	0,946	0,910	0,984
BS+NO	BS	1,110	1,088	1,132
	NO	0,935	0,912	0,959
BS+NO2	BS	1,031	1,012	1,050
	NO2	1,030	1,016	1,045
BS+SO2	BS	1,026	1,009	1,045
	SO2	1,097	1,063	1,132
BS+O3	BS	1,071	1,058	1,085
	O3	1,057	1,040	1,075
BS+sec	BS	1,065	1,046	1,085
	sec	0,995	0,979	1,013
BS+NO3	BS	1,067	1,051	1,085
	NO3	0,993	0,981	1,006
BS+SO4	BS	1,073	1,054	1,092
	SO4	0,990	0,976	1,003
NH3+CO	NH3	0,988	0,975	1,001
	CO	1,070	1,045	1,095
NH3+NO	NH3	1,001	0,988	1,014
	NO	1,018	0,999	1,037
NH3+NO2	NH3	0,968	0,955	0,982
	NO2	1,061	1,049	1,074
NH3+SO2	NH3	0,995	0,983	1,006
	SO2	1,161	1,131	1,192
NH3+O3	NH3	1,008	0,997	1,020
	O3	1,050	1,031	1,068
NH3+sec	NH3	0,982	0,967	0,996
	sec	1,047	1,030	1,065
NH3+NO3	NH3	0,988	0,973	1,002
	NO3	1,030	1,016	1,045
NH3+SO4	NH3	0,995	0,983	1,008
	SO4	1,029	1,017	1,041
CO+NO	CO	1,218	1,164	1,274
	NO	0,887	0,856	0,919
CO+NO2	CO	0,962	0,930	0,995
	NO2	1,065	1,048	1,083
CO+SO2	CO	1,019	0,996	1,043
	SO2	1,120	1,090	1,150
CO+O3	CO	1,077	1,056	1,099
	O3	1,054	1,036	1,072
CO+sec	CO	1,032	1,004	1,061
	sec	1,023	1,007	1,040
CO+NO3	CO	1,047	1,022	1,073
	NO3	1,011	0,999	1,023
CO+SO4	CO	1,044	1,018	1,070
	SO4	1,014	1,002	1,026
NO+NO2	NO	0,951	0,931	0,972
	NO2	1,071	1,057	1,085
NO+SO2	NO	1,001	0,984	1,018
	SO2	1,132	1,105	1,160
NO+O3	NO	1,034	1,017	1,051
	O3	1,049	1,031	1,067
NO+sec	NO	0,994	0,976	1,013
NO+NO3	NO	1,007	0,990	1,025
	NO3	1,023	1,013	1,034
NO+SO4	NO	1,004	0,987	1,022
	SO4	1,026	1,015	1,037
	sec	1,036	1,022	1,051
NO2+SO2	NO2	1,027	1,015	1,039
	SO2	1,092	1,063	1,123
NO2+O3	NO2	1,054	1,044	1,065
	O3	1,060	1,042	1,078
NO2+sec	NO2	1,043	1,030	1,056
	sec	1,006	0,991	1,023
NO2+NO3	NO2	1,049	1,037	1,062
	NO3	0,995	0,983	1,008
NO2+SO4	NO2	1,043	1,032	1,054
	SO4	1,007	0,996	1,018
SO2+O3	SO2	1,142	1,116	1,168
	O3	1,052	1,035	1,070
SO2+sec	SO2	1,122	1,090	1,155
	sec	1,008	0,993	1,024
SO2+NO3	SO2	1,124	1,096	1,153
	NO3	1,004	0,993	1,015
SO2+SO4	SO2	1,125	1,095	1,155
	SO4	1,005	0,994	1,017
O3+sec	O3	1,051	1,032	1,071
	sec	1,039	1,026	1,053
O3+NO3	O3	1,047	1,030	1,065
	NO3	1,028	1,018	1,039
O3+SO4	O3	1,047	1,030	1,065
	SO4	1,030	1,020	1,040
Sec+NO3	Sec	1,047	1,010	1,084
	NO3	0,990	0,961	1,019
Sec+SO4	Sec	1,045	1,009	1,083
	SO4	0,990	0,962	1,020
NO3+SO4	NO3	1,011	0,996	1,027
	SO4	1,018	1,003	1,034

Respiratory mortality

Table A6 shows the associations between air pollution and daily respiratory mortality for different regions in the Netherlands. All statistically significant RR were above unity. Again, significant associations were found for all air pollutants. All Relative Risks, except one (**O_3** lag 1), were larger than the RR for total mortality. RR for the four major cities and the rest of the Netherlands were comparable; substantially higher risks were found for the 'traffic low' region compared with the 'traffic high' region.

Table A6 Association between air pollution and **respiratory mortality** in different regions of the Netherlands. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders.

	NL			Cities			Non-urban			Traffic high			Traffic low		
Pollutant															
PM10 lag 1	1,120	1,084	1,157	1,112	1,035	1,194	1,124	1,084	1,166	1,088	1,027	1,151	1,141	1,071	1,216
PM10 avg	1,167	1,126	1,210	1,141	1,050	1,241	1,179	1,132	1,227	1,084	1,014	1,159	1,243	1,158	1,333
BS lag 1	1,121	1,081	1,163				1,115	1,071	1,161				1,120	1,041	1,205
BS avg	1,184	1,138	1,232				1,186	1,136	1,239				1,246	1,152	1,347
NH3 lag 1	1,065	1,024	1,108				1,074	1,028	1,121				1,043	0,974	1,117
NH3 avg	1,051	1,014	1,089				1,054	1,014	1,096				1,044	0,981	1,110
CO lag 1	1,145	1,089	1,205	1,160	1,050	1,280	1,182	1,103	1,268	1,143	1,057	1,237	1,219	1,078	1,378
CO avg	1,176	1,105	1,251	1,177	1,037	1,336	1,243	1,146	1,349	1,058	0,956	1,170	1,383	1,198	1,597
NO lag 1	1,056	1,017	1,096	1,050	0,993	1,109	1,054	1,001	1,109	1,054	1,009	1,101	1,036	0,940	1,143
NO avg	1,067	1,016	1,121	1,054	0,979	1,136	1,082	1,013	1,155	1,007	0,949	1,068	1,130	0,998	1,280
NO2 lag 1	1,107	1,071	1,145	1,093	1,023	1,168	1,114	1,072	1,157	1,088	1,033	1,147	1,150	1,073	1,232
NO2 avg	1,127	1,094	1,161	1,104	1,036	1,177	1,137	1,099	1,175	1,051	0,999	1,106	1,206	1,136	1,280
SO2 lag 1	1,191	1,116	1,271	1,220	1,100	1,354	1,165	1,080	1,256	1,172	1,079	1,273	1,293	1,134	1,474
SO2 avg	1,393	1,299	1,493	1,321	1,161	1,503	1,400	1,295	1,514	1,224	1,105	1,356	1,634	1,429	1,868
O3 lag 1	1,011	0,948	1,078	1,032	0,888	1,198	1,005	0,937	1,078	0,962	0,855	1,083	1,086	0,960	1,228
O3 avg	1,239	1,167	1,315	1,194	1,033	1,381	1,243	1,166	1,326	1,200	1,071	1,346	1,280	1,143	1,434
Sec lag 1	1,073	1,030	1,118				1,076	1,029	1,125				1,037	0,958	1,122
Sec avg	1,122	1,075	1,171				1,144	1,092	1,198				1,164	1,071	1,265
NO3 lag 1	1,062	1,028	1,097				1,065	1,028	1,104				1,057	0,992	1,125
NO3 avg	1,083	1,049	1,118				1,099	1,062	1,138				1,125	1,058	1,196
SO4 lag 1	1,058	1,025	1,092				1,062	1,026	1,100				1,046	0,984	1,113
SO4 avg	1,085	1,052	1,119				1,104	1,068	1,142				1,131	1,066	1,201

For all pollutants, higher RR were found in the summer period, except for NO and NO_3 (Table A7).

Table A7 Association between air pollution and **respiratory mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Season-specific analysis.

	Summer			Winter		
	RR	RR-lo	RR-hi	RR	RR-lo	RR-hi
Pollutant						
PM10 lag 1	1,290	1,193	1,394	1,093	1,050	1,137
PM10 avg	1,594	1,454	1,749	1,119	1,070	1,170
BS lag 1	1,448	1,253	1,674	1,104	1,059	1,149
BS avg	2,165	1,824	2,570	1,146	1,097	1,198
NH3 lag 1	1,174	1,085	1,271	1,046	0,995	1,100
NH3 avg	1,251	1,162	1,346	1,009	0,965	1,055
CO lag 1	1,436	1,205	1,711	1,118	1,056	1,183
CO avg	1,674	1,336	2,096	1,143	1,067	1,224
NO lag 1	1,086	0,838	1,409	1,052	1,011	1,094
NO avg	0,900	0,621	1,303	1,064	1,011	1,120
NO2 lag 1	1,137	1,058	1,221	1,093	1,048	1,139
NO2 avg	1,280	1,189	1,377	1,084	1,043	1,127
SO2 lag 1	1,423	1,155	1,754	1,153	1,076	1,236
SO2 avg	2,179	1,700	2,793	1,320	1,228	1,420
O3 lag 1	1,149	1,067	1,237	0,861	0,781	0,950
O3 avg	1,642	1,525	1,767	0,937	0,852	1,030
Sec lag 1	1,104	1,013	1,202	1,074	1,021	1,129
Sec avg	1,254	1,138	1,382	1,093	1,038	1,150
NO3 lag 3	1,044	0,970	1,123	1,048	1,003	1,094
NO3 avg	1,104	0,983	1,239	1,097	1,053	1,142
SO4 lag 1	1,211	1,024	1,432	1,035	0,998	1,074
SO4 avg	1,344	1,091	1,655	1,049	1,013	1,087

In two-pollutant models all statistically significant ozone associations disappeared; a statistically significant association between PM_{10}, NO_2, **SO_2** and BS (except when PM_{10} was the co-pollutant) and respiratory morbidity remained, irrespective of the second pollutant added to the model. CO, NO, SIA, NO_3 and SO_4 were less stable when a co-pollutant was entered.

Table A8 Association between air pollution and **respiratory mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Two-pollutant models, weekly average concentrations, for **O_3** lag 1.

Model	Pollut-ant	RR	RR-lo	RR-hi
PM10+BS	PM10	1,121	1,041	1,207
	BS	1,052	0,969	1,141
PM10+NH3	PM10	1,188	1,135	1,245
	NH3	0,970	0,929	1,012
PM10+CO	PM10	1,197	1,136	1,262
	CO	0,945	0,863	1,034
PM10+NO	PM10	1,200	1,151	1,252
	NO	0,934	0,882	0,990
PM10+NO2	PM10	1,106	1,053	1,161
	NO2	1,066	1,023	1,110
PM10+SO2	PM10	1,082	1,031	1,135
	SO2	1,263	1,152	1,385
PM10+O3	PM10	1,171	1,130	1,214
	O3	1,049	0,983	1,119
PM10+sec	PM10	1,248	1,162	1,341
	sec	0,930	0,862	1,004
PM10+NO3	PM10	1,253	1,184	1,326
	NO3	0,930	0,885	0,978
PM10+SO4	PM10	1,273	1,190	1,361
	SO4	0,918	0,867	0,972
BS+NH3	BS	1,209	1,152	1,269
	NH3	0,975	0,936	1,015
BS+CO	BS	1,306	1,211	1,409
	CO	0,847	0,752	0,954
BS+NO	BS	1,310	1,235	1,389
	NO	0,851	0,791	0,915
BS+NO2	BS	1,106	1,045	1,171
	NO2	1,066	1,021	1,113
BS+SO2	BS	1,075	1,019	1,135
	SO2	1,278	1,162	1,405
BS+O3	BS	1,191	1,144	1,240
	O3	1,059	0,992	1,130
BS+sec	BS	1,164	1,100	1,231
	sec	1,022	0,968	1,080
BS+NO3	BS	1,197	1,139	1,257
	NO3	0,991	0,953	1,031
BS+SO4	BS	1,212	1,147	1,280
	SO4	0,979	0,938	1,022
NH3+CO	NH3	1,006	0,966	1,048
	CO	1,156	1,073	1,246
NH3+NO	NH3	1,041	1,002	1,082
	NO	1,029	0,972	1,089
NH3+NO2	NH3	0,970	0,929	1,013
	NO2	1,128	1,087	1,172
NH3+SO2	NH3	1,016	0,980	1,053
	SO2	1,431	1,320	1,552
NH3+O3	NH3	1,052	1,015	1,090
	O3	1,016	0,950	1,087
NH3+sec	NH3	0,970	0,927	1,016
	sec	1,148	1,088	1,211
NH3+NO3	NH3	1,005	0,961	1,052
	NO3	1,079	1,033	1,127
NH3+SO4	NH3	1,020	0,981	1,060
	SO4	1,094	1,054	1,136
CO+NO	CO	1,657	1,440	1,907
	NO	0,739	0,661	0,826
CO+NO2	CO	0,895	0,805	0,995
	NO2	1,182	1,123	1,244
CO+SO2	CO	1,049	0,978	1,126
	SO2	1,356	1,252	1,468
CO+O3	CO	1,182	1,110	1,259
	O3	1,039	0,973	1,110
CO+sec	CO	1,019	0,937	1,109
	sec	1,115	1,059	1,173
CO+NO3	CO	1,121	1,038	1,210
	NO3	1,047	1,007	1,088
CO+SO4	CO	1,115	1,032	1,204
	SO4	1,050	1,011	1,091
NO+NO2	NO	0,874	0,819	0,934
	NO2	1,197	1,150	1,245
NO+SO2	NO	1,007	0,956	1,059
	SO2	1,389	1,292	1,493
NO+O3	NO	1,070	1,017	1,125
	O3	1,024	0,959	1,095
NO+sec	NO	0,957	0,904	1,014
	sec	1,137	1,087	1,189
NO+NO3	NO	1,016	0,965	1,071
	NO3	1,078	1,043	1,115
NO+SO4	NO	1,011	0,959	1,067
	SO4	1,082	1,046	1,119
NO2+SO2	NO2	1,064	1,027	1,103
	SO2	1,280	1,178	1,391
NO2+O3	NO2	1,134	1,099	1,169
	O3	1,065	0,998	1,137
NO2+sec	NO2	1,095	1,053	1,139
	Sec	1,057	1,005	1,112
NO2+NO3	NO2	1,129	1,086	1,173
	NO3	1,002	0,963	1,044
NO2+SO4	NO2	1,116	1,078	1,154
	SO4	1,030	0,994	1,067
SO2+O3	SO2	1,409	1,313	1,511
	O3	1,061	0,994	1,132
SO2+sec	SO2	1,343	1,232	1,465
	sec	1,049	0,999	1,101
SO2+NO3	SO2	1,372	1,270	1,483
	NO3	1,021	0,986	1,058
SO2+SO4	SO2	1,381	1,275	1,496
	SO4	1,021	0,985	1,058
O3+sec	O3	1,007	0,939	1,080
	sec	1,123	1,076	1,172
O3+NO3	O3	1,024	0,960	1,092
	NO3	1,084	1,051	1,119
O3+SO4	O3	1,025	0,960	1,093
	SO4	1,086	1,053	1,121
Sec+NO3	Sec	1,126	1,007	1,260
	NO3	0,997	0,909	1,093
Sec+SO4	Sec	1,113	0,994	1,245
	SO4	1,007	0,919	1,105
NO3+SO4	NO3	1,043	0,994	1,095
	SO4	1,049	1,001	1,101

Chronic obstructive pulmonary mortality

Table A9 shows the associations between air pollution and daily chronic obstructive pulmonary mortality for different regions in the Netherlands. Except for NH_3, significant associations were found for all pollutants and all statistically significant RR were above unity, except for two ozone associations in the urbanised region. Apart from a few associations, overall larger RR were found compared with RR for total mortality, and comparable RR were found with respiratory mortality. In the four major cities and the 'high traffic' regions substantially less significant RR were found than in the non-urban and 'low traffic' areas. All RR were higher for the summer period (Table A10). In two-pollutant models (Table A11) **SO_2** remained statistically significant in all models, while PM_{10}, BS and NO_2 remained statistically significant in all but one model. Ozone associations went down when other pollutants were added to the model. RR for CO, NO, NH_3 and secondary aerosol were less stable when a co-pollutant was entered. For NH_3 and NO negative associations were found when a co-pollutant was added to the model.

Table A9 Association between air pollution and **chronic obstructive pulmonary mortality** in different regions of the Netherlands. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders.

	NL			Cities			Non-urban			Traffic high			Traffic low		
Pollutant															
PM10 lag 1	1,111	1,064	1,161	1,105	0,997	1,226	1,125	1,072	1,180	1,068	0,983	1,160	1,117	1,022	1,220
PM10 avg	1,154	1,100	1,211	1,065	0,943	1,204	1,168	1,107	1,232	1,057	0,959	1,164	1,198	1,086	1,322
BS lag 1	1,089	1,036	1,144				1,087	1,030	1,147				1,077	0,971	1,193
BS avg	1,174	1,114	1,238				1,185	1,119	1,255				1,224	1,098	1,363
NH3 lag 1	1,018	0,964	1,074				1,020	0,962	1,081				1,001	0,909	1,103
NH3 avg	1,017	0,969	1,068				1,017	0,965	1,072				1,021	0,937	1,113
CO lag 1	1,084	1,012	1,161	1,091	0,943	1,262	1,111	1,011	1,220	1,064	0,949	1,194	1,090	0,915	1,297
CO avg	1,202	1,107	1,306	1,119	0,931	1,345	1,284	1,153	1,431	0,986	0,852	1,141	1,361	1,115	1,661
NO lag 1	1,012	0,961	1,065	1,027	0,947	1,113	1,003	0,936	1,074	1,018	0,955	1,086	0,942	0,819	1,083
NO avg	1,088	1,019	1,162	1,024	0,918	1,141	1,136	1,042	1,240	0,967	0,887	1,054	1,154	0,971	1,370
NO2 lag 1	1,063	1,015	1,114	1,101	0,998	1,214	1,055	1,001	1,111	1,076	0,997	1,162	1,057	0,958	1,167
NO2 avg	1,118	1,074	1,164	1,130	1,029	1,240	1,120	1,072	1,171	1,026	0,953	1,104	1,162	1,069	1,263
SO2 lag 1	1,118	1,026	1,218	1,182	1,016	1,374	1,094	0,993	1,206	1,096	0,972	1,235	1,220	1,022	1,456
SO2 avg	1,301	1,185	1,427	1,313	1,089	1,583	1,289	1,162	1,429	1,159	0,999	1,344	1,478	1,227	1,781
O3 lag 1	1,046	0,970	1,127	1,219	1,007	1,477	1,012	0,934	1,096	1,109	0,953	1,290	1,112	0,961	1,288
O3 avg	1,053	0,973	1,141	1,040	0,843	1,283	1,059	0,973	1,153	1,060	0,899	1,249	1,137	0,973	1,329
Sec lag 1	1,071	1,016	1,129				1,073	1,013	1,136				0,988	0,886	1,101
Sec avg	1,118	1,057	1,183				1,152	1,084	1,225				1,163	1,037	1,304
NO3 lag 1	1,071	1,026	1,118				1,076	1,027	1,127				1,015	0,930	1,106
NO3 avg	1,087	1,042	1,134				1,098	1,049	1,149				1,105	1,015	1,202
SO4 lag 1	1,073	1,029	1,119				1,083	1,035	1,133				1,043	0,959	1,135
SO4 avg	1,091	1,047	1,138				1,116	1,067	1,167				1,114	1,026	1,211

All RR were higher for the summer period (Table A10).

Table A10 Association between air pollution and **chronic obstructive pulmonary mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Season-specific analysis.

	Summer			Winter		
	RR	RR-lo	RR-hi	RR	RR-lo	RR-hi
Pollutant						
PM10 lag 1	1,323	1,193	1,467	1,063	1,007	1,123
PM10 avg	1,452	1,282	1,644	1,103	1,039	1,171
BS lag 1	1,576	1,303	1,906	1,058	1,001	1,118
BS avg	1,989	1,579	2,504	1,132	1,067	1,200
NH3 lag 1	1,125	1,013	1,249	0,988	0,920	1,061
NH3 avg	1,116	1,011	1,232	0,989	0,930	1,053
CO lag 1	1,479	1,170	1,869	1,039	0,962	1,123
CO avg	1,650	1,220	2,231	1,145	1,044	1,256
NO lag 1	1,066	0,752	1,511	1,008	0,955	1,063
NO avg	1,123	0,687	1,835	1,071	1,000	1,148
NO2 lag 1	1,152	1,048	1,266	1,019	0,961	1,080
NO2 avg	1,241	1,125	1,370	1,071	1,016	1,129
SO2 lag 1	1,538	1,175	2,013	1,066	0,972	1,170
SO2 avg	1,881	1,354	2,613	1,238	1,123	1,365
O3 lag 1	1,151	1,053	1,259	0,946	0,836	1,071
O3 avg	1,327	1,200	1,467	0,850	0,749	0,965
Sec lag 1	1,112	0,991	1,247	1,033	0,967	1,104
Sec avg	1,184	1,036	1,352	1,090	1,018	1,166
NO3 lag 1	1,142	0,980	1,330	1,023	0,965	1,085
NO3 avg	1,092	0,959	1,243	1,078	1,022	1,138
SO4 lag 1	1,289	1,023	1,624	1,029	0,981	1,081
SO4 avg	1,259	1,011	1,567	1,060	1,012	1,110

In two-pollutant models (Table A11) **SO_2** remained statistically significant in all models, while PM_{10}, BS and NO_2 remained statistically significant in all but one model. Ozone associations reduced when other pollutants were added to the model. RR for CO, NO, NH_3 and secondary aerosol were less stable when a co-pollutant was entered into the model, while for NH_3 and NO negative associations were found when a co-pollutant was added.

Table A11 Association between air pollution and **chronic obstructive lung mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Two-pollutant models, weekly average concentrations.

Model	Pollut-ant	RR	RR-lo	RR-hi
PM10+BS	PM10	1,083	0,980	1,198
	BS	1,083	0,970	1,208
PM10+NH3	PM10	1,192	1,120	1,269
	NH3	0,936	0,883	0,993
PM10+CO	PM10	1,143	1,065	1,226
	CO	1,022	0,906	1,153
PM10+NO	PM10	1,171	1,106	1,239
	NO	0,966	0,894	1,043
PM10+NO2	PM10	1,102	1,032	1,177
	NO2	1,058	1,001	1,117
PM10+SO2	PM10	1,100	1,031	1,173
	SO2	1,160	1,025	1,313
PM10+O3	PM10	1,160	1,105	1,217
	O3	1,071	0,993	1,154
PM10+sec	PM10	1,151	1,048	1,265
	Sec	0,994	0,900	1,097
PM10+NO3	PM10	1,183	1,098	1,276
	NO3	0,972	0,910	1,038
PM10+SO4	PM10	1,200	1,099	1,311
	SO4	0,961	0,891	1,036
BS+NH3	BS	1,226	1,148	1,309
	NH3	0,940	0,889	0,993
BS+CO	BS	1,221	1,103	1,351
	CO	0,936	0,799	1,097
BS+NO	BS	1,266	1,170	1,370
	NO	0,886	0,803	0,977
BS+NO2	BS	1,114	1,032	1,203
	NO2	1,053	0,993	1,116
BS+SO2	BS	1,113	1,034	1,197
	SO2	1,145	1,007	1,302
BS+O3	BS	1,185	1,123	1,250
	O3	1,084	1,005	1,169
BS+sec	BS	1,140	1,058	1,230
	Sec	1,030	0,957	1,107
BS+NO3	BS	1,165	1,091	1,243
	NO3	1,008	0,957	1,062
BS+SO4	BS	1,176	1,093	1,266
	SO4	1,000	0,944	1,059
NH3+CO	NH3	0,957	0,906	1,011
	CO	1,235	1,118	1,363
NH3+NO	NH3	0,993	0,943	1,047
	NO	1,079	1,000	1,165
NH3+NO2	NH3	0,932	0,879	0,988
	NO2	1,145	1,089	1,204
NH3+SO2	NH3	0,988	0,940	1,038
	SO2	1,341	1,203	1,495
NH3+O3	NH3	1,019	0,971	1,069
	O3	1,042	0,963	1,128
NH3+sec	NH3	0,934	0,878	0,994
	Sec	1,160	1,079	1,247
NH3+NO3	NH3	0,948	0,893	1,007
	NO3	1,117	1,054	1,184
NH3+SO4	NH3	0,983	0,933	1,036
	SO4	1,097	1,043	1,153
CO+NO	CO	1,662	1,376	2,009
	NO	0,749	0,645	0,870
CO+NO2	CO	1,008	0,875	1,162
	NO2	1,114	1,040	1,194
CO+SO2	CO	1,109	1,008	1,219
	SO2	1,229	1,105	1,367
CO+O3	CO	1,221	1,122	1,328
	O3	1,082	1,003	1,168
CO+sec	CO	1,032	0,922	1,156
	Sec	1,105	1,032	1,184
CO+NO3	CO	1,140	1,030	1,261
	NO3	1,043	0,991	1,099
CO+SO4	CO	1,138	1,026	1,261
	SO4	1,049	0,997	1,104
NO+NO2	NO	0,926	0,848	1,012
	NO2	1,156	1,095	1,220
NO+SO2	NO	1,037	0,969	1,111
	SO2	1,281	1,163	1,412
NO+O3	NO	1,099	1,027	1,176
	O3	1,067	0,988	1,153
NO+sec	NO	0,976	0,903	1,054
	Sec	1,127	1,061	1,196
NO+NO3	NO	1,036	0,966	1,111
	NO3	1,077	1,030	1,126
NO+SO4	NO	1,032	0,960	1,109
	SO4	1,082	1,034	1,132
NO2+SO2	NO2	1,073	1,023	1,126
	SO2	1,186	1,061	1,326
NO2+O3	NO2	1,128	1,082	1,175
	O3	1,089	1,010	1,175
NO2+sec	NO2	1,070	1,015	1,127
	Sec	1,069	1,000	1,144
NO2+NO3	NO2	1,102	1,047	1,160
	NO3	1,023	0,970	1,079
NO2+SO4	NO2	1,096	1,047	1,147
	SO4	1,046	0,998	1,096
SO2+O3	SO2	1,311	1,195	1,439
	O3	1,067	0,990	1,150
SO2+sec	SO2	1,186	1,056	1,332
	Sec	1,074	1,008	1,145
SO2+NO3	SO2	1,245	1,123	1,380
	NO3	1,043	0,995	1,093
SO2+SO4	SO2	1,246	1,120	1,386
	SO4	1,047	0,999	1,098
O3+sec	O3	1,053	0,970	1,144
	Sec	1,124	1,062	1,189
O3+NO3	O3	1,059	0,982	1,141
	NO3	1,091	1,046	1,138
O3+SO4	O3	1,058	0,982	1,141
	SO4	1,095	1,051	1,141
Sec+NO3	Sec	1,093	0,941	1,270
	NO3	1,020	0,902	1,155
Sec+SO4	Sec	1,116	0,960	1,298
	SO4	1,002	0,884	1,135
NO3+SO4	NO3	1,042	0,976	1,113
	SO4	1,057	0,991	1,127

Pneumonia mortality

The results for pneumonia mortality are shown in Table A12. All pollutants were statistically associated with pneumonia mortality, while overall the highest RR were found for this association compared with total and other cause-specific mortality. Again, all statistically significant associations were above unity. RR for the four major cities and 'high traffic' regions were comparable with RR in non-urbanised and 'low traffic' regions. For both seasons, statistically significant associations were found. PM_{10} and BS were in both seasons associated with pneumonia mortality. For CO, NO and NO_2, **SO_2** and NO_3, most of the winter RR were statistically significant, while for NH3, secondary aerosol and SO_4 most of the summer RR were. Ozone and SO4 were only associated with pneumonia mortality during the summer season. In the two-pollutant model, PM_{10}, **SO_2** and BS were best associated with pneumonia mortality. NO was negatively associated with pneumonia mortality when a co-pollutant was added to the model, while ozone associations reduced to non-significance when co-pollutants were added.

Table A12 Association between air pollution and **pneumonia mortality** in different regions of the Netherlands. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders.

	NL			Cities			Non-urban			Traffic high			Traffic low		
Pollutant															
PM10 lag 1	1,121	1,065	1,180	1,115	0,999	1,245	1,113	1,051	1,179	1,124	1,029	1,227	1,144	1,029	1,272
PM10 avg	1,173	1,109	1,242	1,153	1,014	1,310	1,197	1,124	1,275	1,119	1,011	1,239	1,309	1,165	1,470
BS lag 1	1,147	1,082	1,215				1,136	1,065	1,211				1,141	1,010	1,288
BS avg	1,193	1,121	1,269				1,195	1,116	1,280				1,278	1,124	1,453
NH3 lag 1	1,116	1,050	1,186				1,133	1,059	1,211				1,086	0,971	1,216
NH3 avg	1,102	1,043	1,165				1,108	1,043	1,178				1,065	0,962	1,180
CO lag 1	1,224	1,131	1,325	1,224	1,054	1,421	1,267	1,135	1,415	1,254	1,114	1,412	1,376	1,125	1,683
CO avg	1,157	1,049	1,275	1,142	0,941	1,385	1,241	1,091	1,413	1,153	0,988	1,345	1,439	1,136	1,823
NO lag 1	1,104	1,041	1,171	1,070	0,984	1,163	1,112	1,026	1,205	1,103	1,033	1,178	1,155	0,986	1,353
NO avg	1,068	0,989	1,153	1,043	0,930	1,169	1,074	0,968	1,192	1,070	0,977	1,171	1,133	0,924	1,390
NO2 lag 1	1,141	1,083	1,202	1,070	0,968	1,182	1,163	1,095	1,235	1,107	1,023	1,198	1,224	1,093	1,371
NO2 avg	1,126	1,074	1,180	1,017	0,922	1,121	1,156	1,097	1,218	1,074	0,995	1,161	1,240	1,124	1,368
SO2 lag 1	1,195	1,079	1,324	1,178	1,003	1,384	1,141	1,013	1,286	1,214	1,068	1,379	1,277	1,030	1,583
SO2 avg	1,433	1,287	1,595	1,193	0,978	1,455	1,485	1,317	1,675	1,258	1,075	1,471	1,767	1,427	2,188
O3 lag 1	1,035	0,936	1,144	1,027	0,818	1,289	1,039	0,931	1,159	0,956	0,797	1,146	1,054	0,861	1,290
O3 avg	1,560	1,422	1,712	1,755	1,410	2,186	1,522	1,376	1,684	1,592	1,338	1,894	1,438	1,192	1,735
Sec lag 1	1,088	1,020	1,160				1,116	1,039	1,198				1,124	0,986	1,281
Sec avg	1,103	1,031	1,180				1,130	1,049	1,216				1,190	1,040	1,362
NO3 lag 1	1,051	0,998	1,107				1,071	1,011	1,134				1,099	0,989	1,220
NO3 avg	1,071	1,019	1,127				1,108	1,048	1,172				1,161	1,048	1,286
SO4 lag 1	1,052	1,000	1,107				1,067	1,008	1,128				1,074	0,969	1,191
SO4 avg	1,092	1,040	1,147				1,116	1,058	1,178				1,192	1,081	1,315

For both seasons, statistically significant associations were found. PM_{10} and BS were in both seasons associated with pneumonia mortality. For CO, NO and NO_2, **SO_2** and NO_3, most of the winter RR were statistically significant, while for NH3, secondary aerosol and SO_4 most of the summer RR were. Ozone and SO4 were only associated with pneumonia mortality during the summer season.

Table A13 Association between air pollution and **pneumonia mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Season-specific analysis.

	Summer			Winter		
	RR	RR-lo	RR-hi	RR	RR-lo	RR-hi
Pollutant						
PM10 lag 1	1,331	1,180	1,501	1,104	1,036	1,177
PM10 avg	1,971	1,710	2,271	1,115	1,039	1,195
BS lag 1	1,416	1,130	1,774	1,144	1,073	1,220
BS avg	2,490	1,909	3,247	1,156	1,078	1,239
NH3 lag 1	1,252	1,109	1,413	1,108	1,026	1,198
NH3 avg	1,487	1,329	1,664	1,034	0,964	1,109
CO lag 1	1,319	0,999	1,740	1,226	1,123	1,337
CO avg	1,546	1,085	2,203	1,162	1,044	1,295
NO lag 1	0,909	0,597	1,384	1,108	1,042	1,177
NO avg	0,495	0,272	0,902	1,082	0,999	1,172
NO2 lag 1	1,116	0,998	1,248	1,157	1,084	1,236
NO2 avg	1,303	1,162	1,461	1,096	1,032	1,165
SO2 lag 1	1,288	0,924	1,795	1,158	1,038	1,291
SO2 avg	2,381	1,601	3,541	1,353	1,210	1,514
O3 lag 1	1,249	1,116	1,399	0,762	0,654	0,886
O3 avg	2,219	1,982	2,485	1,035	0,894	1,198
Sec lag 1	1,153	1,011	1,315	1,117	1,031	1,210
Sec avg	1,442	1,241	1,676	1,046	0,964	1,135
NO3 lag 1	1,044	0,952	1,144	1,119	1,044	1,198
NO3 avg	1,170	0,975	1,405	1,085	1,017	1,157
SO4 lag 1	1,257	1,001	1,578	1,040	0,980	1,104
SO4 avg	1,601	1,158	2,215	1,033	0,977	1,092

In the two-pollutant model, PM_{10}, **SO_2** and BS were best associated with pneumonia mortality. NO was negatively associated with pneumonia mortality when a co-pollutant was added to the model, while ozone associations reduced to non-significance when co-pollutants were added.

Table A14 Association between air pollution and **pneumonia mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Two-pollutant models, weekly average concentrations.

Model	Pollut-ant	RR	RR-lo	RR-hi
PM10+BS	PM10	1,154	1,026	1,297
	BS	1,022	0,899	1,161
PM10+NH3	PM10	1,162	1,081	1,248
	NH3	1,029	0,963	1,100
PM10+CO	PM10	1,236	1,138	1,342
	CO	0,892	0,774	1,027
PM10+NO	PM10	1,212	1,135	1,295
	NO	0,925	0,846	1,012
PM10+NO2	PM10	1,127	1,044	1,217
	NO2	1,055	0,990	1,124
PM10+SO2	PM10	1,083	1,004	1,167
	SO2	1,301	1,129	1,500
PM10+O3	PM10	1,182	1,116	1,250
	O3	1,077	0,975	1,191
PM10+sec	PM10	1,306	1,165	1,464
	Sec	0,872	0,770	0,987
PM10+NO3	PM10	1,336	1,222	1,460
	NO3	0,878	0,811	0,951
PM10+SO4	PM10	1,311	1,177	1,459
	SO4	0,903	0,824	0,990
BS+NH3	BS	1,175	1,090	1,267
	NH3	1,038	0,974	1,106
BS+CO	BS	1,373	1,220	1,546
	CO	0,788	0,654	0,950
BS+NO	BS	1,329	1,212	1,457
	NO	0,842	0,752	0,944
BS+NO2	BS	1,123	1,027	1,228
	NO2	1,057	0,988	1,131
BS+SO2	BS	1,065	0,978	1,160
	SO2	1,346	1,161	1,560
BS+O3	BS	1,200	1,127	1,278
	O3	1,090	0,985	1,207
BS+sec	BS	1,149	1,052	1,256
	Sec	1,009	0,924	1,102
BS+NO3	BS	1,239	1,147	1,338
	NO3	0,966	0,907	1,029
BS+SO4	BS	1,224	1,122	1,336
	SO4	0,979	0,915	1,049
NH3+CO	NH3	1,086	1,019	1,157
	CO	1,044	0,929	1,172
NH3+NO	NH3	1,109	1,044	1,177
	NO	0,973	0,890	1,063
NH3+NO2	NH3	1,048	0,980	1,121
	NO2	1,082	1,021	1,148
NH3+SO2	NH3	1,069	1,010	1,131
	SO2	1,522	1,346	1,721
NH3+O3	NH3	1,103	1,043	1,165
	O3	1,040	0,937	1,155
NH3+sec	NH3	1,025	0,955	1,101
	Sec	1,120	1,031	1,217
NH3+NO3	NH3	1,090	1,017	1,169
	NO3	1,025	0,956	1,098
NH3+SO4	NH3	1,072	1,010	1,139
	SO4	1,102	1,040	1,167
CO+NO	CO	1,518	1,214	1,897
	NO	0,785	0,659	0,936
CO+NO2	CO	0,869	0,734	1,028
	NO2	1,192	1,099	1,292
CO+SO2	CO	1,024	0,916	1,144
	SO2	1,415	1,252	1,599
CO+O3	CO	1,167	1,056	1,288
	O3	1,063	0,960	1,177
CO+sec	CO	0,976	0,855	1,114
	sec	1,108	1,021	1,202
CO+NO3	CO	1,134	1,006	1,278
	NO3	1,029	0,967	1,095
CO+SO4	CO	1,100	0,974	1,241
	SO4	1,061	0,999	1,127
NO+NO2	NO	0,879	0,792	0,976
	NO2	1,189	1,116	1,266
NO+SO2	NO	1,008	0,931	1,092
	SO2	1,428	1,277	1,597
NO+O3	NO	1,069	0,988	1,157
	O3	1,050	0,947	1,164
NO+sec	NO	0,931	0,851	1,018
	sec	1,123	1,046	1,205
NO+NO3	NO	1,026	0,945	1,113
	NO3	1,064	1,009	1,123
NO+SO4	NO	1,009	0,928	1,097
	SO4	1,089	1,033	1,149
NO2+SO2	NO2	1,056	1,000	1,116
	SO2	1,348	1,188	1,530
NO2+O3	NO2	1,130	1,078	1,186
	O3	1,092	0,986	1,209
NO2+sec	NO2	1,102	1,037	1,170
	Sec	1,031	0,951	1,117
NO2+NO3	NO2	1,145	1,079	1,215
	NO3	0,984	0,923	1,049
NO2+SO4	NO2	1,116	1,058	1,177
	SO4	1,039	0,983	1,099
SO2+O3	SO2	1,460	1,310	1,626
	O3	1,092	0,987	1,209
SO2+sec	SO2	1,403	1,228	1,603
	sec	1,018	0,942	1,100
SO2+NO3	SO2	1,463	1,298	1,649
	NO3	1,002	0,947	1,060
SO2+SO4	SO2	1,459	1,288	1,654
	SO4	1,019	0,963	1,078
O3+sec	O3	0,998	0,895	1,112
	sec	1,097	1,026	1,174
O3+NO3	O3	1,048	0,948	1,159
	NO3	1,075	1,022	1,131
O3+SO4	O3	1,050	0,949	1,162
	SO4	1,095	1,043	1,150
Sec+NO3	Sec	1,137	0,953	1,356
	NO3	0,973	0,841	1,126
Sec+SO4	Sec	0,993	0,829	1,189
	SO4	1,098	0,949	1,271
NO3+SO4	NO3	1,010	0,934	1,091
	SO4	1,083	1,005	1,167

Cardiovascular mortality

The associations between daily cardiovascular mortality and air pollution for different regions in the Netherlands is shown in Table A15. For all pollutants except NH_3, statistically significant associations were found with cardiovascular mortality. The magnitudes of the RR are comparable with the RR for total mortality. In cities and 'high traffic' regions substantially less significant associations were found compared with rural and 'low traffic' regions. In both seasons, statistically significant associations were found for several pollutants, but the associations for the summer period were more consistent. NO, **SO_2**, **O_3** and secondary aerosols were not associated with cardiovascular mortality in the winter season. NO_3 and SO_4 associations disappeared when the analyses were stratified by season. In two-pollutant models NO_2, **SO_2** and **O_3** were most consistently associated with cardiovascular mortality, while NO and secondary aerosols were least stable in the two-pollutant models.

Table A15 Association between air pollution and **cardiovascular mortality** in different regions of the Netherlands. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders.

	NL			Cities			Non-urban			Traffic high			Traffic low		
Pollutant															
PM10 lag 1	1,025	1,009	1,042	0,991	0,955	1,028	1,030	1,012	1,048	1,009	0,981	1,037	1,044	1,013	1,076
PM10 avg	1,036	1,018	1,055	1,019	0,977	1,063	1,039	1,019	1,059	1,028	0,996	1,062	1,042	1,007	1,078
BS lag 1	1,032	1,014	1,052				1,035	1,014	1,055				1,047	1,010	1,084
BS avg	1,055	1,034	1,076				1,059	1,037	1,081				1,062	1,024	1,103
NH3 lag 1	1,015	0,995	1,035				1,011	0,990	1,033				1,029	0,996	1,064
NH3 avg	1,000	0,982	1,017				0,997	0,978	1,016				1,005	0,976	1,036
CO lag 1	1,044	1,018	1,071	1,014	0,963	1,067	1,052	1,017	1,089	1,025	0,986	1,066	1,087	1,025	1,153
CO avg	1,068	1,036	1,101	1,033	0,969	1,101	1,088	1,046	1,132	1,051	1,001	1,103	1,120	1,046	1,199
NO lag 1	1,017	0,998	1,037	1,003	0,974	1,032	1,020	0,995	1,046	1,010	0,988	1,033	1,018	0,972	1,068
NO avg	1,032	1,008	1,058	1,018	0,980	1,057	1,037	1,004	1,070	1,025	0,996	1,055	1,057	0,996	1,121
NO2 lag 1	1,029	1,012	1,047	1,039	1,005	1,074	1,024	1,005	1,043	1,023	0,997	1,049	1,051	1,017	1,087
NO2 avg	1,049	1,034	1,065	1,048	1,015	1,082	1,048	1,031	1,065	1,037	1,012	1,062	1,079	1,048	1,110
SO2 lag 1	1,029	0,996	1,062	1,008	0,956	1,062	1,022	0,986	1,060	1,046	1,005	1,089	1,042	0,980	1,109
SO2 avg	1,105	1,068	1,144	1,045	0,978	1,116	1,118	1,077	1,162	1,055	1,004	1,109	1,121	1,050	1,196
O3 lag 1	1,032	1,003	1,061	1,007	0,941	1,076	1,031	1,001	1,062	1,013	0,962	1,065	1,021	0,970	1,075
O3 avg	0,997	0,969	1,026	1,027	0,956	1,102	0,990	0,960	1,021	0,993	0,941	1,047	0,981	0,931	1,035
Sec lag 1	1,035	1,015	1,055				1,036	1,016	1,058				1,042	1,005	1,080
Sec avg	1,021	1,001	1,041				1,023	1,001	1,045				1,022	0,984	1,062
NO3 lag 1	1,025	1,009	1,041				1,029	1,012	1,046				1,047	1,017	1,078
NO3 avg	1,017	1,001	1,033				1,017	1,000	1,034				1,024	0,995	1,053
SO4 lag 1	1,018	1,002	1,033				1,023	1,007	1,040				1,033	1,004	1,063
SO4 avg	1,014	0,999	1,029				1,016	1,000	1,033				1,012	0,985	1,041

In both seasons, statistically significant associations were found for several pollutants, but the associations for the summer period were more consistent. NO, **SO_2**, **O_3** and secondary aerosols were not associated with cardiovascular mortality in the winter season. NO_3 and SO_4 associations disappeared when the analyses were stratified by season.

Table A16 Association between air pollution and **cardiovascular mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Season-specific analysis.

	Summer			Winter		
	RR	RR-lo	RR-hi	RR	RR-lo	RR-hi
Pollutant						
PM10 lag 1	1,049	1,011	1,089	1,020	0,999	1,041
PM10 avg	1,119	1,070	1,169	1,023	1,000	1,046
BS lag 1	1,106	1,033	1,184	1,026	1,004	1,047
BS avg	1,348	1,242	1,462	1,033	1,011	1,056
NH3 lag 1	0,971	0,935	1,008	1,034	1,006	1,062
NH3 avg	0,994	0,959	1,029	0,996	0,973	1,020
CO lag 1	1,165	1,074	1,264	1,029	1,000	1,059
CO avg	1,380	1,243	1,532	1,037	1,002	1,074
NO lag 1	1,086	0,966	1,222	1,013	0,992	1,033
NO avg	1,552	1,317	1,830	1,016	0,990	1,042
NO2 lag 1	1,032	0,998	1,067	1,024	1,002	1,047
NO2 avg	1,116	1,078	1,156	1,028	1,008	1,049
SO2 lag 1	1,043	0,948	1,149	1,024	0,989	1,061
SO2 avg	1,328	1,183	1,491	1,075	1,037	1,116
O3 lag 1	1,054	1,020	1,089	1,011	0,965	1,059
O3 avg	1,041	1,005	1,079	0,978	0,933	1,025
Sec lag 1	1,031	0,991	1,073	1,035	1,011	1,060
Sec avg	1,061	1,014	1,111	1,009	0,985	1,034
NO3 lag 1	1,023	0,986	1,061	1,035	1,012	1,057
NO3 avg	1,027	0,985	1,070	1,018	0,998	1,039
SO4 lag 1	1,032	0,984	1,083	1,013	0,995	1,031
SO4 avg	1,040	0,986	1,096	1,009	0,992	1,026

In two-pollutant models, NO_2, **SO_2** and **O_3** were most consistently associated with cardiovascular mortality, while NO and secondary aerosols were least stable in the two-pollutant models.

Table A17. Association between air pollution and **cardiovascular mortality**. Relative risk (RR) and 95% confidence interval (RR-lo, RR-hi) for a relevant change in the pollutant concentration, adjusted for confounders. Two-pollutant models, weekly average concentrations.

Model	Pollut-ant	RR	RR-lo	RR-hi
PM10+BS	PM10	0,957	0,922	0,993
	BS	1,106	1,062	1,152
PM10+NH3	PM10	1,047	1,024	1,071
	NH3	0,979	0,958	1,000
PM10+CO	PM10	1,010	0,984	1,037
	CO	1,055	1,009	1,104
PM10+NO	PM10	1,033	1,011	1,055
	NO	1,008	0,979	1,038
PM10+NO2	PM10	0,990	0,966	1,014
	NO2	1,056	1,034	1,077
PM10+SO2	PM10	1,005	0,982	1,030
	SO2	1,101	1,051	1,154
PM10+O3	PM10	1,040	1,022	1,058
	O3	1,040	1,012	1,070
PM10+sec	PM10	1,068	1,032	1,105
	Sec	0,971	0,937	1,005
PM10+NO3	PM10	1,047	1,019	1,077
	NO3	0,987	0,964	1,011
PM10+SO4	PM10	1,060	1,027	1,095
	SO4	0,975	0,949	1,002
BS+NH3	BS	1,074	1,049	1,100
	NH3	0,973	0,954	0,993
BS+CO	BS	1,056	1,017	1,098
	CO	0,997	0,940	1,058
BS+NO	BS	1,081	1,049	1,114
	NO	0,963	0,928	1,000
BS+NO2	BS	1,008	0,979	1,037
	NO2	1,045	1,023	1,068
BS+SO2	BS	1,027	0,999	1,055
	SO2	1,072	1,021	1,125
BS+O3	BS	1,061	1,041	1,082
	O3	1,048	1,019	1,078
BS+sec	BS	1,078	1,048	1,108
	Sec	0,976	0,951	1,001
BS+NO3	BS	1,065	1,039	1,091
	NO3	0,987	0,969	1,006
BS+SO4	BS	1,084	1,054	1,114
	SO4	0,972	0,952	0,993
NH3+CO	NH3	0,979	0,960	0,999
	CO	1,076	1,038	1,115
NH3+NO	NH3	0,992	0,973	1,011
	NO	1,026	0,998	1,055
NH3+NO2	NH3	0,960	0,940	0,981
	NO2	1,065	1,046	1,085
NH3+SO2	NH3	0,988	0,970	1,006
	SO2	1,155	1,110	1,202
NH3+O3	NH3	1,000	0,983	1,018
	O3	1,029	1,000	1,060
NH3+sec	NH3	0,981	0,959	1,003
	Sec	1,033	1,007	1,060
NH3+NO3	NH3	0,989	0,967	1,011
	NO3	1,016	0,995	1,037
NH3+SO4	NH3	0,995	0,976	1,014
	SO4	1,011	0,993	1,029
CO+NO	CO	1,192	1,112	1,278
	NO	0,907	0,858	0,959
CO+NO2	CO	0,966	0,917	1,017
	NO2	1,065	1,038	1,092
CO+SO2	CO	1,034	0,998	1,071
	SO2	1,084	1,041	1,129
CO+O3	CO	1,078	1,045	1,112
	O3	1,047	1,018	1,077
CO+sec	CO	1,088	1,044	1,134
	sec	0,990	0,966	1,015
CO+NO3	CO	1,068	1,028	1,108
	NO3	0,997	0,978	1,016
CO+SO4	CO	1,075	1,034	1,117
	SO4	0,992	0,973	1,011
NO+NO2	NO	0,961	0,930	0,993
	NO2	1,068	1,047	1,089
NO+SO2	NO	1,011	0,985	1,037
	SO2	1,099	1,060	1,141
NO+O3	NO	1,039	1,014	1,065
	O3	1,041	1,012	1,072
NO+sec	NO	1,027	0,998	1,056
	sec	1,012	0,991	1,034
NO+NO3	NO	1,022	0,996	1,049
	NO3	1,011	0,995	1,028
NO+SO4	NO	1,026	0,999	1,054
	SO4	1,006	0,990	1,023
NO2+SO2	NO2	1,036	1,018	1,055
	SO2	1,053	1,010	1,098
NO2+O3	NO2	1,055	1,039	1,071
	O3	1,053	1,024	1,083
NO2+sec	NO2	1,067	1,047	1,087
	Sec	0,979	0,956	1,003
NO2+NO3	NO2	1,060	1,040	1,080
	NO3	0,981	0,962	1,000
NO2+SO4	NO2	1,053	1,035	1,071
	SO4	0,990	0,974	1,007
SO2+O3	SO2	1,112	1,074	1,151
	O3	1,043	1,015	1,073
SO2+sec	SO2	1,110	1,064	1,159
	sec	0,996	0,973	1,019
SO2+NO3	SO2	1,101	1,059	1,145
	NO3	1,000	0,983	1,017
SO2+SO4	SO2	1,111	1,067	1,157
	SO4	0,995	0,978	1,012
O3+sec	O3	1,017	0,987	1,048
	sec	1,022	1,002	1,043
O3+NO3	O3	1,038	1,010	1,068
	NO3	1,019	1,004	1,035
O3+SO4	O3	1,036	1,007	1,065
	SO4	1,016	1,001	1,031
Sec+NO3	Sec	1,013	0,960	1,069
	NO3	1,007	0,963	1,052
Sec+SO4	Sec	1,069	1,014	1,128
	SO4	0,959	0,917	1,002
NO3+SO4	NO3	1,017	0,993	1,041
	SO4	1,001	0,978	1,025

ATMOSPHERIC SECONDARY INORGANIC PARTICULATE MATTER:

THE TOXICOLOGICAL PERSPECTIVE AS A BASIS FOR HEALTH EFFECTS RISK ASSESSMENT

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ABSTRACT

Numerous epidemiological studies have provided evidence for an association between exposure to ambient particulate matter and increased mortality and morbidity. However, the exact physicochemical nature of the responsible components are not as yet clear. One major constituent of the fine mode of the ambient aerosol is secondary inorganic particles. These are produced in the atmosphere via chemical reactions involving gaseous pollutant precursors, and are dominated by such chemical species as acidic sulfates and nitrates, and their salts. This document reviews the health effects resulting from exposure to secondary inorganic aerosol constituents. It was developed based upon available data from peer review published papers as well as publicly available reports on controlled animal and human clinical exposure studies involving these chemical species. The aim is to provide a basis for addressing the issue of whether ambient concentrations of these secondary aerosols could be contributory to human health effects associated with exposure to ambient particulate matter. Evaluation of the toxicological database does not support a role for ambient acidic sulfates in adverse health outcomes noted in epidemiological studies. Levels of acidic sulfates needed to produce any effect in controlled studies are well above those found in ambient air, especially in the Netherlands. Due to significant neutralization by ambient ammonia, particulate matter in the Netherlands has relatively low acidity, with the major components consisting of ammonium sulfate and ammonium nitrate. Toxicological studies of these constituents of the secondary fraction, as wells as metal sulfates, suggest that these have little toxic potency at environmentally relevant levels in normal humans or animals or in the limited compromised animal models studied. This document also addressed the toxicity of sea salt spray; the available database suggests that ambient levels of sodium chloride are well below those shown to produce any adverse effects in toxicological studies. There are, however, some caveats in this overall evaluation of potential health effects from exposure to ambient secondary aerosols. It is important to consider the relationship between animal exposure studies and actual human exposures, both in terms of particle size and inhaled dose. It is also necessary to consider the physicochemical characteristics of chemical species in the ambient air compared to those used in controlled studies. Finally, the potential for interactions between particulates and ambient gases must be considered in developing conclusions as to effective levels of the former.

LIST OF ABBREVIATIONS

d	days
DLco	diffusing capacity for carbon monoxide
FEV_1	forced expiratory volume
FRC	functional residual capacity
FVC	forced vital capacity
hr	hours
min	minutes
MMAD	mass median aerodynamic diameter
mo	month
NOAEL	no observed adverse effect level
PEFR	peak expiratory flow rate
PM2.5	particles having aerodynamic diameters < 2.5 μm
PM10	particles having aerodynamic diameters < 10 μm
RV	residual volume
TLC	total lung capacity
VC	vital capacity
wk	week

INTRODUCTION

Evidence for an association between exposure to ambient particulate matter and increased mortality and morbidity has been provided by numerous epidemiological studies (e.g., Dockery et al., 1993; Pope et al., 1995; Vonk and Schouten, 1998; Samet et al., 2000; Schwartz, 2000; Dab et al., 2001). However, the exact physicochemical nature of the responsible component or components is not as yet clear. This is partly due to the fact that this association has been noted in various regions of the world, having various pollutant sources resulting in various particulate mixes.

Atmospheric particulate matter may be divided into two broad classes depending upon the manner in which it becomes introduced into the atmosphere. A primary component consists of those particles that are released into the air directly from their source, while a secondary component consists of those particles that are produced within the atmosphere via chemical reactions involving gaseous pollutant precursors (gas-to-particle conversion). Primary and secondary aerosols, both of which can derive from natural or anthropogenic sources, may contain both organic and inorganic chemical species.

This document focuses solely on secondary inorganic aerosols. It provides an overview of the health effects resulting from exposure to these constituents of the ambient pollutant mix. It was developed based upon available data from peer reviewed published papers as well as publicly available reports on controlled animal and human clinical exposure studies. These toxicological studies have been employed in attempts to evaluate the physicochemical properties of particulate matter in relationship to health outcomes observed in epidemiological studies, as well as to provide an understanding of the underlying biological mechanism(s) for these outcomes. The aim of the document is to provide a basis for addressing the issue of whether ambient concentrations of these secondary aerosols could be contributory to human health effects associated with exposure to atmospheric particulate matter in the Netherlands. While not a secondary pollutant. sea salt aerosols are also discussed herein, since that they can contribute to a significant background aerosol component.

In general, the studies evaluated were limited to those performed using particulate concentrations at or below 1 mg/m^3 although studies performed with higher levels may be discussed in the context of relative toxicity or toxicological mechanisms. However, what must always be kept in perspective is the relationship between exposures in the controlled studies and levels of ambient exposure, since the nature and temporal relationship of health effects can differ at different concentrations (Roth et al., 1998).

PHYSICOCHEMICAL PROPERTIES OF SECONDARY INORGANIC PARTICULATE MATTER

Ambient particulate matter is categorized into specific defined size ranges, or modes (Whitby, 1978; Hinds, 1999). The fine mode consists of those particles having aerodynamic diameters < 2.5 μm. This mode actually consists of two subclasses, the nuclei (or ultrafine) mode, containing particles having diameters < 0.1 μm, and the accumulation mode, containing particles extending from 2.5 μm to 0.1 μm. The former consists largely of primary combustion particles and particles formed within the atmosphere by gas-to-particle conversion processes. While the total mass of these particles in the ambient mix may be low, they exist at high number concentrations and, thus, they tend to coagulate rapidly with each other as well as with larger particles in the accumulation mode.

The accumulation mode includes primary combustion particles, additional secondary inorganic particles and coagulated ultrafine particles. Under conditions of high humidity, such as in fogs, the accumulation mode may itself have two submodes, a condensation mode, containing particles having diameters of 0.2–0.3 μm, and a droplet mode, containing particles with diameters of 0.5–0.8 μm. The droplets are formed by the growth of hygroscopic condensation mode particles, which can be facilitated by chemical reactions occurring within these droplets.

Particles having diameters > 2.5 μm exist within the coarse particle mode; this contains windblown dust, large salt particles from sea spray, mechanically generated anthropogenic particles and some secondary aerosols. These secondary aerosols within the coarse mode of the ambient aerosol generally are found within naturally occurring fogs. From a size selective sampling perspective, PM10 is the designation for the complete spectrum of ambient particulate matter of concern to public health, the fine particle mode is designated as PM2.5, and the coarse particle mode as PM10-2.5.

Most of the mass of the ambient secondary inorganic aerosol is contained within the accumulation mode portion of the fine mode of atmospheric particulate matter, and consists largely of sulfate, ammonium and nitrate (Wilson and Suh, 1997; Lippmann and Schlesinger, 2000). Sulfates and metals can also exist within the ultrafine mode (Wilson and Suh, 1997), and about 10–30% of the total mass of sulfates and nitrates within the fine mode can also be found in the coarse mode (Visser et al., 2001).

Secondary aerosol formation can occur by gas phase or aqueous phase reactions, and generally involve either of three generic processes: nucleation, in which gas molecules come together to form a particle; condensation of gases onto an existing particle; or the reaction of adsorbed gases within liquid droplets to form a new chemical species. The formation of secondary aerosols depends upon a number of factors, which include the concentrations of precursors, the concentration of other reactive chemical species and meteorological conditions, such as the intensity of solar radiation and relative humidity.

Sulfate particles result from reactions involving the gas phase conversion of sulfur dioxide (SO_2) to sulfuric acid (H_2SO_4) by hydroxyl ($OH\bullet$) radicals produced within the photochemical smog cycle, or the aqueous phase reactions of sulfur dioxide with hydrogen peroxide (H_2O_2), ozone (O_3) or oxygen (O_2), catalyzed by iron or manganese, metals which are commonly within the fine fraction of ambient particulate matter (Costa, 2001). These heterogeneous reactions may occur within atmospheric water droplets, or within thin aqueous films present on solid atmospheric particles. As sulfuric acid is formed, it can nucleate to form additional ultrafine particles, or it can condense onto existing ultrafine or accumulation mode particles (Clark and Whitby, 1975; Whitby, 1978). The nitrogen dioxide (NO_2) portion of ambient nitrogen oxides (NOx) can be converted to gaseous nitric acid (HNO_3) by reaction with hydroxyl radicals during the day, while at night, nitrogen oxides can be oxidized to gaseous nitric acid by a sequence of reactions initiated by ozone and that involves nitrate radicals and dinitrogen pentoxide.

Both sulfuric acid and nitric acid formed initially can subsequently react with atmospheric gaseous ammonia (NH_3). The ammonia reacts with gaseous nitric acid to form particulate ammonium nitrate (NH_4NO_3), while it reacts with sulfuric acid to form ammonium bisulfate (NH_4HSO_4) and ammonium sulfate [$(NH_4)_2SO_4$].

Fine mode secondary aerosols can have significant long-range transport, traveling up to thousands of kilometers from their site of formation (Wilson and Suh, 1997). Because the oxidation rate of sulfur dioxide is fairly slow, spatial gradients for secondary sulfate particles may be quite small on the scale of tens of kilometers, but can be quite large on the scale of hundreds or thousands of kilometers (Burton et al., 1996; Harrison and Yin, 2000). On the other hand, oxidation of nitrogen dioxide proceeds at a faster rate and, due to such factors as the sensitivity of ammonium nitrate concentrations to ambient ammonia levels which influence dissociation and the volatility of nitrate aerosols, the spatial patterns of secondary nitrates are considerably less uniform than those observed for sulfates (Harrison and Yin, 2000; Walsh and Gilliland, 2001).

Inorganic secondary acidic aerosols within the coarse mode of the ambient aerosol are components of naturally occurring fogs in some regions of the world. These acidic fogs are produced via the reaction of sulfur or nitrogen oxides with atmospheric water droplets, and the most common acidic components are, therefore, sulfuric and nitric acids. In addition, acidic gases, such as nitric acid vapor and sulfur dioxide, may react with coarse particles to form coarse secondary particulate matter also containing sulfate and nitrate. Examples of this include reactions with basic compounds, resulting in neutralization (e.g., the formation of calcium nitrate from the reaction of calcium carbonate with nitric acid) or with salts of volatile acids, resulting in the release of volatile acids [e.g., the production of sodium sulfite and hydrogen chloride by the reaction of sulfur dioxide, sodium chloride (NaCl) and water].

AMBIENT CONCENTRATIONS OF SECONDARY INORGANIC PARTICULATE MATTER

There are strong consistencies in the bulk chemical components of ambient particulate matter in developed countries throughout the world, and the concentrations at which they occur are broadly comparable; this is especially true for components of the secondary aerosol, such as sulfates, nitrates and ammonium (Harrison and Yin, 2000).

In the Netherlands, about 80% of the secondary inorganic aerosol is found within the fine mode of PM10 (Visser et al., 2001). The annual average PM10 mass concentration ranges from 27–42 $\mu g / m^3$ and, of this, the secondary inorganic particulate component accounts for about 10–14 $\mu g / m^3$ (Keuken et al., 1999; Weijers et al., 2000; Bloemen et al., 2001). Since ambient particulate matter in the Netherlands has relatively low acidity, due to the extent of neutralization by ambient ammonia, the major secondary components are ammonium sulfate and ammonium nitrate.

As a comparison, in the United States, the national average PM10 level is about 24 $\mu g / m^3$. Annual mean PM10 concentrations in urban areas are generally > 20 $\mu g / m^3$ (Fitz-Simons et al., 2000), but averages > 50 $\mu g / m^3$ have been found in some regions, such as in some western states (e.g., CA, NV, AZ, TX); this is likely due to an increased contribution from the coarse mode. Annual average PM10 concentration in western rural or remote areas is about 3 $\mu g / m^3$ while that in eastern rural or remote sites is about 5–10 $\mu g / m^3$.

In the United States as well as in the Netherlands, a substantial fraction of the fine particle mass is secondary sulfate and nitrate, especially during the warmer months of the year. However, in the United States, the average percentage of total particulate mass contributed by secondary sulfates and nitrates is regionally variable. For example, it can range from about 4–9% of PM2.5 in areas in Nevada, between 28–42% in California, and between 60–90% in some northeastern urban areas (U.S. EPA, 2002). Furthermore, there are also are clear regional

differences in the relative contribution of sulfate and nitrate to the ambient particulate mode. Particulate matter in the eastern part of the nation (i.e., east of the Mississippi River) tends to be more acidic, and sulfate is a larger fraction of particulate mass than is nitrate. On the other hand, in the western United States, where higher ammonia levels (as in the Netherlands) and lower sulfur dioxide emissions allow for more complete neutralization of sulfuric acid, the concentration of nitrate may be higher than that of sulfate (U.S. EPA, 2002). Thus, secondary sulfate is the major component of PM2.5 in the eastern United States, while in the western region there is a larger contribution from secondary ammonium nitrate. However, as levels of sulfur dioxide emissions decrease in the eastern United States, the ammonia remaining in the atmosphere after reaction with sulfuric acid can react with gaseous nitric acid, forming additional particulate ammonium nitrate in eastern regions as well.

It is not surprising that due to these geographical differences, the nature of acidic fogs in the United States is also regionally distinct. The acidic component of the ambient atmosphere in the western part of the United States can contain significant amounts of nitric acid (Aris et al., 1993; Pierson and Brachaczek, 1988), while that in the eastern part contains more sulfur oxides. In the Netherlands, the generally low level of acidity in the ambient particulate mode would suggest a low incidence of any acid fogs.

In terms of the specific contributions of sulfates and nitrates, sulfates represents 44–56% and 11–33% of total PM2.5 mass in the eastern and western United States, respectively, while nitrates account for 1–5% and 8–15% in these respective areas (U.S. EPA, 1996; IMPROVE network, 1998). While these values may not all represent secondary aerosols, most sulfates are of secondary origin, as are a large fraction of nitrates. Annual mean PM2.5 concentrations in the United States range from 5–> 20 $\mu g / m^3$, with the low values occurring in small relatively non-industrial cities and the larger values found in larger urban areas. The 98th percentile 24 hr average concentrations are generally < 65 $\mu g / m^3$ with most areas having levels > 65 $\mu g / m^3$ located within California (Fitz-Simons et al., 2000). In general, maximum PM2.5 concentrations in the eastern United States occur during the summer months, while in the western regions, higher levels occur during the autumn or winter (Fitz-Simons et al., 2000). Exceptions, however, do occur. Furthermore, sulfate and nitrate tend to have opposing seasonal patterns and show inversely related maxima and minima (Walsh and Gilliland, 2001).

It needs to be mentioned that actual human exposures depend upon specific environment-activity patterns, and that ambient concentrations of secondary aerosols are not necessarily exposure concentrations. For example, the concentration of some aerosols would be reduced indoors compared to outdoors, and acidity of some aerosols may also be reduced by indoor ammonia sources. In the United States, for example, as well as in most industrialized nations, people spend over 80% of their time indoors (Robinson and Nelson, 1998); this includes time at work, at home and in their automobiles or public transportation. However, this average can vary with age, with adults generally spending more time indoors than children. While outdoor air can enter indoor environments, the extent to which this occurs depends upon a number of factors. The ratio of ambient particulate matter indoors to that outdoors, called the infiltration factor, depends upon the air exchange rate between these two environments and the penetration efficiency and deposition or removal rate, which is a function of particle size. For fine particles in the accumulation mode, the infiltration rate is maximum, with the infiltration factor ranging from 0.4–0.7 (Costa, 2001); this factor would decrease for both smaller and larger particles beyond this range.

TOXICOLOGY OF PARTICULATE SULFUR OXIDES

The particulate phase sulfur oxides consist of strongly to weakly acidic sulfates, namely sulfuric acid (H_2SO_4) and its products of neutralization with atmospheric ammonia, namely letovicite [$(NH_4)_3H(SO_4)_2$], ammonium bisulfate (NH_4HSO_4) and ammonium sulfate [$(NH_4)_2SO_4$] (Schlesinger and Jaspers, 1997).

Sulfates may exist in ambient air as pure aqueous or solid particles, or as a surface layer on other solid particles. This latter type of aerosol is formed by sulfuric acid adsorption onto or formation on particles having large surface to volume ratios, such as typical carbonaceous or fly ash particles. Most toxicological studies have evaluated the health effects from pure sulfate particles and, thus, very little information is available concerning biological responses to this other physicochemical form in which ambient sulfates may actually occur. Furthermore, most of the toxicological database for ambient sulfates involves sulfuric acid, although this chemical species rarely occurs alone in ambient air, which generally contains some combination of acid and totally or partially neutralized sulfate salts.

SULFURIC ACID

Fine Mode Particles

This section discusses the biological responses observed due to exposure to the fine mode, generally accumulation, size of sulfuric acid particles. Due to the magnitude of the available database, the discussion is arranged by broad response endpoint.

Pulmonary Mechanical Function

Controlled exposure studies of healthy human adults have shown no consistent effects on pulmonary function or respiratory symptoms, e.g., cough, with acute exposures to sulfuric acid aerosols (0.1–1 µm diameter) at concentrations 1,000 $\mu g / m^3$, even with exercise (Stacy et al., 1983; Avol et al., 1988a; U.S. EPA, 1989; Frampton et al., 1992; Linn et al., 1994). On the other hand, there is some evidence that asthmatics may be more sensitive than healthy individuals to effects on lung mechanical function and that they may experience modest bronchoconstriction following exposure to sulfuric acid particles at concentrations < 1,000 $\mu g / m^3$ (Avol et al., 1988a; Linn et al., 1989; U.S. EPA, 1989; Koenig et al., 1993).

However, all asthmatics may not be equally sensitive to sulfuric acid exposure. While young adult or elderly asthmatics do not appear to be an especially susceptible population (Anderson et al., 1992; Koenig et al., 1993; Linn et al., 1994), adolescents with allergic asthma may be more so. A small decline in FEV_1 and an increase in total respiratory resistance, both of which were statistically significant, were noted following a 40 min exposure to sulfuric acid (1.75 µm MMAD) at 68 $\mu g / m^3$ (Koenig et al., 1989), although effects on lung function have not been consistently observed at exposure concentrations 100 $\mu g / m^3$ (Avol et al., 1979; Utell et al., 1983a). It is likely that the specific characteristics of the asthmatic population studied plays a significant role in the observed sensitivity to acid exposure and the resultant effective concentration. Furthermore, there may be a tendency for the more sensitive asthmatics to respond only when exposed during exercise rather than at rest (Koenig et al., 1983). In any case, epidemiological studies have not as yet identified asthmatics as a specific population at risk to acidic sulfate aerosols.

The effects of exposure to sulfuric acid on healthy humans are consistent with those observed in animal toxicology assessments, in that acute exposures of most species to concentrations 1,000 $\mu g/m^3$ (particles < 1 µm diameter) have generally not been shown to alter standard lung function tests (U.S. EPA, 1989). The notable exception is the guinea pig, which appears to be especially sensitive and does show various pulmonary functional changes at exposure levels as low as 100 $\mu g/m^3$.

While standard measures of lung function generally will not be affected in normal animals or humans, an increase in airway responsiveness with bronchoprovocation challenge was observed following exposure to 1,000 $\mu g/m^3$ sulfuric acid aerosol for 16 min, and in some adult asthmatics following exposure to as low as 100 $\mu g/m^3$ (Utell et al., 1983b); there was also some indication of a delayed response to exposure. Furthermore, it has been suggested that the degree of baseline nonspecific airway responsiveness in asthmatics may predict responsiveness to acid aerosols (Utell et al., 1983b; Hanley et al., 1992), i.e., the response to a constrictor agent was similar to that resulting from exposure to sulfuric acid. However, there appears to be no consistent effect of acute exposure to sulfuric acid particles on airway responsiveness in either healthy or asthmatic individuals, in that other studies have failed to show alterations in responsiveness with submicrometer particles (Linn et al., 1989; Anderson et al., 1992; Linn et al., 1994). As noted, differences in the various studies with asthmatics may have been influenced by differences in the subject population, the extent of disease and the use of medication, as well as differences in the characteristics of the exposure atmospheres.

The ability of sulfuric acid to alter airway responsiveness has been assessed in a number of studies using various animals. While acute exposure of healthy guinea pigs has shown inconsistent results, with effective concentrations ranging from 200 $\mu g/m^3$ to > 1,000 $\mu g/m^3$ (Silbaugh et al., 1981; Kobayashi and Shinozaki, 1993), chronic exposure (1 hr/d, 5 d/wk for 4 mo) of healthy rabbits to 250 $\mu g/m^3$ (0.3 µm MMAD) induced nonspecific airway hyperresponsiveness (Gearhart and Schlesinger, 1986). Furthermore, a single exposure of rabbits to 75 $\mu g/m^3$ sulfuric acid was shown to induce nonspecific hyperresponsiveness in an *in vitro* airway preparation (El-Fawal and Schlesinger, 1994). Thus, it appears that the ability of sulfuric acid to induce airway hyperresponsiveness may depend both upon the exposure regime and the animal species used.

Pulmonary Defenses

Sulfuric acid has can affect various aspects of pulmonary defenses. Acute exposures at levels as low as 100 $\mu g/m^3$ (0.3–0.6 µm diameter) have altered bronchial mucociliary transport in normal humans (Leikauf et al., 1981; Spektor et al., 1989); any special sensitivity of mucociliary transport in asthmatics has not been clearly demonstrated (Spektor et al., 1985; Laube et al., 1993). Furthermore, such exposures result in qualitatively similar effects on mucociliary clearance in humans and various animal species, in that the nature of clearance change, i.e., slowing or speeding, was exposure-concentration dependent, with stimulation of clearance generally occurring at low concentrations and retardation at higher levels (Schlesinger, 1990a), although exceptions to this general scheme of exposure concentration related effects have been noted (Newhouse et al., 1978). Chronic exposure to sulfuric acid also appears to have a similar exposure concentration-response relationship. Rabbits exposed at 125–250 $\mu g/m^3$ (0.3 µm MMAD) for 1–2 hr/d, 5 d/wk showed acceleration or retardation of clearance rate, depending upon the concentration (Schlesinger, 1990a). While the mechanism underlying the effect of acid on mucociliary clearance is unknown, it does not seem to involve changes in the gross structure of airway mucin glycoproteins (Culp et al., 1995).

The clearance of particles from the alveolar region of the lungs is also affected by sulfuric acid. Exposure results in either accelerated or retarded particle clearance, depending upon the exposure regime and animal species examined; effective concentrations range down to 250 $\mu g / m^3$ (Phalen et al., 1980; Schlesinger, 1990b). The mechanistic basis for such effects is not certain, but likely involves acid-induced changes in alveolar macrophage function.

Certain properties of pulmonary macrophages lavaged from various species following single or repeated inhalation exposures have been shown to be altered by sulfuric acid exposures at concentrations < 1,000 $\mu g / m^3$. These include phagocytic activity, surface adherence, random mobility, intracellular pH and the release or production of certain cytokines (e.g., TNFα and IL-1α) and reactive oxygen species (Schlesinger, 1990a; Zelikoff and Schlesinger, 1992; Zelikoff et al., 1994; Chen et al., 1992, 1995). For example, the release and/or activity of TNFα was significantly reduced in alveolar macrophages obtained from rabbits acutely exposed to 75 $\mu g / m^3$ (Zelikoff and Schlesinger, 1992), while the intracellular pH of macrophages was altered by acute exposure of guinea pigs to 341 $\mu g / m^3$. Such effects may ultimately be reflected in alterations in the ability of these cells to adequately perform their role in host defenses, including particle clearance from the alveolar region and resistance to infectious disease. However, the evidence that sulfuric acid alters resistance to bacterial infection is conflicting, and may depend upon the animal model used (U.S. EPA, 1989; Zelikoff et al., 1994).

In a study examining viral infectivity, healthy nonsmoking human adults were exposed for 2 hr to 1,000 $\mu g / m^3$ sulfuric acid (0.9 μm MMAD) with intermittent exercise (Frampton et al., 1992). Bronchopulmonary lavage was performed 18 hr after exposure, and antiviral defenses examined included the release of superoxide during a respiratory burst and the lysis of virus-infected cells in the presence of antibody. There were no acid exposure related changes in neutrophil levels in lavage, in the release of superoxide anion by alveolar macrophages or in the inactivation of influenza virus by these macrophages. Slight, but not statistically significant, changes were observed in the percentage of T lymphocytes (decrease) and antibody-mediated cytotoxicity of macrophages (increase).

The database concerning the effects of sulfuric acid on immunological lung defenses is limited. There is some evidence from animal toxicological studies suggesting that exposure can enhance sensitization to antigens (Osebold et al., 1980) or can modulate the activity of lung mast cells involved in allergic responses (Fujimaki et al., 1992). In this latter study, the effect on stimulated histamine release by lung mast cells isolated from guinea pigs exposed to sulfuric acid aerosols was found to be dependent upon the exposure concentration, with enhanced release at the lowest concentration used (1,000 $\mu g / m^3$) but suppressed release with exposure at the highest concentration (3,200 $\mu g / m^3$); this type of concentration-dependent differential response is similar to that observed with particle clearance.

Pulmonary Morphology and Biochemistry

Acute or chronic exposures to sulfuric acid at high concentrations (> 1,000 $\mu g / m^3$) are associated with a number of characteristic responses in animals, e.g., alveolitis, bronchial and/or bronchiolar epithelial desquamation, and edema (Schwartz et al., 1977; Brownstein, 1980). However, the sensitivity of morphologic endpoints is dependent upon the animal species, with the rat apparently the least sensitive and the guinea pig the most sensitive (Schwartz et al., 1977; Cavender et al., 1977; Wolff et al., 1986). In many cases, the nature of lesions in different species are similar, but differ in location; this is, perhaps, a reflection of interspecies differences in particle deposition patterns.

Acute exposure to sulfuric acid at concentrations 1,000 $\mu g/m^3$ does not produce evidence of inflammatory responses in humans or animals (U.S. EPA, 1989; Frampton et al., 1992). Although one study indicated a change in airway permeability with exposure of guinea pigs for 3 hr/d for 1 or 4 d to 300 $\mu g/m^3$ (Chen et al., 1992), other studies using concentrations 1,000 $\mu g/m^3$ reported no such effects (Wolff et al., 1986; Warren and Last, 1987). Sulfuric acid may also alter the profile of eicosanoids in the respiratory tract (Schlesinger et al., 1990a); reductions in levels of thromboxane-B2 in lavage fluid of rabbits was noted with exposure to 250 $\mu g/m^3$. Chronic exposure to 1,000 $\mu g/m^3$ produced a response characterized by alterations in epithelial secretory cells. For example, rabbits exposed to 125–500 $\mu g/m^3$ (0.3 µm MMAD) for 1–2 hr/d, 5 d/wk showed increases in the relative number density of bronchial secretory cells extending to the bronchiolar level (Gearhart and Schlesinger, 1988; Schlesinger et al., 1992). These changes began within 4 wk of exposure, and persisted for up to 3 mo following the end of all exposures.

An effect of sulfuric acid exposure which may underlie a number of other alterations involves changes in intracellular pH. pH is one of the major determinants of the rate of many cellular functions and has been linked to control of vital cellular processes. Alveolar macrophages obtained from guinea pigs exposed to sulfuric acid showed alterations in internal pH regulation, which was attributable to effects on the sodium ion/hydrogen ion exchanger located within the cell membrane (Qu et al., 1993). Deposited acid may also affect the internal pH of epithelial cells and other functions of these cells, but concentrations used were relatively high compared to ambient levels.

Extrapulmonary Responses

Recent epidemiological studies have noted an association between exposure to ambient particulate matter and cardiovascular disease outcomes or changes in physiological measures of cardiac function (e.g., Schwartz and Morris, 1995; Pope et al., 1999; Dockery et al., 1999; Gold et al., 2000; Peters et al., 2000). Some toxicological studies using ambient particulate matter or various surrogates have examined effects on cardiovascular responses to evaluate the mechanistic basis for such findings (Watkinson et al., 1998; Godleski et al., 2000). However, cardiovascular endpoints have not been extensively evaluated with inorganic sulfates. One study (Sackner et al., 1978) in anesthetized dogs found no changes in various cardiac parameters, including heart rate, pulmonary and carotid arterial pressures, cardiac output and arterial blood gas tensions, in dogs exposed for 4 hr to 4,000 $\mu g/m^3$ submicrometer sized sulfuric acid aerosol.

Carcinogenesis

One of the more controversial biological responses to sulfuric acid is development of cancer, which has been suggested based upon occupational exposures (Beaumont et al., 1987; Soskolne et al., 1989; Siemiatycki, 1991; IARC, 1992); such exposures generally involve particles larger than 1 µm and high concentrations. Various potential mechanisms may underlie any link to carcinogenesis, such as pH modulation of other xenobiotics or low pH-induced changes in cells in mitotic and enzyme regulation (Cookfair et al., 1985). However, the most likely mechanism is simply irritation, the result of acid-induced chronic inflammation, resulting in increased cell proliferation. The ability of sulfuric acid to act as a tumor promoter was suggested by, for example, an assessment of occupational exposure to acid and interaction with tobacco use (Ichinose and Sagai, 1992). The issue of sulfuric acid carcinogenicity is not resolved, especially since there have been no studies of this in animals. Thus, any conclusions for a

potential role of ambient sulfuric acid in respiratory tract cancer cannot as yet be supported by toxicological data.

Coarse Mode Particles (Acid Fogs)

As noted, sulfuric acid can exist within the coarse mode of the ambient aerosol in some regions, as a component of acidic fogs. A limited experimental database exists regarding health effects from exposure to acidic fogs containing sulfuric acid. Differences in biological responses between the submicrometer and the larger fog acid particles may reflect differences in dosimetry and, therefore, ultimate sites impacted by exposure.

The droplets within naturally occurring fogs tend to be hypoosmolar with respect to body fluids (Balmes et al., 1988); most acid fogs have an osmolarity of < 30 mOsm. This property may have some significance in terms of the production of adverse health effects, since the inhalation of hypoosmolar aerosols can be a potent stimulus to bronchoconstriction. Balmes et al. (1988) characterized the relationship between hypoosmolarity and acidity in affecting constriction in asthmatic humans. Using particles consisting of hypoosmolar (30 mOsm) saline at pH 5.5, hypoosmolar sulfuric acid at pH 2, or isoosmolar (300 mOsm) sulfuric acid at pH 2, at sizes in the 5.3–6.1 μm (MMAD) range, specific airway resistance was assessed following a short (3 min) inhalation of each of these aerosols; high gravimetric concentrations were employed, namely > 5 g/m^3. Exposure to the hypoosmolar acid aerosol resulted in bronchoconstriction, while the isoosmolar acid did not. Furthermore, the relative response was greater for both of the hypoosmolar acid aerosols than for the less acidic, hypoosmolar saline aerosol. While these results suggest that, in asthmatics, the acidity of an aerosol can potentiate bronchoconstriction resulting from the inhalation of hypoosmolar particles, high concentrations were used and this may have influenced the results.

Thus, another study performed using acid fogs at much lower concentrations produced different results (Leduc et al. 1995). Asthmatic humans were exposed for 1 hr at rest to fogs having similar physicochemical characteristics as that of naturally occurring fogs collected in Brussels, Belgium. Two fogs were used. One consisted of sulfuric acid suspended in sodium chloride, to produce an isoosmolar fog having an acid concentration of 500 $\mu g/m^3$. The second fog was hypoosmolar and contained sulfate and ammonium ions. The particle sizes of these fogs were 9 μm and 7 μm (MMAD), respectively. No significant changes in pulmonary functional tests or airway responsiveness following bronchoprovocation challenge was noted with exposure to the first atmosphere, indicating no effect due solely to the acidity of natural fogs. Likewise, there was no change in these same endpoints with the second atmosphere, in this case showing no effect of osmolarity. Thus, in this study, a short-term exposure to lower and more realistic concentrations of acid fogs, reproducing the acidity and hypoosmolarity of natural polluted fogs, did not produce bronchoconstriction, nor did it affect airway responsiveness, in adult asthmatics.

Similarly, healthy adult humans exposed for 1 hr, with intermittent exercise, to a hypoosmolar acid fog (10.3 μm MMAD) having a sulfuric acid concentration of approximately 100 $\mu g/m^3$ (Laube et al., 1993) also showed no effect on respiratory function, i.e., forced ventilatory parameters or airway responsiveness. However, mucociliary clearance was found to be accelerated in both large and small ciliated airways by the exposure.

Normal and asthmatic adults exposed for 1 hr to acid fogs (10 μm MMAD) containing 500–2,000 $\mu g/m^3$ sulfuric acid showed some increase in respiratory symptoms, i.e., cough, but no significant change in measures of pulmonary function or in airway responsiveness (Avol

et al., 1988b); this general lack of effect in normals or asthmatics is supported by some other studies (Linn et al., 1989; Aris et al., 1991). Thus, in general, neither healthy nor asthmatic adults show very much change in airway function, including nonspecific responsiveness, following acute exposures to acidic fogs containing sulfuric acid at levels up to about 2,000 $\mu g / m^3$.

SULFATE SALTS

As noted, the bulk of the controlled exposure toxicological database for sulfate aerosols involved sulfuric acid particles. However, the available database on the other particulate sulfates does allow some discussion of the relative potency of these various inorganic salts. In most cases, studies evaluated pulmonary functional parameters following exposure.

Healthy humans exposed for 4 hr, with intermittent exercise, to aerosols (0.55 µm MMAD) of ammonium sulfate (133 $\mu g / m^3$) or ammonium bisulfate (116 $\mu g / m^3$) showed no effect on a pulmonary functional battery, which included FVC, FEV1, and airway resistance (Stacy et al., 1983); it should be noted that there was also no effect from a similar exposure to 100 $\mu g / m^3$ sulfuric acid. Furthermore, healthy adults exposed for 4 hr to 528 $\mu g / m^3$ ammonium sulfate (0.97 µm MMD), with intermittent exercise, showed no significant effect on these parameters, nor on nonspecific responsiveness, either immediately or 24 hr after exposure (Kulle et al., 1984). On the other hand, young adult asthmatics exposed for 16 min showed significant reductions in specific airway conductance (with provocation challenge) and FEV1 with exposure to 1,000 $\mu g / m^3$ of either sulfuric acid or ammonium bisulfate, while exposures at 450 $\mu g / m^3$ produced a change only in conductance and only when sulfuric acid was used; exposure to 100 $\mu g / m^3$ with either aerosol resulted in no effect (Utell et al., 1983a). Neither healthy nor asthmatic adults showed any effect on daily pulmonary function, e.g., FVC, FEV1, nitrogen washout, with exposure for 2 hr / d, with intermittent exercise, for 2–3 consecutive days to either sulfuric acid (101–111 $\mu g / m^3$), ammonium bisulfate (4–47 $\mu g / m^3$) or ammonium sulfate (117–325 $\mu g / m^3$); particle sizes ranged from 0.32–0.55 µm (MMAD) (Avol et al., 1979). Finally, healthy and elderly asthmatic adults exposed for 40 min, with intermittent exercise, to 70 $\mu g / m^3$ ammonium sulfate (0.6 µm MMAD) demonstrated no effect on FVC, FEV1 or total respiratory resistance (Koenig et al., 1993). Thus, if asthmatics are more susceptible to the sulfate salts, this appears to be manifested only at exposure concentrations well above ambient levels.

Some other aspects of respiratory tract function other than pulmonary mechanics have been examined using sulfate salts. No change in the rate of collagen synthesis was found in lung minces obtained from rats exposed continuously for 7 d to 5,000 $\mu g / m^3$ ammonium sulfate (0.8–1 µm MMAD) (Last et al., 1983). In a study designed to examine clearance function (Phalen et al., 1980), rats were exposed for 4 hr, at both high (85%) and low (39%) relative humidity conditions, to 3,600 $\mu g / m^3$ ammonium sulfate or ferric sulfate (0.4 µm MMAD). While ammonium sulfate had no effect on either early (up to 50 hr post-exposure) or late (2–17 d post-exposure) clearance of tracer particles from the lungs, ferric sulfate inhibited the late phase clearance, but only when exposure occurred at low relative humidity. However, the exposure levels used in all of these studies were very much higher than those which would occur in ambient air.

The partially or totally neutralized sulfates are clearly less effective than is sulfuric acid in altering clearance function. When rabbits were exposed for 1 hr to 0.4 µm (MMAD) aerosols of ammonium bisulfate (620–1,150 $\mu g / m^3$), ammonium sulfate (1,800–2,200 $\mu g / m^3$) or sodium sulfate (1,800–1,950 $\mu g / m^3$), only the exposure to ammonium bisulfate produced any change in mucociliary clearance of tracer particles from the bronchial tree and the effect, which was

depression, occurred only at the highest exposure concentration (Schlesinger, 1984). Thus, the response of this endpoint appears to be related to the relative acidity of the aerosol, a relationship which has been noted with other endpoints as well (Schlesinger, 1989). The ranking of irritant potency of the sulfates in terms of alterations in various aspects of lung clearance function appears to be sulfuric acid > ammonium bisulfate > ammonium sulfate, sodium sulfate.

In a study designed to examine the effect of a sulfate salt on pulmonary immune function (Kitabatake et al., 1991), guinea pigs were exposed to ammonium sulfate (0.68–0.85 μm) to determine any effect of exposure upon asthmatic dyspnea subsequently induced by antigen exposure. Animals were exposed at concentrations of 200, 400 or 2,000 $\mu g/m^3$ for 2 hr/d, 5 d/wk for 38 times, following which they were exposed for another 2 hr and then 0.5 hr later were exposed to ovalbumin (antigen) for 0.5 hr. This paired exposure to sulfate and albumin was repeated 7 or 9 times, three times a week. After the above exposures, all animals underwent bronchoprovocation challenge to assess nonspecific airway responsiveness. The sensitized animals exposed to 400 and 2,000 $\mu g/m^3$ showed airway hyperresponsiveness compared to air exposed sensitized animals. There was no effect on responsiveness in the nonsensitized animals exposed to 2,000 $\mu g/m^3$. These results suggested that ammonium sulfate enhanced sensitization by an inhaled antigen, which could have some implications for the immediate asthmatic response in humans. However, exposure concentrations were very high compared to ambient levels.

One study attempted to evaluate the effect of exposure to sulfate aerosols in a compromised animal model. Loscutoff et al. (1985) exposed rats and guinea pigs having elastase-induced pulmonary emphysema, as well as normal animals, to 1,000 $\mu g/m^3$ ammonium sulfate (0.4 μm MMAD) for 6 hr/d for 5 or 20 d (5 d/wk). Pulmonary functional endpoints were examined; these included RV, FRC, VC, TLC, DLco, compliance, and single breath nitrogen washout. There was no effect of exposure in guinea pigs on any of these endpoints, nor was there any interaction between elastase treatment and exposure. In the rats, RV and FRC were significantly greater in sulfate exposed animals compared to air controls, but there was no biologically significant interaction between exposure and elastase treatment. The finding that rats were more sensitive than guinea pigs appears to conflict with other studies at higher concentrations, suggesting that effects between species may differ at different levels of exposure.

Morphologic and morphometric analyses were performed on the animals in the above study that had been exposed for 20 d (Busch et al., 1984). There appeared to be an alteration in secretory activity, characterized by hypertrophy and hyperplasia of nonciliated epithelial cells and an increased number of secretory granules per cell, in the lungs of the guinea pigs, but not the rats, exposed to ammonium sulfate. These changes occurred in small bronchi to terminal bronchioles.

In another study designed to examine the effects of sulfate aerosols on a compromised animal model, Cassee et al. (1998a) exposed mice which had been sensitized to ovalbumin (a model of allergic asthma), as well as normal animals, to ammonium bisulfate for 4 hr/d for 3 consecutive days. The sulfate exposure atmospheres consisted either of fine particles at 78 $\mu g/m^3$ (0.531 μm MMD) or 972 $\mu g/m^3$ (0.453 μm MMD), or ultrafine particles at 235 $\mu g/m^3$ (0.085 μm MMD). Animals were sacrificed at 1 or 4 d after the last exposure. Because of the differences in size and mass concentration, the number concentration of these aerosols also differed, with ultrafine > fine high mass > fine low mass.

A series of endpoints was assessed, which included various cellular, biochemical and immunological parameters using material obtained via bronchopulmonary lavage, as well as measures of serum IgE, nonspecific airway responsiveness (using a tracheal preparation *in vitro*), and histopathology. No exposure related change in levels of protein, lactate dehydrogenase (a marker of cytotoxicity), N-acetyl glucosaminidase (a marker of macrophage activation), IL-4, IL-6 or TNFα in lavage was found in either group of mice with any of the exposure atmospheres. There were no treatment related effects on viability, total counts or differential counts for cells recovered by lavage. There was no significant. difference related to sulfate exposure in the levels of serum IgE in sensitized mice, although IgE levels in these mice exposed to the sulfate particles tended to be lower than those in sensitized air controls, especially at 1 d postexposure. Tracheal responsiveness was also unaffected by ammonium bisulfate, and there were no treatment related effects in either the healthy or compromised mice in lung histopathology. Thus, ammonium bisulfate appeared to exert only marginal responses in some endpoints in this mouse model of allergic asthma, and there seemed to be no indication of any enhanced allergic response following exposure. Furthermore, the study showed there to be no differences in response related to differences in particle size or in particle number concentration of the exposure atmospheres.

Cassee et al. (1999) exposed rats treated with monocrotaline to induce pulmonary hypertension (another type of compromised animal model), and normal animals, to ammonium bisulfate at various concentrations and particle sizes for 4 hr / d for 3 consecutive days. Animals were sacrificed one day after the last exposure. Exposure parameters were as follows: 70 $\mu g / m^3$ (0.070 µm MMD), 275 $\mu g / m^3$ (0.565 µm), 344 $\mu g / m^3$ (0.107 µm), 407 $\mu g / mm^3$ (0.633 µm). The endpoints evaluated were lactate dehydrogenase, N-acetyl glucosaminidase, protein and cell viability, number and types in lavage, histopathology and the phagocytic activity (to *E. coli*) of alveolar macrophages. No significant or consistent exposure-related effects were found on any of these endpoints in either the normal or compromised animals.

Aside from products of the reaction between sulfuric acid and ammonia, the ambient atmosphere may also contain metal sulfates, resulting from the reactions between trace metal oxides derived from fossil fuels with sulfuric acid. Trace metals can be found in the fine aerosol mode in all regions of the world (Harrison and Yin, 2000).

Linn et al (1981) exposed normal young adults and asthmatic humans for 2 hr, with intermittent exercise, to zinc ammonium sulfate at concentrations of 16.2 and 15.6 $\mu g / m^3$ respectively (1.1 µm MMAD). Various measures of pulmonary function, including FVC, FEV1, PEFR, nitrogen washout, and total respiratory resistance, were assessed; sodium chloride particles were added to the exposure atmosphere to model background ambient particulate burdens. Small and inconsistent effects on pulmonary function due to the sulfate aerosol exposures were observed, indicating only minimal to no short-term effects on lung function parameters. Kleinman et al. (1981) exposed normal and asthmatic adults for 2 hr to 81–100 $\mu g / m^3$ ferric sulfate (1.3 µm MMAD) and found no significant exposure related effects on comparable pulmonary functional endpoints.

In a study using the same endpoints as that of Cassee et al. (1998a) discussed above, Cassee et al. (1998b) exposed both normal and ovalbumin sensitized mice for 4 hr / d for 3 d to ammonium ferrosulfate (0.459 µm MMD) at a concentration of 250 $\mu g / m^3$. Animals were sacrificed 1 d after the last exposure. There was found to be no significant exposure related effects on any of the endpoints examined in either group of mice, and there was no evidence for any enhanced allergic response due to the sulfate aerosol.

Finally, Cassee et al. (1999) exposed both normal rats and animals treated with monocrotaline to ammonium ferrosulfate for 4 hr/d for 3 d. Animals were sacrificed one day after the last exposure. Exposure concentrations were 285 $\mu g/m^3$ (0.084 µm MMD) and to 340 $\mu g/m^3$ (0.617 µm). The endpoints evaluated were lactate dehydrogenase, N-acetyl glucosaminidase, protein and cell viability, number and types in lavage and histopathology. No significant or consistent exposure-related effects were found on any of these endpoints in either the normal or compromised animals.

SULFATE-COATED PARTICLES

Ambient air pollution generally consists of complex mixtures rather than only one chemical species. While a general discussion of the toxicology of such mixtures is beyond the scope of this document, one specific mixture involving sulfuric acid bears mentioning. As noted earlier, sulfuric acid may exist as a coating on the surface of other solid particles which may exist in the fine particulate mode derived from combustion processes. Some of these particulate "carriers" may have inherent toxicity themselves, e.g., metals, or some may have low intrinsic toxicity, e.g., carbon. Such acid sulfate-coated particles may be more "realistic" than the aqueous acid particles commonly used in toxicology studies in terms of actual ambient exposures. It is possible that adsorption of acid onto the surface of a solid carrier particle would alter the effective dose of acid delivered to specific sites within the respiratory tract and, thus, affect the ultimate response. Unfortunately, there is a lack of knowledge on the phyicochemical characteristics of these potential carriers in ambient air and of the exact nature of the acidic species which actually occur.

Studies with guinea pigs have suggested that up to an order of magnitude higher exposure levels of pure aqueous sulfuric acid aerosols may be needed to produce comparable biological results than if the exposures involved acid coated on a solid particle core (Amdur and Chen, 1989); exposure to acid present as a coating produced biological effects at concentrations as low as 20 $\mu g/m^3$ (as H_2SO_4). While some confounders in these specific studies included differences in particle sizes between the coated and pure acid droplets and differences in the number concentration of particles in the different exposure atmospheres, it is likely that the physical nature of the inhaled acid sulfate particle is a key factor in determining ultimate response following exposure.

This conclusion is supported by results of studies with acid-coated carbon particles. In one (Jakab et al., 1996), mice were exposed for 4 hr to co-generated carbon black particles (10,000 $\mu g/m^3$, 0.3 µm MMAD) and sulfur dioxide, which resulted, in some cases, in the conversion to acidic sulfates on the carbon. Phagocytosis by lavaged macrophages was significantly suppressed only following those exposures in which significant adsorption of the gas by the carbon particles occurred and there was, in fact, oxidation of sulfur dioxide to sulfate. The investigators suggested that it was possible that the degree of neutralization by endogenous ammonia, which is discussed further below, may differ when acid is present in droplet form than when it is coated on other particles, and that this could ultimately affect the delivery of acid to lung airway surfaces.

The presence of an acid coating may, however, not always be more effective than aqueous droplets. In a controlled clinical study (Anderson et al., 1992), normal and asthmatic adults were exposed, for 1 hr with intermittent exercise, to either 100 $\mu g/m^3$ sulfuric acid aerosol (0.5 µm MMAD) generated from an aqueous solution or to an aerosol consisting of carbon particles (0.5 µm) at 250 $\mu g/m^3$ plus 0.1 mg/m^3 ultrafine sulfuric acid aerosol; the acid became attached to the surface of the carbon. No change in pulmonary functional tests (e.g.,

FEV, specific airway resistance) nor airway responsiveness with provocation challenge was noted with either atmosphere, compared to air control.

An extensive study of the respiratory effects from a model particulate air pollution mixture derived from combustion sources, and which allows some indication of the relative potency of acidic sulfur species, has been reported (Heyder et al., 1999; Kreyling et al., 1999; Takenaka et al., 1999; Schulz et al., 1999). Healthy dogs were exposed over a period of 13 months for 16.5 hr/d to a neutral sulfite atmosphere (used as a surrogate for particle-associated sulfur IV) particles (1.02 μm) at a total mass concentration of 1,530 $\mu g/m^3$ and consisting of particle-associated S-IV (320 $\mu g/m^3$), particle-associated S-VI (70 $\mu g/m^3$) and gaseous S-IV (sulfur dioxide), as well as for 6 hr/d to an atmosphere consisting of similarly sized acidic sulfate particles, namely ammonium bisulfate, at 5,660 $\mu g/m^3$. Controls consisted of animals exposed to clean air. In the exposure atmosphere, the sulfite particles assumed a solid state, while the sulfate particles remained as aqueous droplets. A battery of various endpoints, which included pulmonary function tests, lavaged cell assays, biochemical indicators of injury and particle clearance patterns using tracer particles, were evaluated.

The effects noted were a reduction in macrophage-associated intracellular particle dissolution, an enhanced particle transport across the epithelial membrane and into interstitial tissues and tracheobronchial lymph nodes, and an increased level of alkaline phosphatase in epithelial lining fluid. Morphometric analysis of the conducting airway epithelium and alveolar region showed no differences due to exposure compared to air control, although there was a tendency for an increase in the volume density of bronchial glands and some evidence for type II cell proliferation in the animals exposed to the aerosols. There was no effect on any parameter of respiratory function nor on airway responsiveness to provocation challenge. There was also no change in tracheal mucous velocity or in macrophage phagocytic activity. Thus, the mixture of neutral particle associated S-IV and particle associated acid resulted only in subtle responses and exposure was, therefore, concluded to have not initiated pathological responses in the lungs.

It was also noted that the response to chronic exposure to the mixture of neutral particle-associated S-IV and particle associated hydrogen ions was different from that due to inhalation solely of neutral particle associated S-IV which had been performed earlier (Heyder et al., 1992). For example, exposure to the latter atmosphere produced no change in the proximal alveolar region, so these effects seen with the mixture were likely due either to the total mixture or to the acid components alone. Furthermore, the neutral particle associated S-IV produced inflammatory responses by activation of phospholipid metabolism not seen with the mixture. These results emphasize the complexity of health effects in response to complex mixtures compared to single pollutants, although the former would be the more realistic atmosphere in terms of ambient exposure situations.

TOXICOLOGY OF PARTICULATE NITROGEN OXIDES

Particulate oxides of nitrogen consist primarily of nitrate (NO_3^-) salts. Because of its high saturation vapor pressure, nitric acid generally exists as a vapor under ambient conditions, e.g., within photochemical smog (Ellestad and Knapp, 1988). Within acidic fogs, however, nitric acid may be found within the coarse mode of ambient particulates (Jacob et al., 1985).

Nitrate Salts

The toxicological database concerning health effects from inhaled nitrates is very limited. Anesthetized dogs exposed to sodium nitrate (0.05–0.12 μm, depending upon concentration) at up to 10,000 $\mu g/m^3$ for 7.5 min showed no effect on pulmonary function endpoints, such as respiratory resistance, static lung compliance or functional residual capacity, while exposure for 4 hr to 5,000 $\mu g/m^3$ produced no significant alterations in lung mechanics, pulmonary or systemic arterial blood pressures, cardiac output, heart rate or arterial blood gases (Sackner et al., 1979). Conscious sheep similarly exposed for 4 hr to 5,000 $\mu g/m^3$ demonstrated no alteration in tracheal mucous velocity (Sackner et al., 1979).

In the study of Loscutoff et al. (1985) described above in terms of ammonium sulfate, other groups of normal rats and guinea pigs and those with elastase-induced emphysema were exposed to 1,000 $\mu g/m^3$ ammonium nitrate (0.6 μm MMAD) for 6 hr/d, 5 d/wk for 5 or 20 d. There was no exposure-related effect on lung volumes, compliance or DLco in the guinea pigs, although there was some difference in the slope of the nitrogen washout pattern between nitrate exposed animals and sham controls. However, there was no significant interaction between elastase treatment and particle exposure. In the rats, ammonium nitrate exposure resulted in only minor changes in pulmonary function, and there was no additional effect due to elastase treatment. Furthermore, morphological analysis of animals exposed for 20 d showed no effect of the aerosol (Busch et al., 1984).

In another study using the same endpoints as that in Cassee et al. (1998a), Cassee et al. (1998c) exposed both normal mice and mice sensitized to ovalbumin to two atmospheres of ammonium nitrate aerosol, namely 140 $\mu g/m^3$ (0.584 μm MMD) or 250 $\mu g/m^3$ (0.219 μm). Exposures were for 4 hr/d for 3 d, and animals were sacrificed one day after the last exposure. There were no statistically significant differences in protein or lactate dehydrogenase in lavage. However, the level of N-acetyl glucosaminidase was significantly increased in the sensitized mice exposed to the smaller sized particles, although there was some evidence for an increase in this enzyme in all groups exposed to the ammonium nitrate, suggesting an exposure related activation of macrophages. There were no effects on TNFα, IL-3, IL-4 or IL-6, or on cell viability or total cell number. On the other hand, differential counts showed evidence for increased neutrophil number in the lavage of both the normal and sensitized mice exposed to the larger sized particles. The ammonium nitrate had no effect on levels of serum IgE in the sensitized mice. While there was no exposure related effect on nonspecific airway responsiveness of the trachea in sensitized mice compared to normal mice, it did appear that mice in both groups exposed to the larger, but not the smaller, sized particles showed significantly increased responsiveness; this suggests that ammonium nitrate did not differentially exacerbate airway responsiveness in asthmatic animals. Finally, there were no lung pathologic abnormalities which could be related to ammonium nitrate exposure. It was suggested that since effects from exposure to the larger sized aerosol occurred at lower mass concentrations than did those for the smaller sized aerosol, that particle size, rather than just mass or number concentrations, was an important parameter of response.

Cassee et al. (1999) exposed both normal rats and animals treated with monocrotaline to ammonium nitrate for 4 hr/d for 3 consecutive days. Animals were sacrificed one day after the last exposure. Exposure concentrations were 418 $\mu g/m^3$ (0.087 μm MMD) and 361 $\mu g/m^3$ (0.643 μm). The endpoints evaluated were lactate dehydrogenase, N-acetyl glucosaminidase, protein and cell viability, number and types in lavage and histopathology. There were no consistent, exposure related effects on any of these endpoints. However, exposure to either size of ammonium nitrate aerosol resulted in an increased number and severity of lesions, which

were determined to be due to a background bacterial infection. This suggested some effect of exposure on resistance to infection.

Nitrate salts have been specifically examined in terms of their potential to impact upon susceptibility to respiratory tract infections. Ehrlich (1979) examined the effect of 3 hr exposures to various nitrates (at 1,290–4,500 $\mu g / m^3$) on resistance to respiratory bacterial infection in mice. Only zinc nitrate [$Zn(NO_3)_2$] resulted in any significant mortality increase, the extent of which seemed to be exposure concentration related. However, since the response was similar to that seen with zinc sulfate ($ZnSO_4$), the effect was likely due to the zinc ion rather than to the nitrate ion. In another study, no effect on the expression of surface receptors for the Fc domain of immunoglobulin G (IgG) was noted in pulmonary macrophages obtained from adult rats following inhalation exposure to 90 $\mu g / m^3$ ammonium nitrate for 4 hr/d, 4 d/wk for 8 wk (Ziegler et al., 1994).

In an examination of the effects of nitrate upon the *in vitro* release of histamine by guinea pig lung fragments (Charles and Menzel, 1975), it was found that histamine was released in proportion to the concentration of salt present. However, the response was not totally due to the nitrate ion; ammonium ion was also a possible contributor. The relationship of studies such as this one to actual *in vivo* exposures is, however, not clear. For example, in the study of Ziegler et al. (1994) cited above, acid incubation of macrophages *in vitro* resulted in a decrease in IgG binding for the Fc receptors, an effect not seen with *in vivo* exposure; the difference between the *in vitro* and *in vivo* results may have been due to recruitment of new cells over the exposure period or to differences in pH in the milieu between the two types of studies.

Some controlled clinical studies have been conducted using sodium and ammonium nitrate aerosols, generally in the size range of about 1 µm or below, in both normals and potentially susceptible individuals such as asthmatics (Kleinman et al., 1979, 1980; Sackner et al., 1979; Utell et al., 1979, 1980; Stacy et al., 1983). Exposure concentrations ranged from 80–7,000 $\mu g / m^3$, and pulmonary function (e.g., FVC, FEV1, nitrogen washout, etc) was the usual endpoint. The only effects noted were decreases in airway conductance and peak-expiratory flow-volume curves in subjects with influenza exposed for 16 min to 7,000 $\mu g / m^3$ sodium nitrate ($NaNO_3$) aerosol (Utell et al., 1980). This response was not seen in normals or in asthmatics (Utell et al., 1979), nor did exposure of adult asthmatics to 1,000 $\mu g / m^3$ for 10 min produce any significant effect on lung volumes, distribution of ventilation, dynamic mechanics of breathing or diffusing capacity (Sackner et al., 1979). Thus, there are likely no adverse effects, as far as measured cardiopulmonary function is concerned, from ambient levels of nitrate aerosols, even in presumably more sensitive asthmatic members of the general population. It should, however, be noted that some of the potentially more sensitive cardiopulmonary indices of response, such as heart rate variability, have not been assessed in controlled studies with nitrate particles.

Particulate Nitric Acid

Most studies with nitric acid were designed to examine the histological response to instilled or nebulized acid, a procedure that employed very high levels of acid having no relevance to ambient exposures, to develop models of bronchiolitis obliterans in various laboratory animal species (Totten and Moran, 1961; Greenberg et al., 1971; Mink et al., 1984; Peters and Hyatt, 1986). Those studies which did evaluate toxicological responses generally involved exposure to nitric acid in the vapor state (e.g., Abraham et al., 1982; Koenig et al., 1989; Nadziejko et al., 1992; Aris et al., 1993; Schlesinger et al., 1994; Wong et al., 1996); such

exposures were noted to produce various effects on pulmonary functional and lung defense parameters.

The database concerning potential health effects from exposure to particulate nitric acid is quite sparse. Hypoosmolar particulate nitric acid (5.3 – 6.1 μm MMAD) was found to enhance bronchoconstriction in humans, but the gravimetric concentration of the aerosol was very high, in the g/m^3 range (Balmes et al., 1988); on the other hand, the effect was not seen with an isoosmolar acid exposure. It should be noted that the effect on airway function was equivalent for both nitric and sulfuric acids at the same pH, indicating that the specific anion did not influence the response, which was likely due to deposition of hydrogen ions on airway surfaces.

TOXICOLOGY OF SEA SALT PARTICLES

Sea salt aerosol can contribute to background primary particulate atmospheric levels of anthropogenic-derived pollutants in many regions. It is generated from sea water by spray processes, whereby air bubbles formed by the entrainment of air into water by breaking waves eventually burst, and can exist in both the coarse and fine size modes, this latter accounting for a little under 50% of the total mass (Bloemen et al., 2001). Fine mode sea salt particles can be found at up to hundreds of kilometers from shore.

The chemical composition of sea salt aerosols reflects that of sea water, namely sodium, calcium, magnesium, potassium, chloride and sulfate. Sodium, potassium and chloride are the dominant constituents, accounting for about 87% of the total mass by volume; sulfate accounts for 7.8% by mass (Wilson, 1975; Visser et al., 2001). On an annual basis, the average concentration of sea salt particles in the Netherlands is 4 – 7 $\mu g/m^3$ (Bloemen et al., 2001).

Marine aerosols can exert an influence on the composition of the ambient aerosol in coastal regions, as well as inland, by their involvement in secondary particle production reactions. While there is no known atmospheric sink for sodium, and presumably potassium as well, due to atmospheric chemical reactions, the chloride ion be involved in reactions with ambient sulfuric and/or nitric acids, resulting in the production of particulate sodium sulfate and sodium nitrate (Visser et al., 2001). It should be mentioned that chlorides can also enter atmospheric particles from another source, the result of neutralization of hydrogen chloride vapor by ambient ammonia (Harrison and Yin, 2000).

Sodium chloride aerosols are commonly used in toxicological studies as sham or negative controls for other particulate exposures. Thus, the effects, if any, specifically of these particles have not generally been assessed. However, evaluation of the database can provide some indications of potential health effects from exposure to sodium chloride when responses to these aerosols have been compared to air exposure controls.

Rats exposed for 23.5 hr day for 3 or 7 d to sodium chloride aerosol at 1,000 $\mu g/m^3$ (0.38 μm) showed no change in the rate of collagen synthesis by lung minces after 7 d of exposure, nor in protein content of lavage fluid measured after 3 d of exposure (Last and Warren, 1987). In dogs, exposure for 7.5 min at up to 18,000 $\mu g/m^3$ did not affect respiratory resistance, functional residual capacity, respiratory compliance, or specific respiratory conductance, while exposure for 4 hr at 5,000 $\mu g/m^3$ did not affect any of the above, nor did it alter pulmonary and systemic arterial blood pressure, cardiac output, heart rate, stroke volume or arterial blood gas tensions (Sackner et al., 1978). In sheep, sodium chloride aerosol exposures for 20 min at concentrations up to 22,000 $\mu g/m^3$ produced no change in tracheal mucous velocity, nor was

this parameter affected by a 4 hr exposure at 5,000 $\mu g/m^3$ (Sackner et al., 1978). The particle sizes in the dog and sheep studies ranged from 0.05 -0.3 µm.

A number of human clinical studies used sodium chloride aerosols, with varied results. Adolescent asthmatics exposed for 40 min, with intermittent exercise, to 88 $\mu g/m^3$ (0.6 µm MMAD) showed a small, but significant, drop in maximal expiratory flow (Vmax75) which occurred after the exercise period but not after the first 30 min of exposure at rest (Koenig et al., 1983). On the other hand, exposure of asthmatic adults for 16 min to 100 – 1,000 $\mu g/m^3$ sodium chloride aerosol (0.8 µm MMAD) produced no change in either specific airway conductance or FEV1 with provocation challenge (Utell et al., 1983a). Normals and asthmatics exposed to 278 – 314 $\mu g/m^3$ sodium chloride (1.1 µm MMAD) for 2 hr with intermittent exercise showed no effect on various pulmonary functional parameters, e.g., FVC, FEV1, PEFR, TLC, nitrogen washout, or total respiratory resistance (Linn et al., 1981). No significant effects were noted in normals or asthmatics on total respiratory resistance, FEV1 or VC due to a 10 min exposure to sodium chloride at 10, 100 or 1,000 $\mu g/m^3$ (0.05 – 0.3 µm), while exposure for 10 min to 1,000 $\mu g/m^3$ resulted in no change in these individuals in pulmonary capillary blood flow or diffusing capacity (Sackner et al., 1978).

Adolescents (normal) were exposed to NaCl at 1,000 $\mu g/m^3$ (0.9 µm MMD) (Frampton et al., 1992). Measurements were total respiratory resistance, FRC, maximum flow (Vmax50) and FEV. These parameters were measured after exposure for 30 min at rest and then again after an additional ten minute exposure with exercise. There were no statistically significant changes in pulmonary function following the initial exposure at rest, but aerosol exposure resulted in a small, but not statistically significant, decrease in FEV after the 10 min of exercise; the were no changes in the other endpoints. This suggests that exercise could exacerbate the effects of exposure. Finally, nonsmoking normal humans exposed to 1,000 $\mu g/m^3$ sodium chloride aerosol for 2 hr and undergoing lavage 18 hr later showed no change in neutrophil counts nor in superoxide anion release by macrophages or in the inactivation of influenza virus.

There has been some limited evaluation of larger sized saline particles which may exist in atmospheric fogs. Healthy adult males were exposed twice to 10.9 µm (MMAD) equiosmolar saline droplets for 1 hr with intermittent exercise. Neither respiratory symptoms, forced ventilatory function nor airway responsiveness with provocation challenge was affected by exposure; the liquid water content of the fog was 492,000 $\mu g/m^3$ (Laube et al., 1993). Finally, the bronchoconstrictive response of asthmatics was found to be greater following exposure to a hyposmolar sulfuric acid aerosol than following a hypoosmolar saline aerosol having less acidity (Balmes et al., 1988). As noted earlier, this may indicate that, in asthmatic, the acidity of an aerosol can potentiate the bronchoconstriction caused by inhalation of hypoosmolar aerosols.

CONSIDERATIONS IN THE USE OF TOXICOLOGICAL DATA FOR RISK ASSESSMENT

The development of risk assessments for particulate air pollution should ideally be based upon measured responses in humans exposed under realistic conditions. They should involve the integration of data from epidemiology as well as controlled exposure studies in humans and animals. Toxicological data are also needed to study underlying biological mechanisms which may help to understand the epidemiological findings, as well as to develop detailed exposure-dose-response relationships. Thus, the utility of toxicological studies in the risk assessment process is to establish definitive cause-effect relationships between exposure to a specific material or materials and the development of exposure-dose-response profiles under both acute and chronic temporal conditions. These are clearly essential steps in the process of determining human safety standards (NRC, 1983).

The usefulness of toxicological data in developing any risk assessment is significantly improved when there is information concerning some important issues. These are the chemical basis underlying biological effects, which impacts upon the nature of responses in different species; the relationship between qualitative and quantitative effects noted with controlled exposure studies in animals compared to those found in humans; and the roles of exposure parameters, namely concentration and duration, on response. However, the only database relevant to this document for which such information is available involves acidic sulfate particles, most notably sulfuric acid. Thus, it is this database that is largely discussed in the following sections.

CHEMICAL BASIS OF TOXICITY

Knowledge of the basis of toxicity is important when attempting to determine whether the nature of responses in animal studies could be relevant to those which may occur in humans similarly exposed. There can be significant differences in the manner by which xenobiotics are handled in different species, and this has the potential to impact the relevance to humans of effects observed in other species following exposure to some chemical agents.

Responses to acidic sulfate particles likely result from the deposition of hydrogen ions (H^+) on airway surfaces (U.S. EPA, 1989; Schlesinger, 1989; Schlesinger et al., 1990b; Schlesinger and Chen, 1994), and this is likely also the basis for the toxicity of nitric acid as well (Balmes et al., 1988). Examination of diverse biological endpoints, including mucociliary transport, alveolar clearance, pulmonary function and production of biological mediators, has shown that the relative potency of acidic sulfate aerosols is directly related to their degree of acidity, i.e., the hydrogen ion content within the exposure environment (e.g., Schlesinger, 1984; Schlesinger, 1989; Koenig et al., 1993). What is not clear, however, is which metric for hydrogen ion concentration better relates to these responses, i.e., whether it is total available hydrogen ion concentration, as measured by titratable acidity in lung fluids following deposition, or free hydrogen ion concentration, as measured by pH of the exposure solution (Fine et al., 1987).

Once inhaled, airborne acidic particles may undergo neutralization, affecting their efficacy in producing biological responses. Ammonia exists at fairly high concentrations in the upper respiratory tract, primarily due to bacterial activity within the oral cavity; and its presence can result in partial or complete neutralization of inhaled acidic particles. The extent of neutralization is dependent upon size of the inhaled particles. There would likely be substantial neutralization for smaller sized particles (e.g., those < 0.1 μm), whereas larger ones (e.g., those > 1.0 μm) would show negligible neutralization (Sarangapani and Wexler, 1996). The extent of neutralization of inhaled acids in the respiratory tract has, in fact, been shown to modulate response. Asthmatic subjects inhaling sulfuric acid aerosol demonstrated greater effects when exposure was conducted under conditions in which oral ammonia levels were low compared to when they were high (Utell et al., 1989).

Because of the potential for neutralization in the upper respiratory tract, the mode of inhalation can affect the extent of any biological response. For example, for the same mass (ionic) concentration of acidic sulfates in an exposure atmosphere, oral inhalation will result in more neutralization compared to nasal inhalation and, therefore, less hydrogen ions available for deposition within the lower respiratory tract (Larson et al., 1980, 1982).

Another factor affecting neutralization is the residence time of inhaled acidic particles within the airways, which is a function of ventilation rate. (Larson et al., 1993). Combinations of high ammonia and low ventilation rate, or low ammonia and high ventilation rate, result in

lesser or greater amounts, respectively, of acidic sulfates available for deposition, even if the acid concentration at the point of inhalation remains constant. This is because the former condition results in greater neutralization than does the latter.

Since the biological responses to inhaled acidic particles are likely due to the deposition of hydrogen ions on airway surfaces, then the basis of these responses likely involve local changes in pH at sites at which these particles deposit (Last et al., 1984; Hattis et al., 1987). It is, therefore, likely that a critical number of such particles must be deposited at target sites in order to deliver sufficient hydrogen ions so as to alter local pH. It has been demonstrated that the number of hydrogen ion-containing particles within an exposure atmosphere, at least for sulfuric acid, rather than just the total mass concentration of these ions, is an important factor in determining whether any response follows inhalation (Chen et al., 1995). Furthermore, there appears to be a threshold for both number concentration and mass concentration in this regard. There is some suggestion from population based studies that in some cases at least, the number concentration as well as the mass concentration may be an important parameter in the relationship between dose metric and health outcome (Penttinen et al., 2001).

Once acidic particles deposit upon airway surfaces, they can be subjected to buffering action within airway fluids. However, the buffering capacity of these fluids may be altered in individuals with compromised lungs. For example, sputum (which contains mucus and other fluids) from asthmatics was found to have a reduced buffering capacity compared to that from normals (Holma, 1985, 1989) and this may, at least in part, underlie the apparent increased sensitivity of some asthmatics to acidic sulfates.

INTERSPECIES EXTRAPOLATION

Biological responses derived from exposures to animals need to be evaluated in terms of the potential for similar exposures to produce effects in humans. One consideration in this regard is the equivalency of biological response, i.e., would the same dose delivered to the target site in different species produce the same quantitative and qualitative response. Another consideration involves the relative equivalency of dose, again both qualitatively and quantitatively, when humans and animals are exposed to aerosols of specific size distributions.

In most animal toxicological studies, exposures involved concentrations of particulate matter which are very much above ambient concentrations encountered by humans. When concentration relevance is sometimes considered, the levels to which the animals are exposed are those which humans may actually encounter, but there is usually little consideration of any interspecies dosimetry differences in these studies. There can be significant interspecies differences in dose (Schlesinger, 1995a), and this may be reflected, to some extent, in observed differences in response between animals and humans following exposure to particulate matter when the results from various controlled exposure studies are compared.

There have been some attempts at direct comparisons of exposure-response patterns between humans and animals. These involved sulfuric acid aerosols, and employed involved identical exposure concentrations across species. In one study (Zelikoff et al., 1997), rabbits and normal human adults were exposed to sulfuric acid at $1{,}000\ \mu g/m^3$ (0.8 μm) for 3 hr. The biological endpoints examined were biochemical mediators in lavage and various aspects of pulmonary macrophage activity, such as reactive oxygen species production and phagocytosis. Exposed rabbits showed significant effects on more of these endpoints than did humans. The only endpoint in which both species responded was stimulated production of superoxide by macrophages and, in this case, the change from air control levels induced by acid exposure in

the rabbit was greater than the change found in the human subjects.

Schlesinger (1985) compared changes in tracheobronchial mucociliary clearance which had been reported in previous studies. Clearance had been evaluated following single, 1 hr exposures to sulfuric acid at 100 – 1,000 $\mu g/m^3$. In humans, upper tracheobronchial tree clearance was accelerated, and lower tracheobronchial tree clearance was retarded, following exposure at 100 $\mu g/m^3$. In the rabbit, regional clearance was not assessed, but overall tracheobronchial clearance began to be accelerated following exposures between 100 and 2,000 $\mu g/m^3$, while clearance was retarded following exposures at levels > 1,000 $\mu g/m^3$. In the rabbit, clearance was not altered at 100 $\mu g/m^3$. These results suggest that, at least for this endpoint, the human was more sensitive than the rabbit.

These cited studies appear to indicate that the relative sensitivity of the animal species likely depends upon the biological endpoint being examined, While this may not necessarily be surprising, there was no attempt at dose equivalency in these studies. However, in spite of this, it does seem that the qualitative exposure concentration-response relationship for the human and rabbit are similar, and clearly do not differ by orders of magnitude. Based upon this, it can be concluded that observed responses obtained in studies with rabbits, which were often performed using low exposure concentrations of sulfuric acid, do not need uncertainty factors to apply to the human response. Since the toxicity of the inorganic secondary acidic sulfates is due to nonspecific responses to hydrogen ion deposition, this conclusion could also be extended to other animal species in which the deposition of acids was similar to that in the rabbit. Furthermore, if the effects of other inorganic secondary aerosols also reflect their acidity, these conclusions may apply to other types of aerosols as well.

It is clear that in risk assessment using animal toxicological data that the relationship between animal studies to actual human exposures in regards to inhaled doses and ultimate must be considered. For example, while the deposition of particles on a mass per unit alveolar surface area may be similar in different species, dose metrics based upon particle number per various other anatomical parameters, such as alveolus or alveolar macrophage, do exhibit interspecies differences, especially for particles in the accumulation mode range which includes the bulk of the mass of secondary inorganic aerosols (Miller et al., 1995). As another example of interspecies differences, the mass deposition of 0.4 μm particles per unit alveolar surface of humans has been suggested to be about 17% of the deposition per unit surface area in the rat (Miller, 2000). This difference was estimated to become less at larger particle sizes.

In addition to qualitative and quantitative considerations of initial dose distribution following particle inhalation, evaluation of dosimetry also must take into account differences in clearance rates and patterns between species, since these processes affect retention in the respiratory tract and ultimate dose to target tissues (Schlesinger, 1995a). There can be significant interspecies differences in the anatomic sites of particle retention within the respiratory tract (Nikula et al., 2001), which will likely influence the responses to inhaled particulate matter.

EXPOSURE CONCENTRATION – EXPOSURE DURATION – RESPONSE RELATIONSHIP

The nature and severity of the response to an inhaled toxicant is a function of both the exposure concentration © and the exposure duration (T). The first attempt at a quantitative generalization to relate response to exposure scenario was proposed by Haber (1924); this stated that the magnitude of the toxic response, W (for Wirkung or effect), was directly equal to the product of exposure concentration © and exposure duration (T), i.e., W = C x T. This suggested

that the toxicological response from different exposures to a chemical should be the same as long as the product of C x T was held constant. An understanding of the roles that C and T play in the responses to inorganic secondary aerosols is needed to help predict the severity of effects under different exposure conditions. This is especially important when attempting to determine the significance of acute exposures to the repeated or chronic scenarios likely encountered in ambient exposure situations.

The earliest reported examination of the roles of C and T in response to acute exposure to sulfuric acid in animals was that of Amdur et al. (1952). Exposure of guinea pigs to 8,000 $\mu g/m^3$ sulfuric acid aerosol for either 8 or 72 hr did not result in mortality, leading to the conclusion that lethality depended upon C rather than upon T. However, the cause of death was laryngeal or bronchospasm, and the concentration used in this study was below the threshold for such an effect. Once this threshold concentration was exceeded, however, longer duration exposures did affect percentage mortality, although the relative contribution of exposure duration to mortality varied depending upon the concentration, rising with increasing concentration. On the other hand, this same study indicated that the extent of pulmonary histological damage was highly dependent upon both C and T, i.e., cumulative exposure or dose, since exposure to 8,000 $\mu g/m^3$ sulfuric acid for 72 hr (dose rate = 576,000 $\mu g/m^3/hr$) produced more extensive tissue damage than did exposure to 20,000 $\mu g/m^3$ for 8 hr (dose rate = 160,000 $\mu g/m^3/hr$). These results suggested that while the toxicological effects may be dependent upon both exposure concentration and duration, the relationship between these factors is often complex. It is likely that the biological endpoint being assessed in relation to the exposure scenario will affect any decision as to how response to a specific chemical depends upon concentration and duration.

By far, the largest database concerning the CxT relationship for acute sulfuric acid exposures involved tracheobronchial mucociliary clearance in the rabbit as the endpoint. With a series of single 1, 2, and 4 hr exposures (Chen and Schlesinger, 1983; Schlesinger et al., 1984; Schlesinger, 1989) to concentrations ranging from 100 to 2,200 $\mu g/m^3$, it was noted that exposures at the same concentration, but for different durations, did not necessarily result in responses of the same magnitude. Multiple regression analyses to assess the relative contribution of C and T to the observed responses indicated both to be highly significant contributors to sulfuric acid-induced changes in mucociliary clearance (Schlesinger, 1989). These analyses indicated that C had a greater influence on the ultimate response than did T, and that C was more important for a fixed level of C x T.

Other acute studies involved exposure of rabbits to sulfuric acid at concentrations of 75 - 200 $\mu g/m^3$ for various durations. Significant changes in clearance were found following exposures at 100 $\mu g/m^3$ for 2 or 3 hr , and at 200 $\mu g/m^3$ for 1 or 1.5 hr. These exposures resulted in C x T values of 200 and 300 $\mu g/m^3/hr$, respectively. On the other hand, no significant changes in clearance were obtained with exposures at 75 $\mu g/m^3$ for any of the exposure times, which ranged from 1 to 4 hr. This was in spite of the fact that the longest duration at this lowest concentration also resulted in a C x T value of 300 $\mu g/m^3/hr$. Thus, these data suggested that a level of 75 $\mu g/m^3$ was below (or at) the concentration threshold needed to produce any effect, at least with an exposure duration of up to 4 hr. It should also be noted that exposure to 200 $\mu g/m^3$ for 0.5 hr did not alter clearance, while the same concentration for 1 hr resulted in clearance acceleration. This suggests that there is also an exposure duration threshold for response, with exposures for shorter than a certain time (which may depend upon concentration) having no observable effect.

Aside from mucociliary clearance, the relative roles of C x T in eliciting changes in clearance from the alveolar region of the lungs of rabbits have also been fairly extensively examined (Schlesinger and Gearhart, 1986; Schlesinger, 1989; Schlesinger, 1990b). These studies involved acute sulfuric acid exposures at 250 –1,000 $\mu g/m^3$ for 1 to 4 hr daily for 14 d. As with mucociliary clearance, multiple regression analyses indicated that both C and T were highly significant contributors to the observed responses, suggesting that the product of concentration and duration was a better predictor of response than was merely C alone.

There have been no studies which systematically examined the relative contributions of C and T to sulfuric acid-induced responses in humans. Spektor et al. (1989) examined the effect of exposure duration on mucociliary clearance by exposing normal subjects to 100 $\mu g/m^3$ sulfuric acid for either 1 or 2 hr. The acid-induced change in clearance following the 1 hr exposure was less than that following the 2 hr exposure. In fact, the change following the longer duration exposure was similar to that observed with a 1 hr exposure to 1,000 $\mu g/m^3$ sulfuric acid (Leikauf et al., 1981, 1984). The investigators concluded that doubling the duration of exposure had an effect similar to that of a ten-fold increase in the exposure concentration.

One of the major difficulties in risk assessment is bridging the gap between effects with acute exposures and those that may occur with chronic exposures. The relationship between any cumulative toxicity with repetitive exposures compared to effects noted with acute exposures is often complex, and it may not always be possible to predict from the later scenario all responses from the former. The database that allows assessment of the roles of C and T with chronic exposure to sulfuric acid is quite limited. There are only two studies that can be used for such an evaluation. One (Gearhart and Schlesinger, 1988) involved exposure of rabbits to 250 $\mu g/m^3$ sulfuric acid for 1 hr/d, 5 d/wk for 12 mo, while the other (Schlesinger et al., 1992) involved exposures to 125 $\mu g/m^3$ for 2 hr/d, 5 d/wk also for 12 mo. Cumulative daily acid exposures in these two studies were comparable, i.e., 250 $\mu g/m^3$/ hr. The common endpoints examined were mucociliary clearance from the tracheobronchial tree and the number density of epithelial cells in conducting airways.

At the higher exposure concentration, a slowing of clearance was observed during the first month of exposure, and this became progressive with time through the end of the exposure period. At the lower concentration, exposures resulted in acceleration of clearance during the first months of exposure, with a slowing trend seen only towards the end of the 12 mo exposure period. An acceleration of clearance is generally associated with lower concentration exposures to an irritant, while retardation occurs as exposure levels increase. Thus, the results of these two chronic studies suggest that, for long-term exposures, clearance response was more dependent upon daily inhaled concentration than upon daily exposure duration.

In terms of epithelial secretory cell hyperplasia, exposures at 250 $\mu g/m^3$ for 1 hr/d resulted in increased cell number density after 4 mo, while exposures to 125 $\mu g/m^3$ for 2 hr day resulted in similar changes only after 12 mo. Results with this endpoint also suggest a more pronounced effect of acid concentration than of daily exposure duration on the ultimate response.

CONCLUSIONS

This document evaluated the health effects from exposure to secondary inorganic aerosols based upon results of controlled exposure studies in animals and humans. It is clear that, by far, the largest database for these aerosols involves sulfates, with much less information available for the particulate nitrates. Furthermore, for sulfates, the greatest database is for

aqueous sulfuric acid droplets. It does appear that the effects noted with the secondary aerosols discussed are likely due to the acidity of the inhaled chemical species, rather than to the specific anion. Thus, based upon the available information, the toxicologically significant secondary aerosols are likely those having strong acidity, namely sulfuric acid, ammonium bisulfate and nitric acid.

In the evaluation of effects from ambient pollution, an important consideration is the potential for special susceptibility of specific subgroups within the general population. Adolescent asthmatics may represent a sensitive segment of the population with respect to bronchoconstrictive effects of fine mode acidic aerosols. Controlled exposures have produced transient changes in pulmonary function in asthmatics, including enhanced nonspecific airway hyperresponsiveness in some cases. Although epidemiological evidence indicates that exacerbation of symptoms in asthmatics may be related to atmospheric particulates, the contribution of chronic ambient particulate exposure to the development of airway hyperresponsiveness in normal individuals remains unclear.

Many of the controlled animal studies and all of the human clinical studies involved acute exposures. The available evidence suggests that the minimally effective concentration of sulfuric acid to alter pulmonary mechanical function, including nonspecific airway responsiveness, in normal humans following acute exposure is > 1,000 $\mu g / m^3$, but in asthmatics it may be around 68 - 100 $\mu g / m^3$. However, effects on asthmatics, especially at these low concentrations, are quite inconsistent. This may be due to the normally large variability in asthmatic responses to low level air contaminant exposure and also to susceptibility differences within segments of the asthmatic population. For example, elderly asthmatics do not seem to be especially susceptible, but adolescent asthmatics may be more susceptible. In any case, the extent of effect on pulmonary function is small at low acid exposure concentrations, with changes of < 10% in one commonly measured parameter, namely FEV1, following exposures of asthmatic subjects to sulfuric acid aerosols at concentrations $500 \mu g / m^3$. There are likely no adverse or irrreversible effects, as far as cardiopulmonary function is concerned, from ambient levels of sulfate or nitrate aerosols, even in presumably more sensitive asthmatics.

Another biological endpoint which has been extensively examined with acute exposure of humans is mucociliary clearance from the tracheobronchial tree. This has been shown to be transiently altered in normal individuals by sulfuric acid aerosol at a concentration of 100 $\mu g / m^3$, with no evidence for any susceptibility difference for asthmatics. The nature of the effect at this concentration can be acceleration or slowing of clearance, depending upon the region within the tracheobronchial tree that is being examined. However, the pathological significance, if any, of such transient effects is not certain.

Perhaps more relevant to repeated ambient exposures is the results of longer term controlled exposure studies. These have all involved animals, and indicate the potential for production of nonspecific airway hyperresponsiveness, persistently retarded mucociliary clearance and changes in airway secretory cell function with repeated exposures to sulfuric acid at concentrations ranging from 100 – 250 $\mu g / m^3$. The development of hyperresponsive airways in healthy animals at exposure levels below that producing any change in standard lung function indices may have implications for the pathogenesis of airway disease, and alterations in mucociliary function could have implications in terms of the development of chronic obstructive pulmonary disease. However, as noted, considerations of dose equivalency must be included in any evaluation in this regard.

The issue of exposure concentration (C) and duration (T) comes into play when evaluating acute exposure responses in terms of circumstances where repeated exposures are the pattern of concern. Evaluation of the role of C and T must be performed using the acute exposure database, since the database for chronic exposures is much too sparse in this regard. The response to acute exposures, at least in terms of their effect on two of the most commonly evaluated endpoints, namely tracheobronchial mucociliary clearance and respiratory region clearance, appears to be a function of both C and T. In the rabbit, for example, a concentration of 75 $\mu g/m^3$ sulfuric acid was at or below the no-observed-effect level for altering mucociliary clearance, even when the C x T was equivalent to that obtained using concentrations at 100 or 200 $\mu g/m^3$. Exposure to 50 $\mu g/m^3$ for 2 hr/d for 14 d was found to be ineffective in altering alveolar region clearance. It appears that a threshold exists for both the number of deposited acid particles as well as the mass concentration needed to produce any biological response, at least for some endpoints.

One important consideration in evaluating the health effects from secondary inorganic aerosols is this issue of threshold. The current epidemiological database for ambient particulate matter suggests that the concentration-response relationship presents no indication of a clear threshold. However, for some specific chemical constituents of ambient particulate matter, a threshold may exist. For example, the occurrence of a threshold concentration, exposure to which would not result in any effect regardless of the exposure duration, is not unexpected for acidic particles. The response to acid sulfates and other inorganic acidic chemical species is likely due to the deposition of hydrogen ions on airway surfaces. The extent of available hydrogen ions may be altered in the inhaled air, or once the aerosol deposits on airway surfaces, due to the presence of endogenous ammonia in the respiratory tract and buffers in airway surface fluids, respectively. Thus, it is likely that a certain threshold concentration is needed to overcome these processes, and result in deposition of sufficient hydrogen ions so as to alter localized airway surface or cellular pH. The occurrence of such a threshold may also contribute to the inconsistencies which are often noted in human exposure studies involving low acid concentrations.

In addition to a threshold for exposure concentration, a threshold for exposure duration also seems to occur, such that acute exposure for longer than this critical time is needed at some effective exposure concentration in order to overwhelm the buffering capacity of the airway surface fluids and to produce an observable response. For example, exposures to sulfuric acid aerosol for 30-40 min at 1,000 $\mu g/m^3$ were required before any effect on mucociliary clearance could be observed in animal studies (Schlesinger et al., 1978). Thus, a normal individual should be able to tolerate certain exposure regimes with no observable effect. However, the particular C and T values which do not result in observed responses could differ for people with airway disease, and the product is likely not a constant value. Furthermore, the specific threshold for C or T likely depends upon both the sensitivity of the endpoint being examined and the exact exposure protocol.

The available database for acidic sulfates strongly suggests that long duration exposures to low concentrations and short duration exposures to high concentrations may not necessarily be toxicologically equivalent. This would likely apply to other inhaled acidic particulates as well. Given this database, it can be concluded acute effects at 100 $\mu g/m^3$ on potentially sensitive human asthmatics are small and very inconsistent, and that the observed change in airway epithelial secretory cells in rabbits chronically exposed at 125 $\mu g/m^3$ was not associated with a persistent or permanent alteration in mucociliary transport function. Taking into consideration similarities in the relationship between animal and human effects when responses have been directly compared, a threshold NOAEL level of 50 $\mu g/m^3$ may be appropriate for the irritant effects of pure sulfuric aerosol droplets in humans. However, there is the caveat that effects may

occur at lower levels if acid is coated on a solid “carrier” particle.

The toxicological database does not as yet support a role of ambient acidic sulfates in adverse health outcomes noted in epidemiological studies, at least from the viewpoint of effective exposure concentrations. Levels of acidic sulfates needed to produce any effect in controlled studies are well about those found in ambient air. However, the exact physicochemical characteristics of ambient acid particulate matter in the epidemiological studies, and to which human populations are normally exposed, may differ from those used in the controlled toxicological studies, and such characteristics will likely affect the exposure concentration- response paradigm. For example, as noted, most controlled studies used pure sulfuric acid, droplets, but in ambient air these may be neutralized or occur as a pure acid surface coating on other particles. Furthermore, the most direct measure of acid aerosol is the strong hydrogen ion concentration of ambient particulate matter, and this parameter is difficult to accurately assess at the low ambient levels at which it occurs; thus, it has not been widely used and this limits those analyses that can directly assess the potential health effects from strongly acid aerosols (Gwynn et al., 2000). However, in any case, ambient particulate matter in the Netherlands has relatively low acidity, and the major components of secondary ambient particulate matter are ammonium sulfate and ammonium nitrate.

To the extent that they have been evaluated, toxicological studies of these constituents of the secondary fraction, as well as metal sulfates and nitrates and ammonium bisulfate, suggest there to be little toxic potency at concentrations which are environmentally relevant for the Netherlands. The available database presents little evidence for any marked effects on normal humans or animals, or in the limited compromised animal models used with relatively realistic (for humans) ambient exposure concentrations. Finally, ambient levels of sodium chloride are well below those shown to produce any effects in toxicological studies.

It should be noted again that many controlled exposure studies used concentrations of particulate matter that were much higher than those occurring in ambient air. Thus, some of the mechanisms elicited may not occur with exposure to lower levels. Clearly, controlled exposure studies have not as yet been able to unequivocally determine the particle characteristics and the toxicological mechanisms by which ambient particulate matter may affect biological systems. Furthermore, one must always consider that with ambient exposures, health outcomes may be the result of interactions between secondary particulate matter with other co-existing ambient gaseous pollutants, such as ozone (Thurston et al., 1992, 1994; Bates and Sizto, 1995; Schlesinger, 1995b; Frampton et al., 1995). Unfortunately, the role of gaseous pollutants in association with particulate matter in eliciting health effects is still uncertain (Sarnat et al., 2001).

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Annex C

NF-κB

Exposure of various cell types to many inflammogenic stimuli such as lipopolysaccharide, asbestos, silica and oxidants has been shown to enhance the DNA binding of NF-κB, a redox-sensitive transcription factor (Kopp and Ghosh, 1995). NF-κB is a protein transcription factor that is required for maximal transcription of many proinflammatory molecules thought to be important in the generation of inflammation (Christman *et al.*, 2000). These include:

- Adhesion molecules (intercellular adhesion molecule 1 [ICAM-1])
- Critical enzymes (inducible nitric oxide synthase, cycloxygenase-2)
- Cytokines (interleukin IL-1, tumor necrosis factor TNF-α, IL-6
- Chemokines (IL-8)

How is NF-κB involved in the pathogenesis of lung disease?

NF-κB binds to DNA in the promoter regions of target genes as a dimer usually composed of two Rel family proteins, p50 (also called NF-κB1) and RelA (also called p65). The most potent inducers of the NF-κB activation pathway are:

- Gram-negative endotoxin or lipopolysaccharide (LPS)
- TNF-α
- IL-1β

But T cell activators, UV irradiation, growth factors and viral infection are known to induce NF-κB activation (Baeuerle and Henkel, 1994; Baldwin, Jr., 1996; Kopp and Ghosh, 1995). These all result in activation of a specific mitogen-activated protein kinase kinase kinase, NIK (NF-κB-inducing kinase) which in turn activates the IKK signalsome by phosphorylation. The IKK signalsome is an IκB kinase and consists of IKKα and IKKβ, which catalyses phosphorylation of serine residues on both IκBα and IκBβ. Regulation of NF-κB, and therefore activation and the generation of acute inflammation, is then finally mediated by this inhibitory kappa B (IκB) family. IκB-α has a prominent role in down-regulation of NF-κB, and IκB-β is capable of resulting in sustained activation of NF-κB. In the latter case, NF-κB activation requires cytosol dissociation of the inhibitory subunit IκB from the NF-κB protein complex, allowing its nuclear translocation and binding to DNA consensus sites. A specific IκB-α kinase complex phosphorylates IκB, leading to its rapid ubiquitination and subsequent degradation by the proteasome (Zandi *et al.*, 1997). In several cases, NF-κB activation has been shown to be dependent on reactive oxygen species.

It is hypothesised that in ALI (acute lung injury) and SIRS (systemic inflammatory response syndrome), the triad endotoxin (LPS), TNFα, and IL-1β results in activation of NF-kB in the lung and other organs, which leads to cytokine and chemokine gene expression and neutrophil-associated organ disfunction, clinically recognised as multiple organ dysfunction syndrome. It has been shown that activation of NF-κB occurs in alveolar macrophages obtained by BAL from patients with ARDS (Schwartz *et al.*, 1996). The alveolar macrophage seems to have a critical sentinel role in mediation NF-κB activation in the lung and in generating neutrophilic inflammation. However, the NF-κB is not always a predictive marker and certainly not the sole determinant of pathophysiology or outcome.

Many *in vitro* studies have shown that a wide array of particulate air pollutants result in activation of NF-κB, including residual oil fly ash (Quay *et al.*, 1998), copper ion-

containing particulate from the atmosphere of Provo, Utah (Kennedy *et al.*, 1998), and diesel exhaust particles (Takizawa *et al.*, 1999).

AP-1

AP-1 is composed mainly of the Jun and Fos gene products, which form homodimeric (Jun/Jun) or heterodimeric (Jun/Fos) complexes. Perturbation of the cellular thiol redox status has shown to provide a signal for AP-1 activation by the induction of stress-activated signal transduction pathways by c-Jun N-terminal protein kinase (JNK) and p38 kinase.

Annex D.

Further PM emission reductions in transport

D.1. Introduction

As shown previously in subsection 2.5.1.1., total PM_{10} emissions from transport in 2010 lie between 11 and 12 ktonnes per year. The major contributors are inland shipping (~15%), Light Duty Vehicles (vans) and diesel-engined passenger cars (~10%), mobile machines (~15%) and tyre, brake lining and road surface wear (~35%). The following two sections deal with the possibilities for reducing PM_{10} emissions from these successive major sources. Section D.6 addresses PM_{10} reduction measures for other source categories like Heavy Duty diesel Vehicles. For all measures, the effects are assessed in 2010, 2020 and 2030 (only for the EC scenario), assuming a realistic penetration of new technologies (see subsection D.7.1). In addition, the effects in 2010 are estimated based on the assumption of full penetration of new technologies in 2010 (subsection D.7.2). Both estimates are indicative.

D.2. Inland shipping

The EC current legislation scenarios assume the entry into force from 2001 of only the first stage of the CCR[1] emission legislation for inland ships. However, the maximum level of PM_{10} emissions (between 0.54 and 0.85 g/kWh) in this first stage is not stringent enough to result in a significant reduction of the energy-specific PM_{10} emissions from inland ships in 2010 compared with 1995. If it is possible to harmonise emission legislation for inland shipping and heavy duty road vehicles (HDV) from 2005 (when Euro4 comes into force for HDV), PM_{10} emissions from inland shipping in 2010 could be reduced by almost 25% (0.4 ktonnes). In 2020 PM_{10} emissions from inland shipping would be reduced by approximately 70% (1.4 ktonnes). These estimates are optimistic in view of the current problems in the CCR involved in achieving an agreement on the second stage and are still far removed from the current emission standards for heavy road vehicles.

D.3. LDV and diesel-engined passenger cars

A comparison of the PM_{10} emission standards for Euro4 diesel-driven passenger cars and for Euro5 Heavy Duty Vehicles shows that the PM_{10} standards for Euro4 diesel cars are approximately four times higher. Assuming the Euro5 technology, which uses a particulate trap and will be applied to new Heavy Duty Vehicles from 2008, will also be applied to diesel passenger cars and assuming these Euro5 standards will come into force from 2010, the PM_{10} emitted by diesel passenger cars in 2020 will decrease by almost 70% (~ 0.3 ktonnes).

If the same measures are implemented for Light Duty diesel Vehicles (vans), PM_{10} emissions in 2020 will decrease by another 0.4 ktonnes.

[1] Central Commission for Rhine Navigation

D.4. Mobile machines

The EC current legislation scenario assumes the implementation of both the first stage (1998) and the second stage (2000/2003[2]) of the emission legislation for mobile machines. The average PM_{10} emission rate of mobile machines is 1.8 g/kg fuel in 2010 and 0.8 g/kg fuel in 2020. If it is possible to harmonise emission legislation for mobile machinery and heavy duty road vehicles from 2005 (when Euro4 comes into force for HDV), the PM_{10} emitted by mobile machines in 2010 could be reduced by almost 20% (~ 0.3 ktonnes). In 2020, PM_{10} emissions from mobile machines could be decreased by approximately 55% (~ 0.5 ktonnes).

D.5. Wear of tyres, brake linings and road surfaces

PM_{10} emissions due to the wear of tyres, brake linings and road surfaces become increasingly important in the coming decades. The share of these emissions in total PM_{10} emissions will increase from 10% in 1995 to 30% in 2010. No technical measures for reducing emissions due to wear were found in the literature. Only a reduction in traffic volumes would reduce these PM_{10} wear emissions. It has to be stressed that the emissions database does not include the re-suspension of PM by vehicle-induced turbulence. In mass contribution this fraction is substantial.

D.6. Other source categories

D.6.1. Gaseous fuels for HDV

A switch from diesel to gaseous fuels like LPG or CNG would produce a substantial decrease in PM_{10} emissions from heavy duty vehicles. Presently, the best CNG-fuelled heavy-duty engine (without particulate trap) emits around 0.03 g/kWh PM_{10}, which is equal to the Euro5 emission limit (Bunting, 1997). Given the fact that the average particle size of exhaust PM_{10} from diesel- and LPG/CNG-fuelled engines are similar, the use of a particulate trap will probably also reduce the PM_{10} emissions of CNG engines by 80% (Kampman *et al.*, 2001). So, PM_{10} emissions from LPG/CNG-fuelled heavy duty vehicles with particulate traps could be five times lower than Euro5 emission limits. The additional cost of LPG/CNG-fuelled engines compared with diesel engines is approximately EUR 10 000–25 000, depending on production volume (Kampman *et al.*, 2001).

If, from 2005, 5% of HDV sales consist of LPG- or CNG-fuelled vehicles, PM_{10} emissions from heavy duty vehicles will have decreased by less than 1% in 2010 (0.0 ktonnes) and 4% in 2020 (0.0 ktonnes). If 25% of sales consist of LPG or CNG vehicles, PM_{10} emissions will have decreased by less than 5% in 2010 (0.0 ktonnes), and 20% in 2020 (0.1 ktonnes).

[2] The commencement date depends on engine size. For engines between 18 and 37 kW engine power, phase 2 came into force on 1 January 2000, for larger engines (37–75 kW) it comes into force on 1 January 2003.

D.6.2. Retrofit of particulate traps for HDV

The retrofitting of a particulate trap on pre-Euro4 Heavy Duty Vehicles without traps[3] will reduce PM_{10} emissions from these vehicles by approximately 80%. The cost of a particulate trap is assessed at EUR 5000 (Kampman *et al.*, 2001).

If in 2010 100% of all Euro2 and Euro3 Heavy Duty Vehicles are equipped with a particulate trap, retrofit or otherwise, PM_{10} emissions will have decreased by 0.4 ktonnes in 2010. In 2020, the effect of retrofitting old vehicles will have been reduced to 0.1 ktonnes, because in 2020 the share of older HDV in total kilometres will be only 2% (Van Beek *et al.*, 1997).

D.6.3. Emulsified diesel fuel for road vehicles

According to the American companies Lubrizol and Caterpillar, a mixture (emulsion) of diesel fuel and water (15–20%) reduces PM_{10} emissions by 20–65%. These effects have been confirmed by measurements performed (on vehicles without particulate traps) by the independent California Air Resources Board (CARB) (Zuidema, 2001). Because emulsified diesel fuel reduces particle formation in the engine, we are assuming that particle emissions from Euro4 and Euro5 diesel vehicles with particulate traps will also benefit from emulsified diesel fuel. If this fuel fully replaces conventional diesel in the Netherlands by 2010, a reduction in PM_{10} emissions of between 0.4 and 1.3 ktonnes could be possible in 2010, and between 0.4 and 1.1 ktonnes in 2020, depending on the actual effect of Lubrizol on PM_{10} emissions.

D.6.4. Further reduction of sulphur content to 10 ppm

European manufacturers stress that sulphur levels in fuel need to be cut in 2005 from the agreed 50 ppm to 10 ppm in order to be able to meet future emission standards. The NO_x catalysts especially are extremely sensitive to sulphur, and heavy duty vehicles need these catalysts to meet the Euro5 emission standards. It is questionable, however, whether lowering the sulphur content will reduce the PM_{10} emissions of older vehicles in the fleet. Measurements performed during the Auto Oil Programme suggested that reducing the sulphur content in diesel fuel from 50 ppm to 4 ppm did not cut the PM_{10} emitted by Euro2 diesel passenger cars significantly (ECMT, 2001). Therefore, no effect has been attributed to this measure.

D.6.5. Lowering sulphur content of residual oils for seagoing ships

Seagoing ships using residual oils emit approximately 6–8 g/kg PM_{10} (Lloyd's, 1995) (Kean *et al.*, 2000). Cutting the sulphur content of residual oil from its current 3 mass percent to 1.5 mass percent reduces the PM_{10} emitted by seagoing ships in Dutch ports by almost 60%. This sulphur cut has already been included in the CLE scenario. However, it is possible to reduce the sulphur content of residual oil even further. If the maximum sulphur content is reduced from 1.5% to 0.5%, PM_{10} emissions from seagoing ships in Dutch ports will have been cut by another 15% (0.1 ktonnes) in 2010.

[3] From Euro4, which comes into force in 2005/2006, new diesel-engined heavy duty vehicles need particulate traps to meet the PM_{10} emission standards.

D.6.6. Particulate traps on European ships

Technically speaking, particulate traps, which reduce PM_{10} emissions by 80–90%, can also be used for seagoing ships. According to Statistics Netherlands (CBS, 2001) around 15% of all seagoing ships visiting the Netherlands sail under Dutch colours, 50% under European colours. Retrofitting particulate traps to all European-flagged seagoing ships that visit the Netherlands would, if this were possible, result in a 0.4 ktonne reduction in PM_{10} emissions from seagoing ships in Dutch ports in 2010.

D.6.7. Fuel cells in passenger cars

Vehicles powered by fuel cells only emit water vapour. When petrol or methanol is used in the fuel cell instead of hydrogen, fuel cells also emit CO_2. Fuel cell-powered passenger cars are not yet commercially available, although most manufacturers are currently developing these cars and are planning to introduce them onto the market from 2005. Assuming the market potential of fuel cell-powered passenger cars shows linear growth – from 0% in 2005, 10% in 2010 up to 50% in 2030 (annual market potential growth is 2%), the PM_{10} emission reduction in 2010 will be negligible. This is because in 2010 only 3% of the total passenger car fleet will consist of cars powered by fuel cells. In 2020, the reduction will amount to 0.1 ktonnes and in 2030 to 0.2 ktonnes. Assuming a larger potential annual market growth of 4% per year (market potential is 20% in 2010, 60% in 2020, 100% in 2030), PM_{10} emission reduction will amount to 0.2 ktonnes in 2020.

D.6.8. Electric cars

Electric cars are currently available. However, the competition between electric and fuel cell-powered cars will probably be decided in favour of the latter because of the limited travelling range of electric cars. For this reason, electric cars are not expected to reduce PM_{10} emissions significantly by 2010 and beyond.

D.6.9. Road pricing

The Dutch government is planning to adopt a road pricing policy before the end of this decade in order to reduce congestion and diminish the environmental impact of road transport. A recent assessment of the effects on PM_{10} emissions made by Dijkstra *et al.* (1999) concludes that road pricing for Heavy Duty Vehicles (16 eurocent/km) will probably not result in a decrease in PM_{10} emissions and might even lead to an increase – due to the switch to rail transport and inland shipping resulting from road pricing. Assuming low-emission controls for rail transport and inland shipping, as in the CLE scenario, emissions per tonne kilometre for rail transport and inland shipping in 2010 will probably be higher than for road transport. Road pricing for passenger cars (2 eurocent/km) will probably produce only minor PM_{10} reductions (< 0.1 ktonnes in 2010). Higher levels of road pricing will lead to higher reductions for passenger cars. Road pricing is not considered capable of reducing PM_{10} emissions significantly by 2010.

D.7. Summary of results and conclusions

D.7.1. Realistic penetration

Table D.1 presents the effects on PM_{10} emissions of individual measures and the package of measures assuming a realistic penetration of emission control measures. All measures together will reduce PM_{10} transport emissions in 2010 by between 15 and 20% in comparison with the current legislation scenario. The maximum reduction in 2020 and 2030 will be larger, 35% and 40% respectively.

*Table D.1 Effects of individual measures and effect of package of measures in the case of **realistic penetration** of technologies (ktonnes).*

Category	Measure	2010	2020	2030
Total emission in EC current legislation scenario		11.4	11.5	13.0
Inland shipping	Euro5 from 2005	0.4	1.4	1.9
LD diesel vehicles	Euro5 from 2010	0.0	0.8	0.8
Mobile machines	Euro5 from 2005	0.3	0.5	0.8
HD diesel vehicles	LPG/CNG-fuelled engines with traps (sales share: 5–25% from 2005)	0.0	0.0–0.1	0.0–0.2
HD diesel vehicles	100% retrofit particulate trap pre-Euro4	0.4	0.1	0.0
Diesel road vehicles	lubrizol (emulsified diesel fuel)	0.4–1.3	0.4–1.1	0.4–1.4
Seagoing ships	lowering sulphur in residual oil from 1.5 to 0.5 mass percent	0.1	0.1	0.2
Seagoing ships	particulate traps (50% retrofit)	0.4	0.4	0.5
Passenger cars	fuel cells (market potential grows to between 50/100% in 2030)	0.0–0.0	0.1–0.2	0.2–0.4
Total effect on implementation of all measures [a) b)]		2.0–2.6	3.4–3.8	4.4–5.0
Total emission assuming all measures are implemented		8.7–9.4	7.7–8.1	8.0–8.6

a) The total effect is less than the sum of the effects of individual measures.
b) The effect of measures in the GC current legislation scenario does not differ significantly from that in the EC scenario.

D.7.2. Full penetration in 2010

Table D.2 shows the effect on PM_{10} emissions in the case of full penetration of new technologies. In this hypothetical situation it would be possible to reduce PM_{10} emissions from transport by around 50% in 2010 in comparison with the current legislation scenario.

*Table D.2 Effects of individual measures and effect of package of measures in the case of **full penetration** of technologies in 2010 (ktonnes).*

Category	Measure	2010
Total emission in EC current legislation scenario		11.4
Inland shipping	Euro5 from 2005	1.8
LD diesel vehicles	Euro5 from 2010	1.2
Mobile machines	Euro5 from 2005	1.4
HD diesel vehicles	LPG/CNG-fuelled vehicles (sales share: 100% from 2005)	0.1
HD diesel vehicles	100% retrofit particulate trap pre-Euro4	0.4
Diesel road vehicles	lubrizol (emulsified diesel fuel)	1.3
Seagoing ships	lowering sulphur in residual oil from 1.5 to 0.5 mass percent	0.1
Seagoing ships	particulate traps (100% retrofit in 2010)	0.7
Passenger cars	fuel cells (market potential grows to 100% in 2030)	0.0
Total effect on implementation of all measures [a) b)]		5.8
Total emission assuming all measures being implemented		5.6

a) The total effect is less than the sum of the effects of individual measures.
b) The effect of measures in the GC current legislation scenario does not differ significantly from that in the EC scenario.

D.7.3. Conclusions

In the past twenty years PM_{10} exhaust emissions from road vehicles have decreased by almost 60% despite the fact that transport volumes have increased. However, in the same period both PM_{10} emissions from the wear of tyres, road surfaces and brake linings and exhaust PM_{10} emissions from non-road transport have increased as a result of the rise in traffic volume. Nevertheless, total transport emissions decreased by 40% between 1980 and 1998. The share of wear PM_{10} emissions and non-road exhaust emissions in total transport PM_{10} emissions increased from 25% in 1980 to almost 50% in 1998.

Assuming current legislation (measures which were agreed upon before 1 January 2000), which focuses mainly on emission control for road vehicles, PM_{10} exhaust emissions from road vehicles will be reduced by 80%, whilst PM_{10} emissions from tyre, road and brake wear will increase by more than 50%. Total transport PM_{10} emissions will decrease by 45% between 1998 and 2010. After 2010, transport volumes will increase further, whereas emission control policy (current legislation) will no longer be able to reduce emission factors. In the current legislation scenario, PM_{10} emissions from transport will therefore be higher in 2020 and 2030 than in 2010.

Additional measures to reduce PM_{10} emissions will be able to reduce transport PM_{10} emissions in 2010 by 15–20% compared with current legislation, assuming a realistic penetration of technologies. This emission reduction corresponds to 2–3 ktonnes PM_{10}. These measures consist of retrofitting particulate traps to trucks and seagoing ships and of tightening emission standards for inland ships and mobile machinery to an Euro5 level, for example. The maximum potential of these additional measures in 2010 (100% penetration in 2010) is a reduction in emissions of almost 50%, which corresponds to 5–6 ktonnes PM_{10}.

D.8. Emission of Ultrafines (UF)

This Section deals with emissions of ultrafine particles (UF) from traffic and transport. Subsection D.8.1 presents some general information about the size distribution of PM_{10} originating from the combustion engine and from wear. Subsection D.8.2 gives some data on particle numbers from different engine technologies.

D.8.1. Size distribution of PM by traffic

From a study conducted by Färnland *et al.* (2001) it can be concluded that the average diameter of diesel exhaust particles (DEP) is around 0.06 µm, well within the UF range, which starts below 0.1 µm. Färnland and colleagues also found that petrol cars emit relatively more UF particles than diesel cars, meaning that the average size of particles produced by petrol-fuelled cars is smaller. ACEA also concludes that the average diameters of diesel exhaust particles (DEP) lie between 0.055 and 0.08 µm (ACEA, 1999). There is no significant difference between conventional indirect-injected diesel engines and advanced direct-injected diesel engines, though. ACEA gives 0.03 µm for the average size of particles from conventional indirect-injected petrol engines, but the average size of particles from direct-injected petrol engines, which will probably soon dominate the passenger car market, is comparable to diesel particles, i.e. 0.06 µm. This size difference between petrol and diesel particles theoretically means that even if the total particle mass emitted by diesel cars were eight times higher than the mass emitted by petrol cars, the total numbers of UF would probably be fairly comparable[4].

Particles originating from the wear of tyres and road surface are in general considerably larger than 1 µm (QUARG, 1996). Particles from the wear of brake linings are smaller, the average diameter being around 1.5 µm. However, 25–45% of the total mass of airborne particles is smaller than 0.1 µm (Bhagwan *et al.*, 2000).

D.8.2. Particle numbers per vehicle kilometre

There is some evidence that not only PM_{10} but also the numbers of UF are associated with health effects. This section gives a quantification of the numbers of particles emitted by both petrol- and diesel-fuelled cars for different engine technologies.

Table D.3 Numbers of particles per vehicle kilometre ($1* 10^{12}$)

Fuel type	Engine technology	[ACEA, 1999]	[Färnland *et al.*, 2001]	[TNO, 1997]	[TNO, 2000]	[Concawe, 1998]
Petrol	IDI	0–5	0–40	n.d.	0–15	0–20
	DI	10–35	5–85	n.d.	n.d.	n.d.
Diesel	IDI	50–100	n.d.	n.d.	n.d.	120–160
	DI	35–65	100–160	30–40	25–40	n.d.
	DI + trap	n.d.	30	n.d.	n.d.	n.d.

n.d. = no data

[4] Assuming all diesel and petrol particles have an aerodynamic diameter of 60 and 30 nm respectively and the specific weight of both particles is equal.

As Table D.3 shows, conventional (IDI) petrol cars in general emit fewer particles than diesel cars. The difference depends on the vehicle speed: for low driving speeds diesel cars emit around 1000 times more particles than petrol cars, while for high speeds diesel cars emit only four times more particles than petrol cars (Concawe, 1998). Also, Table D.3 shows that the range in emitted particle numbers is wider for petrol than for diesel cars. Research by SNRA concluded that the highest emitting petrol car emitted 2000 times more particles than the lowest emitting petrol car. In the Netherlands, vehicle fleet emissions in particle numbers per vehicle kilometre were estimated to be $9*10^{15}$ at the end of the seventies (Buringh, 1980). The variation coefficient was 20% for the larger particles and 70% for particles of 0.01 µm. These historical numbers seem to be considerably higher than the current emission estimates of Table D.3. Other current estimates by ECN and TNO (Ten Brink *et al.*, 2000) of emissions of particle numbers by traffic come to $7*10^{14}$ for LDV and $200*10^{14}$ for HDV per vehicle kilometre.

A possibly alarming conclusion that can be drawn from Table D.3 is that modern DI petrol cars emit far more particles than conventional IDI petrol cars. DI petrol cars emit almost as many particles as diesel cars and, possibly, even more than modern diesel cars with particulate traps. Because diesel cars will have to have particulate traps from 2005 (Euro4) and because the particle emissions from petrol cars, DI or other, will not be included in agreed European emission legislation up to 2010, petrol cars will probably become a more dominant factor in future emissions of particle numbers.