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HEALTH IMPACT OF PM_{10} AND OZONE IN 13 ITALIAN CITIES

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Abstract

Over the last few decades, the evidence on the adverse effects on health of air pollution has been mounting. A broad range of adverse health outcomes due to short- and long-term exposure to air pollutants, at levels usually experienced by urban populations throughout the world, are established.

This report estimates the health impact of PM_{10} and ozone on urban populations of 13 large Italian cities. To do so, concentration-response risk coefficients were derived from epidemiological studies, and 25 adverse health outcomes and different exposure scenarios were considered. Average PM_{10} levels for the years 2002–2004 ranged from 26.3 $\mu g/m^3$ to 61.1 $\mu g/m^3$. The health impact of air pollution in Italian cities is large: 8220 deaths a year, on average, are attributable to PM_{10} concentrations above 20 $\mu g/m^3$. This is 9% of the mortality for all causes (excluding accidents) in the population over 30 years of age; the impact on short-term mortality, again for PM_{10} above 20 $\mu g/m^3$, is 1372 deaths, which is 1.5% of the total mortality in the whole population. Hospital admissions attributable to PM_{10} are of a similar magnitude. Also, the impact of ozone at concentrations higher than 70 $\mu g/m^3$ amounts to 0.6% of all causes of mortality. Higher figures were obtained for the effects on heath that result in morbidity.

The magnitude of the health impact estimated for the 13 Italian cities underscores the need for urgent action to reduce the health burden of air pollution. Compliance with European Union legislation can result in substantial savings, in terms of ill health avoided. Also, local authorities, through policies that aim mainly to reduce emissions from urban transport and energy production, can achieve sizeable health gains.

Keywords

ENVIRONMENTAL EXPOSURE; ENVIRONMENTAL MONITORING;
AIR POLLUTANTS – adverse effects; OZONE – adverse effects; RISK ASSESSMENT;
HEALTH STATUS INDICATORS; URBAN HEALTH; ITALY.

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Executive Summary

Over the last few decades, a growing body of evidence points to ambient air pollution as a cause of adverse effects on health. The vast scientific literature on the subject includes epidemiological, clinical and toxicological studies, and research has systematically documented a broad range of adverse health outcomes, ranging from respiratory symptoms to mortality from cardiopulmonary diseases and lung cancer. These outcomes result from both short- and long-term exposure to air pollutants, at levels usually experienced by urban populations throughout the world, in both developed and developing countries. In support of the plausibility of the observed associations, clinical and toxicological studies have provided significant information on pollutant specific effects and possible mechanisms for these effects. Research continues to progress and, though many questions still need answers, air pollution is one of the most developed subjects today in the field of environmental health.

Thanks to this solid evidence and to the good quality of ambient monitoring networks, which provide daily measurements of air pollutants, it is now possible to reliably assess the health impact of air pollution on urban populations. Studies like the present one use existing evidence to estimate the proportion of mortality and morbidity (cases attributable to air pollution) that could be prevented if average ambient concentrations were reduced to target concentrations.

In 1998, the WHO Regional Office for Europe first estimated the health impact of particulate matter with an aerodynamic diameter smaller than 10 microns (PM_{10}) on the population of the eight largest Italian cities. Given the magnitude of the impact in this assessment, the continuing scientific and policy debate and the growing evidence on the adverse effects on health of air pollution, the Italian Agency for Environmental Protection and Technical Services commissioned the WHO Regional Office for Europe to update of the first study.

This new study does the following.

- It updates the results to 13 Italian cities with populations of more than 200 000 inhabitants – Turin, Genoa, Milan, Trieste, Padua, Venice-Mestre, Verona, Bologna, Florence, Rome, Naples, Catania, Palermo – with an overall population of about 9 million people, 16% of the total national population.
- It uses health data from national statistical sources and from consolidated international literature.
- It considers pollutant data for the triennium 2002–2004.
- It estimates the exposure of urban populations to PM_{10}, based on data from traffic and background monitoring stations.
- It broadens the analysis to include ozone and estimates its separate health impact.
- It uses concentration–response risk coefficients from epidemiological studies updated to November 2005.
- It considers 25 adverse health outcomes, including cause-specific chronic and acute causes of mortality and several morbidity end-points.
- It describes the health impact of PM_{10} and ozone, in terms of deaths and cases attributable to these air pollutants and in terms of years of life lost that could be prevented under different alternative scenarios: the reduction of the average

concentration of PM_{10} to 20 µg/m^3, 30 µg/m^3 and 40 µg/m^3 or by 10% in every city; and the reduction of the concentration of ozone to 70 µg/m^3.

- It presents detailed results by age groups and sex.

Air pollution has a large impact on health in Italian cities. In the period 2002–2004, average yearly PM_{10} concentrations range from 26.3 µg/m^3 (Trieste) to 61.1 µg/m^3 (Verona), with a population weighted mean of 45.3 µg/m^3. 8220 deaths a year, on average, were attributed to PM_{10} concentrations above 20 µg/m^3. This is 9% of the mortality for all causes, excluding accidents, in the population older than 30 years of age. This figure is estimated by considering the long-term effects on mortality. Considering the short-term effects on mortality (within a week after an exposure), the impact of PM_{10} above 20 µg/m^3 was 1372 deaths or 1.5% of the total mortality in the whole population. Concentrations measured in Italian cities during the years 2002–2004 were higher than the European average concentration, and so were, proportionately, the health impacts.

The greater detail now available in the literature on the effects of particulate matter on mortality allows a breakdown, by cause of death, in both the long and short term. Long-term impact on mortality includes lung cancer (742 cases a year), infarction (2562 cases a year) and stroke (329 cases a year). Short-term impact on mortality includes cardiovascular diseases (843 cases a year) and respiratory diseases (186 cases a year).

Large numbers of cases attributable to these pollutants were estimated for other outcomes, including morbidity in children and adults (such as bronchitis, asthma and respiratory symptoms), hospital admissions for cardiac diseases and respiratory conditions, and ill health that results in restricted activity and in the loss of work days. For Italian cities, these impacts are sizeable, with estimates in line with those obtained in analogous impact assessments in Europe and the Americas.

Unlike the previous assessment, the present one includes the impact of ozone. Ozone is a pollutant of growing concern, especially in southern European countries. The concentrations observed are on the increase, and their adverse effects on health are being more firmly established. Using the SOMO35 indicator as the standard for concentrations, ozone was estimated to have a yearly impact of 516 deaths in Italian cities (0.6% of the total mortality), with a loss of 5944 years of life. This impact adds to that of particulate matter, because the two pollutants are uncorrelated and are used as independent indicators of air quality.

The health impact of particulate matter and ozone represent important public health issues. The burden of disease is great at the individual and family level, among adults and children, and includes premature death, and chronic and acute diseases, such as cancer, bronchitis, asthma and the prevalence of respiratory symptoms. The burden on society is also great: loss of life due to a significant reduction in life expectancy, and the loss of economic productivity due to mild and severe impairments. Finally, it is a great burden on health care systems, because of thousands of hospital admissions.

By itself, PM_{10} is considered a good measure of the complex mix of gaseous and dust pollutants that originate from fuel combustion in vehicles and power generators, and it remains the pollutant of choice for assessing the health impact of air pollution. Epidemiological evidence continues to grow, with new studies using PM_{10} as the exposure indicator for particulate matter, and most monitoring data are presently based on PM_{10} measurements. However, it is desirable to have systematic measures of the concentrations of

finer particles, because the effects on health of particles with an aerodynamic diameter smaller than 2.5 microns, called $PM_{2.5}$, are presently well known, and fine particles can be more easily traced to emission sources: $PM_{2.5}$, for example, correlates more closely with motor vehicle traffic than does PM_{10}. It is not by chance that $PM_{2.5}$ has been routinely monitored in several European and North American countries in recent years.

The impacts estimated are likely to provide an incomplete picture of the total burden of disease. Other health end-points are also affected, but they are not included in the assessment, because the risks are not estimated reliably. Infant mortality, for example, is not included, due to the difficulties of extrapolating risks estimated in studies carried out in Latin America and Asia. Also, other health end-points are mild, difficult to measure and have positive, but unquantified risks.

The magnitude of the health impact of air pollution estimated for the 13 Italian cities of the present report underscores the need for urgent action to reduce the burden of disease in these cities and, likely, in many others. Compliance with European Union legislation results in substantial savings, by avoiding ill health, and it is important that the limits on PM_{10} introduced in Directive 1999/30/EC (European Union, 1999) are met and that they should not be relaxed.

Italy, however, is one of the European Union Member States where this may be a challenge. In 2005, in Italy, many of the major cities had reached the allowed 35 days in excess of 50 $\mu g/m^3$ of PM_{10} by the end of March. Also, only some cities are in compliance with the annual average of 40 $\mu g/m^3$ of PM_{10}, and none is in compliance with the average value of 20 $\mu g/m^3$ of PM_{10}, which is the limit to be reached in 2010.

Information on sources can be used to identify the most profitable areas of policy response. The data in the present report suggest that substantial gains can be achieved through policies aimed mainly at reducing emissions from two sources: urban transport and energy production. Emissions of PM_{10} from these sources are the main contributors to total primary emissions in Italian metropolitan areas.

Identifying specific policies for reducing concentrations is necessary. With regard to emissions of particulate matter, health gains can be obtained by reducing concentrations through different strategies. Since the association between air pollution and its adverse effects on health is linear and has no threshold, the effects of air pollution will decrease in proportion to the average concentration, for all health outcomes. So different interventions that produce the same yearly average will provide the same health benefits. In principle, this suggests that a variety of policy options are available. However, empirical data show that measures that reduce peak concentrations also reduce average concentrations (Cirillo, 2003). Thus, emissions from the main urban sources, notably motor vehicles, must be reduced substantially, through policies that aim to contain private motorized transport and promote public transport, cycling and walking. In Italian cities, special attention should also be paid to the contribution to air pollution of motorcycles, especially those with two-stroke engines.

Within the general policy goal of reducing emissions, attention should be given to local circumstances. In particular, PM_{10} concentrations observed in the present study were high in northern cities (50 $\mu g/m^3$), as compared with urban areas located in central (43 $\mu g/m^3$) and southern Italy (35 $\mu g/m^3$). These differences are likely to be due mainly to differences in transport, industrial activities, and heating-related emissions at the city level and at the regional level – together with climatic factors. For example the cities of the Po-Venetian Plain (Verona, Milan, and Padua) have high concentrations of PM_{10} (59 $\mu g/m^3$, annual average for the period 2002–2004) due to intense local urban traffic, intense regional traffic and intense

industrial activities, combined with climatic conditions that limit the dispersion of pollution. Under these circumstances, action taken by one municipality to reduce, for example, emissions from motor vehicles is likely to have modest results. Instead policy initiatives at the regional level may be needed to achieve substantial gains in reducing concentrations of air pollutants and in improving health.

Similar considerations apply to ozone. Ozone contributes a considerable additional health impact, although its impact is smaller than the one for particulate matter. Repeated epidemiological studies have demonstrated that risks to health increase linearly with ozone concentration and are observed not only on days with ozone peaks, but are also observed on non-peak days. For this reason, as with particulate matter, strategies for reducing ozone levels should target not only peak days but should also target average concentrations. Given that precursors of ozone are produced mainly by combustion processes, preventive action, again, should target emissions from transport and, where relevant, industry.

Policies directed at the traffic sector are particularly appropriate for several other reasons. Apart from the importance of emissions of primary particulate matter by traffic, other emissions from road transport (such as resuspended road dust and wear of tires and brake linings) are the main source of the coarse fraction of particulate matter ($PM_{10–2.5}$). Finally, restrictions on private motor vehicle traffic would result in a number of health co-benefits through, for example, reduction of road accidents, of exposure to noise, of psychosocial effects and through the possible increase of walking and cycling. In the case of road accidents, the number of fatal injuries recorded among residents of the 13 Italian cities in 2001 is of the same order of magnitude as the short-term impact of PM_{10}. Indeed, methods that quantify the health impacts of broad policies, rather than individual risk factors (such as air pollution), are of growing interest in the fields of environment and health.

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Abbreviations

Organizations, other entities and studies

ACS	American Cancer Society
AIRNET	Thematic Network on Air Pollution and Health
APAT	Italian Agency for Environmental Protection and Technical Services
APHEA (2)	Air Pollution and Health: a European Approach
APHEIS	Air Pollution and Health: a European Information System
ARPA	the Italian Regional Agency for Environment Prevention and Protection in (a particular region)
ARPAC	Regional Agency for Environmental Prevention and Protection in Campania
ARPA FVG	Regional Agency for Environmental Prevention and Protection in Friuli Venezia Giulia
ARPAT	Regional Agency for Environmental Prevention and Protection in Tuscany
ARPAV	Regional Agency for Environmental Prevention and Protection in Veneto
CAFE	Clean Air for Europe
ENEA	Italian National Agency for New Technologies, Energy and the Environment
EPA	United States Environmental Protection Agency
EU	European Union
HEARTS	Health Effects and Risks of Transport Systems
HEI	Health Effects Institute
HIS	Health Interview Study
ICD IX	International Classification of Disease, ninth revision
IIASA	International Institute for Applied Systems Analysis
ISAAC	International Study of Asthma and Allergies in Childhood
ISTAT	National Institute of Statistics
LFS	Labour Force Surveys
MISA	Meta-analysis of the Italian Studies on Short-term Effects of Air Pollution
MISA-2	Meta-analysis of the Italian Studies on Short-term Effects of Air Pollution 1996–2002
NCHS	United States National Center for Health Statistics
NPHS	National Population Health Survey (of Canada)
NMMAPS	United States National Morbidity Mortality and Air Pollution Study
PAHO	Pan American Health Organization
PEACE	The Pollution Effects on Asthmatic Children in Europe
SCARPOL	Swiss Surveillance Program of Childhood Allergy and Respiratory Symptoms with Respect to Air Pollution and Climate
SIDRIA(-2)	Italian studies on respiratory disorders in children and the environment (second phase)
UNECE	United Nations Economic Commission for Europe

Technical terms

AOT60	accumulated ozone exposure over a threshold of 60 ppb (120 µg/m^3)
AQCD	air quality criteria document
CI	confidence interval
CoH	coefficient of haze
COPD	chronic obstructive pulmonary disease
CrI	credibility interval
CVD	cardiovascular diseases
DALYs	disability-adjusted life-years
DM	Ministerial Decree
GAM	semi-parametric extension of the generalized linear model
GBD	global burden of disease
GLM	generalized linear model
LRS	lower respiratory symptoms
MRADs	minor restricted activity days
OR	odds ratio
PEF	peak expiratory flow
PM	particulate matter
PM_{10}	particulate matter with an aerodynamic diameter smaller than 10 microns
PM_{15}	particulate matter with an aerodynamic diameter smaller than 15 microns
$PM_{2.5}$	particulate matter with an aerodynamic diameter smaller than 2.5 microns
ppb	part per billion
RADs	restricted activity days
RR	relative risk
SEM	standard error (of estimate of mean value)
SOMO35	sum of means over 35
SOMO0	sum of means over 0
TEOM	tapered element oscillating microbalance
TSP(s)	total suspended particulate(s)
WLDs	work loss days
YLL	years of life lost
YLDs	years lived with disability

Chapter 1. Introduction

Over the last few decades, the body of evidence on the adverse effects of ambient air pollution on health has grown. Today the vast scientific literature on the subject includes epidemiological, clinical and toxicological studies. Research has systematically documented a broad range of adverse health outcomes for both short- and long-term exposure to air pollutants at levels usually experienced by urban populations throughout the world – in both developed and developing countries. Supporting the plausibility of the strong associations observed, clinical and toxicological studies have provided significant information on pollutant-specific effects and the possible mechanisms for these effects. Research continues to progress and, though many questions are still to be answered, one of the most developed subjects in the field of environmental health today is the adverse effects on health of air pollution.

Thanks to this solid evidence base, it has not only been possible (within the last decade or so) to assess the strength and degree of the associations observed (the relative risks, which are the main output of epidemiological studies), but it has also been possible to use this information to estimate the impacts on the health of selected populations. The *impact* – that is, the number of cases of ill health due to air pollution – is a function of the relative risks, the intensity of exposure of the population under study, and the prevailing mortality and morbidity rates. Although these studies do not generate new evidence, they use existing evidence to derive the burden of disease caused by air pollution. These studies follow the same principles of risk assessment – that is, where one estimates the risk associated with exposure to a given agent, expressed (for example) as the probability of developing the disease in the course of the lifetime of a subject exposed to a given level of the agent (Hertz-Picciotto, 1995). Ambient air pollution in urban settings, however, has some distinct characteristics: it comprises a mix of pollutants, many of which are correlated; it causes a variety of adverse effects on health; the relevant metric of exposure is the time-averaged concentration measured, which affects *all* subjects of the population – that is, no subjects are unexposed. These characteristics have contributed to the development of health impact assessment studies, which are based on a methodology (described in Chapter 2) conceptually equivalent to risk assessment, which is now firmly established.

The increasing availability of routinely collected data on air pollution concentration and on health statistics has fostered numerous impact assessment studies. These studies have invariably indicated that the adverse effects on health of air pollution are large. This is not surprising, given the ubiquitous nature of air pollution and the large size of the populations exposed. The estimates of its impact are impressive, and they are very compelling for public health agencies. In a recent WHO publication (Cohen et al., 2005), an assessment of the burden of disease worldwide due to urban ambient air pollution found that:

> … about 3% of mortality from cardiopulmonary disease, about 5% of mortality from cancer of the trachea, bronchus, and lung, and about 1% of mortality from acute respiratory infections in children under five years [are attributable to ambient air pollution]. … This amounts to about 0.8 million (1.2%) premature deaths. ... This burden occurs predominantly in developing countries: 65% in Asia alone. These estimates consider only the impact of air pollution on mortality (i.e., years of life lost) and not morbidity (i.e., years lived with disability), due to limitations in the epidemiologic database. If air pollution multiplies both incidence and

> mortality to the same extent (i.e., the same relative risk), then the DALYs [disability-adjusted life years] for cardiopulmonary disease increase by 20% worldwide.

Studies of national or regional populations have also been carried out. A seminal study (Künzli et al., 2000) prepared an estimate of the health impact in Austria, France and Switzerland. The study attributed more than 40 000 deaths a year to man-made particulate matter (PM) with an aerodynamic diameter smaller than 10 microns (PM_{10}). In the United Kingdom, a study carried out by the Committee on the Medical Effects of Air Pollutants (1998) calculated that 8100 deaths and 10 500 respiratory hospital admissions a year in urban areas were due to exposure to $PM_{10,}$ and 700 deaths and 500 respiratory hospital admissions were due to exposure to levels of ozone over 100 $\mu g/m^3$, in both urban and rural areas. Italy, too, was among the countries that embarked on an air pollution health impact assessment. Apart from the participation of Italian cities in collaborative projects in Europe, the Ministry of Environment commissioned the WHO Regional Office for Europe to assess the impact on health of urban air pollution. Using PM_{10}, the study estimated that in the eight major Italian cities being studied, in 1998, about 3500 deaths and many more cases of disease were attributable to levels of PM_{10} over 30 $\mu g/m^3$ (Martuzzi et al., 2002). Looked at in another way, about 3500 deaths could have been prevented if PM_{10} had had an annual average concentration of 30 $\mu g/m^3$.

These figures underscore the importance of air pollution as a public health issue. They also indicate that many different impacts can be estimated, by using different metrics (such as number of deaths or proportion of mortality, life expectancy, and morbidity) and different concentration levels, hypothetically considered for comparison with observed concentrations. These hypothetical concentration levels are called *counterfactuals*. Moreover, as the evidence on the adverse effects on health of air pollution grows almost daily, the numerical coefficients to be used for health impact assessments are updated frequently, to take into account the results of new studies. Given the importance of the problem, its evolution and its complexity, the Italian Agency for Environmental Protection and Technical Services (APAT) again commissioned the WHO Regional Office for Europe to update the first assessment of Italian cities (Martuzzi et al., 2002).

The present report and assessment builds on the previous one and updates it in many ways: it covers the period 2002–2004; it covers the 13 largest Italian cities for which environmental data were systematically available (Turin, Genoa, Milan, Trieste, Padua, Venice-Mestre, Verona, Bologna, Florence, Rome, Naples, Catania, Palermo); and it is based on scientific literature published up to November 2005. As a result, the methodology is substantially updated. An important additional element is that health impacts are estimated for both PM and ozone.

The ideal summary indicator for estimating the health impact of urban air pollution is still PM. It is the pollutant associated most consistently with a variety of adverse health outcomes, ranging from acute symptoms, morbidity and premature mortality to long-term effects. These effects extend to children and adults and to a number of large, susceptible groups within the general population, including subjects already affected by respiratory, cardiovascular (all cardiovascular causes for mortality and only cardiac causes for admissions) and cardiac problems. Although the risk for several health outcomes has been shown to increase with exposure to PM, there is no evidence to suggest a threshold below which no adverse effects on health would be observed. In fact, effects have been observed at levels nearing the natural background, about 6 $\mu g/m^3$.

A large amount of the epidemiological evidence is based on studies that use PM_{10} as the indicator of exposure to PM, and most monitoring data is presently based on measurements of

PM_{10}. Given the very high correlation between PM_{10} and other air pollutants, including finer particles, PM_{10} is considered a good measure of the complex mix of particles and dust that result from fuel combustion in vehicles and power generators. The adverse effects on health of PM_{10} therefore reflect possible effects due to other correlated pollutants or their interactive effects. Assessments made using PM_{10} are conservative – that is, they underestimate the impact – and they prevent the double counting of events due to one pollutant that can be mistakenly attributed to other correlated pollutants.

A number of epidemiological investigations have found adverse effects on reproductive outcomes due ambient air pollution, including spontaneous abortion, fetal growth, preterm delivery and infant mortality (Xu, Ding & Wang, 1995; Wang et al., 1997; Woodruff, Grillo & Schoendorf; 1997, Pereira et al., 1998; Dejmek et al., 1999; Ritz & Yu, 1999; Dejmek et al., 2000; Ritz et al., 2000; Maisonet et al., 2001; Wilhelm & Ritz, 2003; Gilboa et al., 2005), along with three recent reviews of the literature (Glinianaia et al., 2004a, b; Maisonet et al., 2004). This indicative toxicological evidence and the growing epidemiological evidence for the reproductive toxicity (such as restricted fetal growth and shortened gestation) of air pollution raise the question of whether air pollution is also an environmental teratogen. The findings in this literature support the hypothesis that the developing embryo and growing fetus constitute a subpopulation susceptible to exposure to air pollution. The adverse effects include not only reducing fetal growth and shortened gestation, but also include somatic and inheritable gene mutations (Perera et al.; 1992, Perera et al., 1999; Somers et al., 2002; Samet, DeMarini & Malling, 2004; Somers et al., 2004). However, it is premature to include these outcomes in the impact assessment exercise. Although these findings are indicative, they have a lower degree of consistency than the ones that form the basis of the current methodology for assessing the adverse effects of air pollution.

The assessment of the present report includes the health impact of ozone, a pollutant of growing concern, especially in southern Europe. Ozone, however, is not routinely included in impact assessment exercises. It is not correlated with PM or other gaseous pollutants, and its health impacts can be added to those of PM. Current evidence – now much more robust, compared with a few years ago – allows quantification of the acute effects of ozone, although specific adverse effects of long-term exposure cannot be ruled out. Recent epidemiological studies on short-term exposures have described its adverse effects on health, in terms of morbidity and mortality from all causes and mortality from cardiovascular diseases. As with PM, no threshold for ozone can be assumed below which there are no effects at the population level. In recent epidemiological studies, proportionally increased risks were observed on days with so-called ozone peaks, as well as on days with average concentrations. Thus, strategies for reducing ozone concentrations would be beneficial for peaks and also for the whole summer.

This report is organized as follows. Chapter 2 describes the rationale and provides background information for making quantitative estimates of the health effects of PM_{10} and ozone, and it outlines the available data on exposure and baseline population health for the 13 Italian cities. It also describes the derivation and use of the concentration–response information from epidemiological studies. Chapter 3 summarizes the results of quantitative estimates. Finally, Chapter 4 provides conclusions, assesses the findings critically, details the uncertainties, strengths and weaknesses of the study, and addresses the implications for public policies.

Chapter 2. Materials and methods

This chapter is organized as follows. The sources of demographic and health data are described in Sections 2.1 and 2.2. The sources of environmental data, the characteristics of PM_{10} and ozone, the classification of fixed-site monitoring stations and methods to derive estimates of population exposure for the two pollutants are described in Sections 2.3–2.6. The choice of the counterfactual factors and the characteristics of concentration–response functions are explained in Sections 2.7 and 2.8. Sections 2.9–2.12 review the scientific evidence for the adverse effects of PM_{10} and ozone on mortality and morbidity and describe the choice of risk estimates applied in this report. Finally, the methods for quantifying the health impact are explained in Section 2.13.

2.1 Study population and data

The population covered in this report consists of residents of Italian cities with over 200 000 inhabitants (Table 1 and Fig. 1) for which the environmental data needed for the analysis were available. These cities are Turin, Genoa, Milan, Trieste, Padua, Venice-Mestre, Verona, Bologna, Florence, Rome, Naples, Catania and Palermo. Overall, the study population comprises about nine million people.

Demographic data by age and sex were retrieved from a national statistical database (ISTAT, 2001). For most of the analyses, the population was generally grouped by five-year subdivisions (younger than 1 year old, 1–4 years old, 5–9 years old, …, older than 95 years); larger age-group subdivisions were used to calculate years of life lost.

Fig. 1. Italian cities with a population over 200 000 inhabitants under study

Table 1. Population of major Italian cities by sex (2001)

City	Males	Females	Total
Turin	409 954	455 309	865 263
Genoa	284 959	325 348	610 307
Milan	586 128	670 083	1 256 211
Padua	96 223	108 647	204 870
Verona	119 700	133 508	253 208
Venice-Mestre	128 172	142 901	271 073
Trieste	98 179	113 005	211 184
Bologna	172 331	198 886	371 217
Florence	165 176	190 942	356 118
Rome	1 199 092	1 347 712	2 546 804
Naples	480 620	523 880	1 004 500
Catania	148 045	165 065	313 110
Palermo	328 424	358 298	686 722
Total	**4 217 003**	**4 733 584**	**8 950 587**

Source: ISTAT (2001).

2.2 Health data

Health statistics on mortality for the year 2001 were retrieved from an updated version of the *Italian Mortality Atlas* (Cislaghi, 2005). The *Atlas* contains cause-specific mortality data at the municipality level, from 1981 to 2001. As for data on population, the figures for the number of deaths were retrieved for every city by sex and age group.

Data on morbidity and hospital admissions are routinely collected in Italy but are not available from centrally maintained public databases. Morbidity data for this study were gathered from different sources, as follows.

- Data on hospital admissions for respiratory and cardiac causes were taken or derived from the "Meta-analysis of the Italian studies on short-term effects of air pollution" (MISA-2) (Biggeri, Bellini & Terracini, 2004) for a variable period of years (1996–2002; see Annex Table 1). The meta-analysis adopted a standard protocol for the analysis of hospital admissions in every city, based on selecting only emergency admissions and excluding pre-scheduled admissions. This choice, however, was slightly different in each city, depending on the variable recorded in each region (not all regions had a variable corresponding to an emergency admission). Hospital admissions for Padua were selected with the MISA-2 protocol but were not available from the MISA-2 publications; instead they were retrieved from the web (Department of Environmental Medicine and Public Health, University of Padua – Office of Hygiene, ARPAV – Padua Department, and Local Health Authority No. 16 of Padua, 2005).

- Data on the prevalence of asthma were taken or derived from the report on the second phase of the "Italian studies on respiratory disorders in children and the environment" (SIDRIA-2) (Galassi, De Sario & Forastiere, 2005).
- Data on acute bronchitis were abstracted from the first SIDRIA report (1997).

- Other morbidity data on chronic bronchitis, lower respiratory symptoms (LRS), days of bronchodilator usage for asthma in children and adults, restricted activity days (RADs), minor restricted activity days (MRADs) and work loss days (WLDs) were extrapolated from international studies and used in impact functions (see Subsection 2.13.2), following guidance provided by Hurley and colleagues (2005).

Mortality and morbidity end-points were chosen from the scientific evidence available and from recent evaluations of impact assessments. As described in the remainder of this chapter, current evidence is strongest for overall mortality (excluding accidental causes), cardiovascular disease, infarction, stroke, respiratory disease and lung cancer. Morbidity end-points included in the present study, chosen largely by adopting the methodology used by the European Commission's Clean Air for Europe (CAFE) programme (Hurley et al., 2005), include hospital admission for cardiac and respiratory diseases, bronchitis, asthma, respiratory symptoms and days with restricted activities. Details are given in Tables 2 and 3. Data on all causes of mortality, by city and sex, are reported in Table 4.

Table 2. Causes of death selected for the health impact assessment

Mortality outcomes	ICD IX code[a]	Age (years)
Chronic effects		
All causes (excluding accidents)	0-799	> 30
Lung cancer	162	> 30
Infarction	410-414	> 30
Cerebrovascular diseases (stroke)	430-438	> 30
Acute effects		
All causes (excluding accidents)	0-799	All
Cardiovascular diseases	390-459	All
Respiratory diseases	460-519	All

[a]WHO (1978).

Table 3. Morbidity outcomes selected for health impact assessment

Morbidity outcomes	Age (years)
Hospital admissions for cardiac diseases (ICD IX 390-429)	All
Hospital admissions for respiratory diseases (ICD IX 460-519)	All
Chronic bronchitis	>27
Acute bronchitis	<15
Asthma (medication use)	6–7 and 13–14
Asthma (medication use)	>15
RADs	15–64
MRADs	18–64
WLDs	15–64
LRS	5–14
LRS	>15

Table 4. Mortality from all causes of deaths (excluding accidents) for major Italian cities (2001)

City	Males	Females	Total
Turin	4 288	4 580	8 868
Genoa	3 879	4 390	8 269
Milan	6 367	7 241	13 608
Padua	934	1 190	2 124
Verona	1 139	1 319	2 458
Venice-Mestre	1 504	1 637	3 141
Trieste	1 381	1 681	3 062
Bologna	2 194	2 466	4 660
Florence	1 938	2 275	4 213
Rome	11 648	12 026	23 674
Naples	4 525	4 673	9 198
Catania	1 498	1 544	3 042
Palermo	2 673	2 933	5 606
Total	**43 968**	**47 955**	**91 923**

Source: Cislaghi (2005).

Detailed mortality data from the other specific causes of death analysed in this study are reported in the Annex (Tables 2–6).

2.3 Environmental data

Hourly data on PM_{10} and ozone were obtained for the years 2002, 2003 and 2004. Since 2002 was a colder than average year and 2003 was characterized by summer heat-waves, data for a third year (2004) were collected to stabilize the pollutant average, which is affected by climatic conditions. Monitoring stations were selected by using criteria illustrated in Section 2.6 (Subsections 2.6.1 and 2.6.2) and are reported in the Annex (Tables 7 and 8).

Data for PM_{10} and ozone have been partially retrieved through BRACE (2004), an air-quality online database created by APAT, in compliance with Commission Decision 2001/752/EC (EU, 2001) and Directive 2002/3/EC (EU, 2002).

BRACE is a user-friendly database that allows the downloading of hourly records of concentration data, as well as information about fixed-site monitoring stations (such as location, characteristics, pollutants and measurement method). However, not all Italian monitoring stations are included in the database. The missing monitoring stations for the triennium 2002–2004 needed for the analyses performed for the present report were obtained: (a) by contacting the local authorities through APAT and (b) directly from other sources – that is, either from the annual reports on air quality or from official online databases run by environmental authorities.

At the end of this process, three years of PM_{10} and ozone data were available for the following 13 cities: Turin, Genoa, Milan, Trieste, Padua, Venice-Mestre, Verona, Bologna, Florence, Rome, Naples, Catania and Palermo.

The remaining three Italian cities with populations over 200 000 inhabitants (Bari and Taranto in Apulia and Messina in Sicily) could not be included in the present report because complete series of environmental data were not available on BRACE or could not be systematically retrieved through other sources, or both.

Compared with the first WHO Regional Office for Europe report on the health impact of air pollution in Italian cities (Martuzzi et al., 2002) five more cities (three in the Veneto region (Padua, Venice-Mestre and Verona), and Trieste and Catania) have been included.

2.4 PM_{10}

Parts of this section are based on a WHO Regional Office for Europe (2005b) fact sheet.

2.4.1 Sources and components

Particulate matter is a complex combination of organic and inorganic substances, consisting of a mixture of particles in the condensed (liquid or solid) phase. These particles vary in size, composition and origin. Their properties are summarized according to their aerodynamic diameter, called particle size.

- Particles with an aerodynamic diameter smaller than 10 microns are called PM_{10} and may reach the upper part of the airways and lungs.
- Smaller or "fine" particles, with an aerodynamic diameter smaller than 2.5 microns, are called $PM_{2.5}$; these are more dangerous, because they penetrate more deeply into the lungs and may reach the alveoli.
- The coarse (or thoracic) fraction is defined as the subset of particles with an aerodynamic diameter between 2.5 and 10 microns ($PM_{10–2.5}$).

The size of the particles also determines the time they spend in the atmosphere. While sedimentation and precipitation remove PM_{10} from the atmosphere within a few hours of emission, $PM_{2.5}$ may remain there for days or even weeks. Consequently, these particles can be transported over long distances.

In many countries, $PM_{2.5}$ has been measured regularly for several years. In Italy, however, with few exceptions (Florence, for example), only PM_{10} is routinely monitored at fixed-site stations.

The major components of PM are sulfates, nitrates, ammonia, sodium chloride, carbon, mineral dust, water, metals and polycyclic aromatic hydrocarbons. Particles may be classified as primary or secondary, depending on the mechanism by which they were formed. Primary particles are emitted directly into the atmosphere through man-made (anthropogenic) and natural processes. Anthropogenic processes include combustion within car engines (both diesel and petrol), solid-fuel (coal, lignite and biomass) combustion in households, industrial activities (such as building, mining, manufacturing of cement, ceramics and bricks, and smelting), erosion of the pavement by road traffic, abrasion of brakes and tyres, and work in caves and mines. Secondary particles are formed in the air, usually by chemical reactions of gaseous pollutants; they are products of the atmospheric transformation of nitrogen oxides, emitted mainly by traffic and some industrial processes, and sulfur dioxide, resulting from the combustion of fuels containing sulfur. Secondary particles are found mostly in the fine fraction of PM.

According to PM emission inventories available from 2000, developed by the International Institute for Applied Systems Analysis (IIASA) and European Commission Member States

for the CAFE programme (Amann et al., 2005), transport and households contributed 29% and 28%, respectively, to total primary emissions of PM_{10} and 34% and 36%, respectively, to total primary emissions of $PM_{2.5}$ in the 15 countries that belonged to the EU before May 2004.

Given the high correlation between PM_{10} and other pollutants, PM_{10} is considered to be a measure of the complex mix of particles, dust and gases that result from fuel combustion in vehicles and power generators. Using PM_{10} alone for a health impact assessment avoids multiple counting: in principle, impacts can be estimated for several pollutants, but cannot be added, given the limited knowledge about the independent effects on health of various pollutants. On the other hand, this entails an underestimation of the global burden of air pollution on human health, because the correlation is not perfect. Ozone, however, is not correlated with PM_{10}, hence its impact can be calculated separately, and the two health effects can be summed.

2.4.2 WHO GUIDELINES AND EUROPEAN LEGISLATION

Given the lack of a threshold below which no adverse effects on health occur, no specific concentration value for PM has been proposed by WHO Regional Office for Europe air quality guidelines (WHO Regional Office for Europe, 2000). Adverse effects on health, however, have been observed at levels not far from natural background concentration values, about 6 $\mu g/m^3$. If there is a threshold for PM, it lies therefore in the lower band of currently observed PM concentrations in the European Region (WHO Regional Office for Europe, 2003). Because of the almost continuous production of new scientific evidence, WHO Regional Office for Europe air quality guidelines are currently being revised.

In Directive 1999/30/EC (EU, 1999), two different limits for protecting human health were introduced for PM_{10}: a limit on 24-hour means and one on 1-year averages. In the first stage (by 1 January 2005), the limit of 50 $\mu g/m^3$, calculated as a daily (24-hour) mean, was not to be exceeded more than 35 days in a calendar year, while the annual average (a less stringent target) was not to exceed the limit of 40 $\mu g/m^3$. In the second stage (to be met by 1 January 2010), the limit of 50 $\mu g/m^3$ is not to be exceeded more than seven times in a calendar year, while the annual average is not to exceed the limit of 20 $\mu g/m^3$.

2.4.3 ADVERSE EFFECTS ON HEALTH

The occurrence of a variety of adverse effects on health due to PM_{10} has been reported consistently by hundreds of epidemiological studies of different designs (such as meta-analyses and multi- and single-city studies). Although the biological mechanisms through which PM effects health are only partially understood, toxicological evidence strongly corroborates the associations observed in epidemiological studies. Toxicological studies have been reviewed by, among others, the United States Environmental Protection Agency (EPA) (2004, 2005a) and are partially described in Subsections 2.10.1 and 2.10.2. The overall evidence points strongly at the causality of the association between PM and health. This was suggested in studies published in the 1990s (van der Heijden & Krzyzanowski, 1994; EPA, 1996) and was recently reinforced by newly published influential studies (AIRNET Work Group 3, 2004; National Research Council Committee on Research Priorities for Airborne Particulate Matter, 2004; EPA, 2004, 2005a). The association has also been made for cardiovascular outcomes (Brook et al., 2004).

Concentration–response functions for mortality and morbidity outcomes (see Section 2.8) and for most of the relevant chronic and acute effects examined in other sections of this study (Sections 2.9 and 2.10) were derived from meta-analyses and multi-city studies.

2.4.4 Previous assessments in Italy

A previous WHO Regional Office for Europe study assessed the health impact of PM in eight major Italian cities in 1998 (Martuzzi et al., 2002). With an observed population-weighted average PM_{10} concentration of 52.6 µg/m^3, an estimated 3500 deaths and a wide range of non-fatal health outcomes were attributable to levels of PM_{10} concentration over 30 µg/m^3 (see Table 5 for details). Put another way, if PM_{10} average levels had been reduced to 30 µg/m^3, about 3500 deaths could have been prevented.

Table 5. Health outcomes attributable to PM_{10} concentrations above 30 µg/m^3 in major Italian cities

Health outcomes	Cases attributable to PM_{10}
All causes of mortality (excluding accidental causes) (age ≥ 30 years)	3 472
Hospital admissions for respiratory diseases	1 887
Hospital admissions for cardiovascular diseases	2 710
Acute bronchitis (age < 15 years)	31 524
Exhacerbation of asthma attacks (age < 15 years)	29 730
Exhacerbation of asthma attacks (age ≥ 15 years)	11 360
RADs (age > 20 years)	2 702 461
Respiratory symptoms	10 409 836

Source: Martuzzi et al. (2002).

Equivalent methods were applied at the regional level in a study carried out in the Tuscany region: health impacts and health and social costs were estimated for a wide range of causes in the largest cities (Chellini, 2005). Another Italian study, which focused on acute effects on health, has been published recently for the city of Trieste (Tominz, Mazzoleni & Daris, 2005). An estimate of the reduction in life expectancy due to air pollution in Italy was calculated within the CAFE programme (Amann et al., 2005): converting PM_{10} and total suspended particulate (TSP) concentration values to the $PM_{2.5}$ metric, a loss of 9 months of life attributable to fine particles in Italy (compared with 8.6 months in Europe) in 2000 was estimated.

2.5 Ozone

2.5.1 Sources and components

Ozone is the most important photochemical oxidant of the troposphere, the part of the atmosphere extending from sea level to 10 000 meters. It is a secondary pollutant – that is, it is indirectly generated by emissions sources, but is produced by a series of chemical reactions (precursors) between substances present in the atmosphere. Activated by sunlight, these reactions involve mostly nitrogen oxides and volatile organic compounds. Nitrogen oxides are emitted mostly by traffic and the production of energy and heating. Volatile organic compounds are emitted by traffic and by a wide array of products, numbering in the

thousands; examples of these include paints and lacquers, paint strippers, cleaning supplies, pesticides, building materials, and furnishings.

Concentrations of ozone are lower in busy urban areas, because it reacts rapidly with nitrogen oxides from traffic exhausts. This explains the relatively low concentrations measured by stations monitoring busy traffic. Concentrations, however, are higher in many other parts of cities (such as upper floors, parks and gardens, and residential areas with modest traffic) and in adjacent suburban and rural areas, especially during the summer and in the afternoon, when ultraviolet radiation is more intense, temperatures are higher and wind speed is lower. Daily average values are largely determined by concentrations reached in the afternoon hours. The role of temperature is relevant: in the Netherlands the 400 deaths associated with the 2003 heat-wave were probably accompanied by high levels of ozone (Fischer, Brunekreef & Lebret, 2004). The same effect was reported in Belgium (Sartor, 2004), France (Cassadou, Chardon & D'Helf, 2004) and the United Kingdom (Stedman, 2004). Indoor exposures, however, come from a few sources, such as photocopiers and electrostatic air cleaners. Because ozone can be transported for long distances by the wind, it can be considered a transboundary pollutant.

2.5.2 WHO GUIDELINES AND EUROPEAN LEGISLATION

WHO Regional Office for Europe air quality guidelines (2000), which are currently being revised, recommend an ozone guideline value of 120 µg/m^3 (for no more than 8 hours) for the protection of human health. This value is based on studies carried out on restricted groups of exposed populations for effects other than cancer or odour/annoyance. The same limit has been adopted by current European legislation on ozone, in Directive 2002/3/EC (EU, 2002), as a reference value for the protection of human health.

2.5.3 ADVERSE EFFECTS ON HEALTH

There is an increasing amount of evidence on the adverse effects on health of ozone. As epidemiological observations are replicated, a large amount of toxicological data is becoming available – toxicological studies were reviewed by the EPA in the second draft of its air-quality criteria document (EPA, 2005b), which will be published in 2006. Thus, many of the reported epidemiological associations of ambient ozone with effects on health are supported by robust evidence on biological plausibility. Recent epidemiological studies on short-term exposures (1–8 hours) to ozone, described in Sections 2.11 and 2.12, have documented the occurrence of adverse effects on health, in terms of all causes of mortality, mortality due to cardiovascular diseases and morbidity due to respiratory causes. These effects are observed mostly in the summer and are independent of the role of other pollutants. Also, an association between ozone levels and the occurrence of stroke has been found in an Asian study (Hong et al., 2002).

While some studies have found no threshold for adverse effects on health due to ozone (EPA, 2005b), others have found that a very low-level threshold may be present (Kim et al., 2004). Adverse effects on health below the WHO Regional Office for Europe guideline value for protection of human health have been reported (Anderson et al., 1996; Ponce de Leon et al., 1996), but current scientific evidence is too limited to establish a value below which there are no effects on mortality at the population level. This view was confirmed in the summary report prepared by the joint UNECE Task Force on the Health Aspects of Air Pollution for the convention on long-range transboundary air pollution (UNECE, 2004) and in a recent WHO Regional Office for Europe meta-analysis (Anderson et al., 2004). In the UNECE document, a

cut-off at 70 µg/m^3, considered as a daily maximum 8-hour mean (see Section 2.8.2), was proposed, to quantify the adverse effects of ozone on health.

Guideline values and thresholds for the chronic effects on health of ozone are unknown. Few epidemiological studies that examine all causes of mortality, mortality from lung cancer, the incidence of asthma and decreasing lung function have been carried out. The most frequent associations have been found for the decrease of lung function in children (WHO Regional Office for Europe, 2003).

Concentration–response functions used in the present study for outcomes for mortality and morbidity and for most of the relevant acute effects have been derived from meta-analyses and multi-city studies (see Sections 2.11 and 2.12).

2.5.4 Vulnerable groups

The acute effects of ozone on mortality and on hospital admissions have been shown to vary with age and to be unfavourable to the elderly (Gouveia & Fletcher, 2000; Goldberg et al., 2001), with no differences between sexes. Several other differences in susceptibility to the adverse effects of ozone on health have been observed: the effects on respiratory symptoms were higher in asthmatic children (Jalaludin et al., 2000); decreases in lung function were higher in children that spent more time outdoors (Gauderman et al., 2002); the incidence of asthma was higher in children exercising more (McConnell et al., 2002); and school absences were more frequent (Gilliland et al., 2001; Park et al., 2002; Hubbell et al., 2005). Also, the levels of ambient ozone and emergency hospital admissions for respiratory diseases are strictly connected; recently, a New Jersey research group concluded that levels of ambient ozone can be reliably predicted from asthma emergency room visits and admission data (Weisel et al., 2002).

2.5.5 Previous assessments in Italy

The CAFE project (Amann et al., 2005) used the sum of means over 35 (SOMO35) indicator (see Subsection 2.6.2) for its estimates and calculated that about 4000 premature deaths in 2010 and about 3500 in 2020 will be attributable to ozone in Italy, if no new climate-control measures were applied after 2002. The calculation combined information on the energy use related to the economic development of European countries, the costs of controlling pollutant emissions, and the characteristics of pollutant dispersion in the atmosphere.

2.6 Monitoring stations

The concentration–response coefficients used in the present study to calculate the adverse effects on health are derived from epidemiological studies that used concentration data from fixed-site monitoring stations located in metropolitan areas. Hence, the Italian network of fixed-site monitoring stations was used as the source of data on ambient PM_{10} and ozone. It has been noted that “changes in ambient air pollution as measured at fixed-site monitoring stations are a good surrogate measure of changes in the average exposure of a population attributable to outdoor sources” (AIRNET Work Group 2, 2004). Moreover, Zeger and colleagues (2000) stated that measuring the exposure of a population through data obtained from fixed-site monitoring stations is unlikely to involve a bias.

In Italy, in the last few years, the air pollution monitoring network has improved substantially. Unlike in the first WHO Regional Office for Europe report (Martuzzi et al., 2002), PM_{10} concentrations were available as direct measurements, and only in two cases (two background monitoring stations in Verona and Bologna) was it necessary to convert from values for TSPs. In these two cases, a coefficient of $PM_{10} = 0.83*TSP$ was applied, as recommended by the 1999/30/EC Directive (EU, 1999).

In Italy, the characteristics of the monitoring stations are specified by DM 20.5.91 (Italian Ministry of the Environment, 1991). The qualitative criteria that classified the monitoring stations in four groups were recently replaced by another classification, as indicated in APAT guidelines (de'Munari et al., 2004), according to DM 2.4.2002 n.60 (Italian Ministry of the Environment, 2002), as illustrated in Table 6.

The original qualitative criteria that classified the monitoring stations in four groups are as follows: type A = urban background; type B = highly dense population; type C = high traffic; and type D = suburban photochemical. The shortcomings of this scheme are that it does not recommend quantitative criteria for the location of the monitoring stations and that stations of type B and type C often overlap.

Table 6. Classification of fixed-site monitoring stations

Station type[a]	Area type	Area characteristics
Traffic	Urban	Residential Commercial
Background	Suburban	Industrial Agricultural
Industrial	Rural	Natural A combination of the preceding

Source: de' Munari et al. (2004).

[a]Any given station is classified by the combination of the three classifications.

Each fixed-site monitoring station is classified by the combination of the characteristics described in Table 6 – for example, traffic/urban/residential. Not all the combinations, however, are possible – for example, traffic/rural/commercial. Urban traffic stations, used to monitor the level of pollution from busy traffic roads or from point sources, have to be located between 4 and 10 meters from the road and at least 25 meters from traffic lights, bus stops and crossroads. To distinguish between high and middle–low traffic monitoring stations, carbon monoxide levels were examined. As in MISA-2 (Biggeri, Bellini & Terracini, 2004), a yearly average of 1.5 mg/m^3 carbon monoxide was chosen as the value, to differentiate between very busy roads and more residential zones with less intense traffic.

Urban background stations, used to monitor the hypothetical background level of pollution in urban environments, have to be located inside parks and pedestrian areas and far from traffic and industrial sources. These stations are particularly important for measuring ozone levels (see Section 2.6.2).

2.6.1 METHODS TO DERIVE ESTIMATES OF POPULATION EXPOSURE: PM_{10}

2.6.1.1 Selection of monitoring stations

As in the previous WHO Regional Office for Europe study that assessed the effects on health of air pollution in eight major Italian cities (Martuzzi et al., 2002), several criteria have been identified to select the appropriate monitoring stations. Among these criteria are the following.

- Each station must be located within the city border and close to the population centroid.
- Each station must be located far from industrial emission sources and must be representative of the general exposure of the population.
- At least two monitoring stations, if possible, must be selected for every city.

In the present study, three types of urban monitoring stations (traffic – "high" and "low", depending on carbon monoxide levels – and background) were selected to represent the general exposure of the population and to calculate the adverse effects of PM_{10} on human health, as suggested by WHO for assessing the outdoor air pollution burden of disease at the national level (Ostro, 2004). In every city, a combination of the three kinds of stations was chosen. When urban background stations for measuring PM_{10} were not available, TSP values were used and converted to the PM_{10} metric.

The criterion used to validate concentration data was that each monitoring station be considered eligible for the study only if daily data were available for more than 50% of the days. In Florence and Genoa, however, the monitoring stations did not work every day, because they were set up to measure alternatively PM_{10} and $PM_{2.5}$ or for other reasons. In these cases the validation procedure was different. In the other monitoring stations, a daily average value of concentration was considered valid only if more than 50% of hourly data were available. Also, the data validation process could not be carried out for gravimetric fixed-site monitoring stations; in that case, the average daily value reported was considered as valid. In most cases, more than 90% of daily data were valid and uniformly distributed within each year.

When hourly data were not available and only validated daily data could be retrieved, the process was different. In two cases, a daily mean was considered valid if more than 75% of hourly data were available and validated daily (Verona) or yearly (Catania) averages were provided by the local authorities (Municipality of Catania, 2003, 2004, 2005; Municipality of Verona – Environmental Division, 2006). In another case (Florence), when the efficiency of the monitoring station was between 15% and 90%, missing data were generated through a statistical procedure by the regional environmental agency (Regional Agency for Environmental Prevention and Protection in Tuscany, 2003, 2004, 2005), and yearly averages were provided. In a third case (Venice-Mestre (Municipality of Venice – Local Environmental Authority & Regional Agency for Environmental Prevention and Protection in Veneto – Air Observatory, 2003, 2004, 2005), yearly averages were provided for two stations in Padua (Regional Agency for Environmental Prevention and Protection in Veneto – Meteorological Center of Teolo Regional Air Observatory, 2005), one station in Trieste (Regional Agency for Environmental Prevention and Protection in Friuli Venezia Giulia, 2003, 2004), one in Bologna (Regional Agency for Environmental Prevention and Protection

in Emilia-Romagna, 2005) and one in Genoa (Liguria Region, Department of the Environment, Policy Division for Sustainable Development, 2005)), by the local environmental authorities or regional environmental agencies, or both.

In the next step, Pearson's correlation coefficients between daily concentrations for all the pairs of monitoring stations selected in the given period were calculated. The rationale for this is as follows: if pairs of stations measure a homogeneous exposure of the population in different parts of the city, then the daily correlation among the selected stations should be reasonably high (approximately 0.7 or more). In this way outliers or monitors measuring so-called hot spots are excluded.

For every city, the yearly value of concentration was obtained as the average concentration from the monitoring stations selected, as recommended by WHO guidelines (Ostro, 2004). The average concentrations for three years were then combined for every city in a triennium average. This final value, reported with the yearly averages in Chapter 3 (Subsection 3.1.1), was used to estimate the health impact of PM. The use of an average of the concentrations collected in several years is recommended to reduce random and systematic errors due to seasonal fluctuations or to a non-representative year (Ostro, 2004).

The monitoring stations selected for the study, using the criteria outlined above, are given in the Annex (Table 7) where, for all the cities in the study, the name and type of station, its location, zone, measurement method and source of data are reported.

2.6.1.2 Measurement methods: correction coefficients

Three different methods are used to measure PM_{10} concentrations: BETA automatic, tapered element oscillating microbalance (TEOM) and gravimetric. The method recommended by European legislation is the gravimetric method, and the risk estimates calculated in United States cohort studies and applied in this report are derived from monitoring stations using gravimetric methods. While the use of the BETA automatic method does not seem problematic (Biggeri, Bellini & Terracini, 2004) and has been certified equivalent to the gravimetric method, it has been demonstrated that the use of the TEOM method underestimates PM_{10} concentrations, especially at high levels of concentration (Moorcroft et al., 1999) and, to compensate the losses of volatile PM, a standard correction coefficient of 1.3 for annual averages is usually recommended (1.3*TEOM = gravimetric; EC Working Group on Particulate Matter, 2004). The TEOM method was used in the study period of the present report in only two monitoring stations of the city of Milan. A monthly local correction coefficient, ranging from 1 (month of July) to 1.35 (month of January), introduced by the Regional Agency for Environmental Prevention and Protection in the Lombardy (ARPA Lombardia) (2005a, b) and was applied in the present report (Annex, Table 9).

2.6.1.3 PM_{10} and $PM_{2.5}$: conversion coefficient

All chronic effects and several important risk estimates of acute effects used for assessing the impacts are based on studies of $PM_{2.5}$. Since $PM_{2.5}$ is not routinely measured yet in Italy, a conversion factor between PM_{10} and $PM_{2.5}$ data is necessary. Annual mean $PM_{2.5}$ levels are roughly two thirds of those of PM_{10}; however, variations in time and space can be substantial, with reported ratios from 0.4 to 0.8 (CAFE Working Group on Particular Matter, 2004).

In a recent report by the WHO Global Burden of Disease (GBD) project on comparative quantification of risks to health (Ezzati et al., 2004), urban air pollution was considered one of the major risk factors. In the GBD study, the basic ratio between $PM_{2.5}$ and PM_{10} was

assumed to be 0.5, even though, as the authors claim, higher and lower levels were observed (Cohen et al., 2004).

In this study (Cohen et al., 2004), a standard conversion coefficient of 0.7 was used, as recommended by the Air Pollution and Health: a European Information System (APHEIS) study (Medina et al., 2005). This estimate is based on the average value of two recent studies, weighted with standard errors. In the first study, the second position paper on PM by the CAFE group (CAFE Working Group on Particular Matter, 2004), based on 72 European locations, a ratio of $PM_{2.5}/PM_{10} = 0.65$ (standard error (SEM) = 0.09) was found; in the second study (Van Dingenen et al., 2004), based on 11 stations, a ratio of $PM_{2.5}/PM_{10} = 0.73$ (SEM = 0.15) was found.

Local conversion factors are available only for some of the cities in the present study, and all of them are between 0.5 and 0.8: Genoa (0.65) (Prati et al., 2004), Milan (from 0.62 to 0.84 (Regional Agency for Environmental Prevention and Protection in Lombardy, 2003, 2004, 2005a)), Bologna (0.8) (Zanini, 2004), Florence (from 0.5 to 0.7 (Regional Agency for Environmental Prevention and Protection in Tuscany (ARPA Toscana), 2005)) and Rome (0.58) (Marconi et al., 2004). Most of them were calculated using readings from a limited number of temporary monitoring stations; their reliability is therefore uncertain. For this reason, the following standard conversion coefficient has been applied in the present report:

$$PM_{2.5} = 0.7 * PM_{10}. \qquad \text{(Equation 1)}$$

2.6.2 METHODS TO DERIVE ESTIMATES OF POPULATION EXPOSURE: OZONE

2.6.2.1 Selection of monitoring stations

Because ozone levels in heavily trafficked urban areas are lower than in the surrounding background or suburban rural areas, due to chemical reactions with nitrogen oxides that scavenge ozone, background urban (or, if missing, suburban) fixed-site monitoring stations that provide hourly data were selected for each city (UNECE, 2004). These stations are listed in the Annex (Table 8), by name and type of station, location, zone and data source, for all the cities in the present study.

2.6.2.2 SOMO35 and SOMO0

The UNECE Task Force on Health has recently suggested a new indicator for the calculation of the adverse effects on health due to ozone. The indicator "accumulated excess concentration over the guideline value of 60 ppb [120 µg/m^3]" (AOT60) has been replaced by the SOMO35 indicator as an annual estimate of human exposure to ozone (UNECE, 2004).

For every day *i*, SOMO35 is calculated, in line with the metric used in the health studies to derive the summary estimate (Anderson et al., 2004). It uses 24-hourly running averages, from 00:00 (midnight) to 23:00, as follows:

$$Av_i = Average(h, h-1, ..., h-6, h-7) \qquad \text{(Equation 2)}$$

with h = 0 to 23.

For every day *i*, the maximum average is then considered:

$$M_i = Max(Av_i). \qquad \text{(Equation 3)}$$

As an example, the 8-hour running average for 11:00 is formed by the values from 4:00 to 11:00 (to calculate the 8-hour running averages from midnight to 6:00, hourly values of the previous day are needed).

For data validation, the so-called 75% rule has been applied: an 8-hour running average (Equation 2) was considered valid if at least 75% of its hourly values (6 of 8) were available, while every daily maximum 8-hour running average (Equation 3) was considered valid if at least 75% of the 8-hour running average values (18 of 24) on that day were available.

From Equation 3, the excesses of ozone concentrations over 70 µg/m^3 were calculated as follows:

$$Ov_i = M_i - 70 \;\; if \;\; M_i \geq 70; \; Ov_i = 0 \; otherwise \qquad \text{(Equation 4)}$$

from which the SOMO35 indicator is:

$$SOMO35 = \frac{\sum_i Ov_i}{N}. \qquad \text{(Equation 5)}$$

SOMO35 (Equation 5) is the sum of these excesses divided by *N*, the number of valid days – that is, the number of days for which a valid maximum 8-hour running average is available. For so-called valid days with ozone concentrations above 70 µg/m^3 as the maximum 8-hour running average, only the increment exceeding 70 µg/m^3 contributes to the estimation of the impact on health. No adverse effects on health for ozone concentrations below 70 µg/m^3 are considered, as shown in Fig. 2. The largest contribution to SOMO35 obviously comes from the summer months, but there are also contributions throughout the year, every day where the maximum 8-hour running average reaches 70 µg/m^3. This approach is conservative, as it effectively involves the equivalent to a relatively high counterfactual value. It is motivated, however, by the uncertainties about the shape of the concentration–response function at very low concentrations and reflects seasonal cycles (UNECE, 2004). Consistent with the general criterion guiding health impact assessment studies, this choice of approaches may underestimate the adverse effects of ozone on mortality.

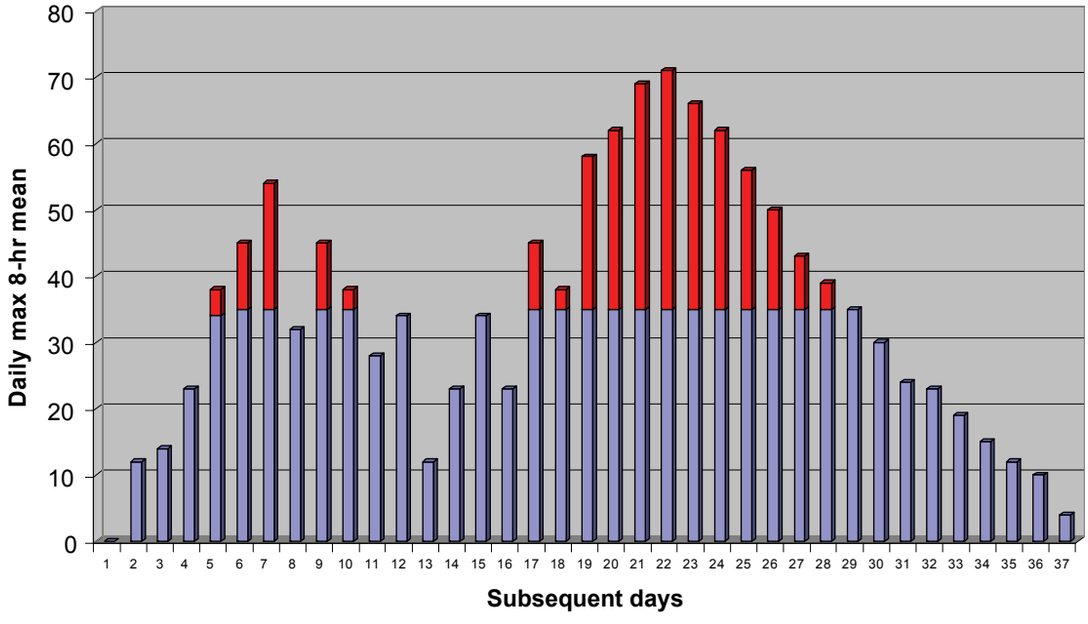
To provide an upper limit to the estimates of the adverse effects on health of ozone, a sensitivity analysis with no cut-off was proposed by the UNECE Task Force on Health

(UNECE, 2004), by applying the SOMO0 indicator, which is given by the sum of all the maximum valid daily 8-hour running average divided by *N* (Equation 6):

$$SOMO0 = \frac{\sum_i M_i}{N}. \qquad \text{(Equation 6)}$$

The SOMO0 indicator is more robust than the SOMO35 indicator and can be considered an annual average of the maximum daily running averages.

Fig. 2. Building SOMO35: excesses of daily maximum 8-hour means greater than 70 µg/m^3



Source: Amann et al. (2005).

For every year, the average values of the SOMO35 and SOMO0 indicators were calculated. The values of the SOMO35 and SOMO0 indicators for the whole period, used for the quantification of adverse effects on health, were calculated by averaging these yearly values.

The approach taken by the United States for quantifying the effect of ozone on mortality is quite different, both in terms of the threshold and the summary indicator to be used. In 1997, the EPA changed the ozone limit for human protection from 240 µg/m^3 to 160 µg/m^3 and used as an annual indicator the fourth highest daily maximum 8-hour average occurring every year, averaged over a three-year period. These guidelines are currently being revised.

As already mentioned, the use of the SOMO35 indicator has been proposed by UNECE and been applied by the CAFE programme to quantify the adverse effects of ozone on health. In the CAFE programme, it was used as a cumulative indicator of the excesses of ozone concentrations over the cut-off value of 70 µg/m^3 for the whole year, as indicated in Equation 7:

$$SOMO_{CAFE} = \sum_i Ov_i . \qquad \text{(Equation 7)}$$

In the present report, to calculate the impact on health, the SOMO35 indicator is proposed for the first time as a *proxy* for the yearly average exposure of a population over the same *cut-off* value. To date, only a few studies have assessed the impact on health of ozone, and different types of metrics have been used (Levy JI et al., 2001; Bell et al., 2004; Cassadou, Chardon & D'Helf, 2004; Hubbell et al., 2005).

2.7 Choice of counterfactual concentrations

The impact of PM on health (or the health impact of PM) refers to the proportion of ill health that is attributable to the PM concentration observed in a given city or population. This is the amount of mortality and disease that would be prevented if PM were removed altogether, which is a(n) (unrealistic) counterfactual of zero. Different counterfactuals were selected to reflect different scenarios, as described in the following subsections on PM_{10} and ozone.

2.7.1 PM_{10}

Four counterfactual concentrations were used for PM_{10}:

1. 40 µg/m^3: the scenario of compliance with the Euroepan Union limits to be reached by 2005 (and legally binding until 2010);
2. 30 µg/m^3: a scenario used in previous impact assessments;
3. 20 µg/m^3: the scenario of compliance with Euroepan Union limits to be reached by 2010; and
4. a 10% proportional reduction of the average value of PM_{10} concentration in every city: a policy-based scenario.

In the first WHO Regional Office for Europe study that assessed the effects on health of air pollution in eight major Italian cities (Martuzzi et al., 2002), three counterfactual levels were used. Reference levels of 40 µg/m^3 and 20 µg/m^3 were chosen because of limits imposed by European legislation (40 µg/m^3, to be reached by 2005; 20 µg/m^3 to be reached by 2010). A level of 30 µg/m^{3}was also chosen, converting to the PM_{10} metric the 15 µg/m^3 guideline value proposed by the EPA for $PM_{2.5}$ in the 1996 Air Quality Criteria Document (AQCD)(EPA, 1996). In the WHO Regional Office for Europe study, a $PM_{2.5}/PM_{10}$ coefficient of 0.5 was applied. In the present report, one additional counterfactual level was selected, to consider a policy-based scenario of a proportional 10% reduction in city concentrations, achievable through feasible measures for abating emissions.

2.7.2 OZONE

For ozone, the use of the SOMO35 indicator implicitly introduces a counterfactual level of 70 µg/m^3. The specification of further counterfactuals is not needed, because the structure of the SOMO35 indicator already excludes every maximum 8-hour running average below 70 µg/m^3.

2.8 Concentration–response functions

Concentration–response functions, obtained from epidemiological evidence, are used to make health impact assessments. Concentration–response functions are normally expressed in terms of the relative risk (RR) for a unit change in concentration. The concentration–response risk estimates used in the present report were derived from:

- published meta-analyses;
- pooled analyses in which the RR estimate was calculated by using averaging coefficients from different studies, by weighting each study with its uncertainty – that is, its SEM;
- individual studies, the relevance of which is acknowledged by established scientific working groups, such as WHO or the CAFE programme.

In the following sections, RRs are given for every health end-point, along with 95% confidence intervals (95% CIs), selected. The RR estimates used in this study are summarized in Table 7. Details on the derivation of these RRs are given in Sections 2.9–2.12

Table 7. Summary of RRs and 95% CIs[a]

PM_{10}-$PM_{2.5}$			
Cause	**RR**	**95% CI**	**Age (years)**
Mortality (excluding accidents)[b,c]	1.06	1.02–1.11	≥ 30
Lung cancer[b,c]	1.08	1.01–1.16	≥ 30
Infarction[b,d]	1.18	1.14–1.23	≥ 30
Stroke[b,d]	1.02	0.95–1.10	≥ 30
Acute mortality (excluding accidents)[e]	1.006	1.004–1.008	All
Acute mortality, cardiovascular causes[e]	1.009	1.005–1.013	All
Acute mortality, respiratory causes[e]	1.013	1.005–1.020	All
Hospital admissions for cardiac diseases[f]	1.003	1.000–1.006	All
Hospital admissions for respiratory diseases[f]	1.006	1.002–1.011	All
Acute bronchitis[g]	1.306	1.135–1.502	<15
Ozone			
Cause	**RR**	**95% CI**	**Age (years)**
Acute mortality (excluding accidents)[e]	1.003	1.001–1.004	All
Acute mortality, cardiovascular causes[e]	1.004	1.003–1.005	All
Hospital admissions for respiratory diseases[e]	1.005	0.998–1.012	≥ 65

[a]Adopted by the present study for selected mortality and morbidity outcomes;
[b]$PM_{2.5}$ estimates;
[c]Pope et al. (2002);
[d]Pope et al. (2004);
[e]Anderson et al. (2004);
[f]Biggeri, Bellini & Terracini (2004);
[g]Martuzzi et al. (2002).

2.8.1 PM_{10}

Cohort studies have yet to identify a threshold concentration below which PM_{10} has no effect on human health – risk persists at the lowest end of the concentration range observed in cities with low levels of pollution (WHO Regional Office for Europe, 2003). Approximate linear concentration–response functions have been found in cohort studies, with the only exception being lung cancer, for which the function is steeper at concentrations up to 13 $\mu g/m^3$ of $PM_{2.5}$ (an average concentration not observed in the Italian cities in the present study).

In time-series analyses, no-threshold linear concentration–response functions have been repeatedly observed for the adverse effects of PM_{10} on mortality and morbidity (Daniels et al., 2000; Pope, 2000; Schwartz & Zanobetti, 2000; Schwartz et al., 2001; Dominici et al., 2003a; Samoli et al., 2005). At a very high concentration, the slope of the function might be shallower, but again this does not apply to Italian cities. Consistent with this epidemiological evidence, linear RR concentration–response functions with no threshold were used in this study. This means, for example, that if the RR is 1.06 per 10-$\mu g/m^3$ increase in PM_{10} concentration, then people exposed to a concentration of 30 $\mu g/m^3$ are 6% more likely to suffer from the adverse effects on health of PM_{10} than people exposed to 20 $\mu g/m^3$; people exposed to a concentration of 40 $\mu g/m^3$ are 12% more likely to suffer from adverse effects on health than people exposed to 20 $\mu g/m^3$; and so on.

2.8.2 Ozone

A recent WHO Regional Office for Europe systematic review of the literature (Anderson et al., 2004) demonstrated that a threshold for ozone's effect on acute mortality has yet to be identified.

The shape of the concentration–response function is somewhat uncertain, however, for low levels of ozone concentrations. The UNECE study (2004) proposed that the RR coefficient has to be calculated for increases of 10 $\mu g/m^3$ in the daily maximum 8-hour average, the same metric used to obtain the SOMO35 indicator. As in a previous impact assessment of ozone (Cifuentes et al., 2001), a linear function was used to calculate the impacts on health. As indicated by the UNECE Task Force on Health (UNECE, 2004), data for all days of the year were utilized, but only days with a daily maximum 8-hour running average over 70 $\mu g/m^3$ contribute to the estimation of the adverse effects on health.

2.9 PM_{10}: health end-points – mortality

Exposure to PM increased the risk of mortality, both in the long term and through acute, short-term effects. Long-term exposures have been associated with reductions in life expectancies, due to cardiopulmonary mortality and lung cancer.

Chronic effects on health have been studied by a limited set of cohort studies that followed a defined population over time and compared the occurrence of disease with exposure levels, often estimated by ambient concentrations. Most of them were carried out in the United States and are described in detail in Subsections 2.9.1–2.9.4, because they provide the main evidence for assessing the adverse effects on health. The EPA review (EPA, 2005a) evaluated the results from the Harvard Six Cities Study and American Cancer Society (ACS) cohorts, observing significant associations between mortality and long-term exposure to $PM_{2.5}$. Based on these studies, the EPA estimated that the increased risks of mortality from all causes and from cardiopulmonary causes fall in a range of 6–13% and 6–19% per 10-$\mu g/m^3$ increment in

$PM_{2.5}$ concentration, respectively. For mortality from lung cancer, the reported estimate of the effects, in the extended analysis of the ACS cohort (Pope et al., 2002), was a 13% increase per 10-μg/m^3 increment in $PM_{2.5}$ concentration.

In Europe, only a few papers on the adverse effects of PM_{10} have been published recently. A Dutch study (Hoek et al., 2002) analysed long-term exposure to traffic-related air pollutants in a random sample of 5000 people from the full cohort of the Netherlands Cohort Study on Diet and Cancer; the study covered a cohort of people 55–69 years of age, from 1986 to 1994 (and used black smoke and nitrogen dioxide as indicators). A Norwegian group (Nafstad et al., 2003, 2004) carried out a cohort analysis, following a cohort of about 16 000 Oslo men from 1972/1973 to 1998, studying several health end-points (nitrogen dioxide was also used in this study). A French study (Filleul et al., 2005) followed up about 14 000 adults from seven French cities for 25 years, and consistent positive associations were found between lung cancer and cardiopulmonary mortality and levels of TSPs, black smoke, nitrogen dioxide and nitrogen oxide. In a Swedish study, a case-control analysis of lung cancer (1042 cases and 2364 controls were selected in a Stockholm County population) was carried out (Nyberg et al., 2000); the study analysed the levels of urban air pollution that caused an increased risk of lung cancer (nitrogen dioxide was used). Although none of these European studies measured PM directly, their measurements supplement the evidence of the studies in the United States for the effects of urban air pollution mixtures on chronic mortality.

Also, short-term exposures are correlated with acute mortality from all causes and from cardiovascular and respiratory diseases. These associations have been observed in many studies, with highly consistent results that indicate positive, statistically significant effects. A review of recent studies on short-term exposure to PM_{10} and mortality has been published by the EPA (2005a).

Acute effects of PM_{10} on health have been documented by time-series studies, which examine changes in a given health outcome over time within a specific area as air pollution levels fluctuate. As computing and statistical techniques have improved and pollution and health data have become more available, many time-series studies have been published – much more numerously than cohort studies. Time-series studies were first reviewed by Pope & Dockery (1999), who found a large set of health outcomes associated with short-term exposure to air pollution: daily mortality (from all causes and respiratory and cardiovascular causes), hospitalization for respiratory diseases (for all causes, pneumonia, asthma, chronic obstructive pulmonary disease (COPD)) and for cardiovascular disease (such as infarction and congestive heart failure).

Time-series studies generally capture deaths attributable (for all ages) to recent exposures and, using PM_{10} as the metric, the part of the adverse effects on health due to the coarse fraction of PM. The estimate of these effects partially overlaps the chronic effects estimated in populations over 30 years of age (for the $PM_{2.5}$ metric), which gives a more complete assessment of the adverse effects of air pollution, since it includes long-term, cumulative effects. This implies that, in terms of public health, the limited use of results of time-series studies underestimates the global adverse effects on health of air pollution. Key time-series studies are described in Subsections 2.9.5–2.9.7.

2.9.1 CHRONIC EFFECTS: MORTALITY FROM ALL CAUSES

2.9.1.1 Scientific evidence

Cohort studies, where populations with different levels of exposure are followed up over time, cover the long-term effects of air pollution. Unlike studies of acute mortality, a very limited number of cohort studies are available, because of the complexity and the costs of the studies.

In the Harvard Six Cities Study, a cohort of about 8000 adults, living in six United States cities with different PM_{10} concentrations, was followed up for a period of 14–16 years (Dockery et al., 1993). The effects of air pollution on mortality, controlled for individual risk factors, were studied through a survival analysis, which included Cox proportional-hazards models. The RR between the most and the least polluted city (PM_{10} concentrations, respectively, 46.5 $\mu g/m^3$ and 18.2 $\mu g/m^3$) was 1.27 (95% CI: 1.08–1.48). In another large cohort study in the United States, the ACS study, ambient air pollution data for 1980, from 151 metropolitan areas, were linked with individual risk factors in a cohort of 552 138 adults who resided in these areas in 1982 (Pope et al., 1995). Vital status (alive or dead) and cause of death were ascertained until December 1989. An RR of 1.17 (95% CI: 1.09–1.26) for a 24.5-$\mu g/m^3$ difference of $PM_{2.5}$ between the most polluted and the least polluted city was found. Results were controlled for such individual risk factors as body mass, diet, present and past tobacco smoking, occupational exposure, marital status, alcohol use, race, age, and sex. The estimate for total mortality was 1.066 (95% CI: 1.035–1.098) for a 10-$\mu g/m^3$ increase in $PM_{2.5}$ concentration (EPA, 2004).

The ACS study was subsequently updated (Pope et al., 2002), and the cohort was followed up until 1998, doubling the follow-up time to more than 16 years. Results were expressed in terms of $PM_{2.5}$ for three different periods of observation (1979–1983, a follow-up period during 1999/2000 and an average of the preceding two periods), as shown in Table 8. The relationship between long-term exposure and mortality persisted with the longer follow-up. In the three periods, RRs were statistically significant and stable, ranging from 1.04 to 1.06 for all causes of mortality. Interestingly, the RRs for cardiopulmonary causes and for lung cancer (described in more detail in Subsection 2.9.2) were higher than those of all causes of mortality, while no excess risk was observed for all the other specific causes.

An estimate based on combining the Six Cities and ACS studies was calculated by Künzli (1999) for a health impact assessment study in Austria, France and Switzerland and was also adopted in the APHEIS study (Medina et al., 2005). The meta-analytic estimate of RR was 1.043 (95% CI: 1.026–1.061) for a 10-$\mu g/m^3$ increase in PM_{10} concentration. The same estimate was used in the first WHO Regional Office for Europe report on eight Italian cities, but the lower limit of the CI was used conservatively, rather than the central estimate (Martuzzi et al., 2002).

The Health Effects Institute (HEI) also revised the two studies in the United States, because the findings involved some controversy (Kaiser, 1997). Data were independently reanalysed in two phases. The first phase involved the design of data audits, to determine whether each study conformed to the consistency and accuracy of their data. The second phase consisted of conducting a series of analyses that used alternative statistical methods. Alternative models were also used to identify covariates that might modify the relationship with particulate air pollution or identify susceptible population subgroups. The audit found that the data in the original analyses were of high quality, as were the risk estimates reported in the original publications. Results of the reanalysis confirmed those of the original studies (Krewski et al., 2000a, 2003; Willis et al., 2003).

In a third study in the Unites States, the Seventh Day Adventist study (Abbey et al., 1995a), a cohort of about 6000 nonsmoking California Seventh Day Adventists was followed for a period of 15 years. Concentration data were derived from values of TSPs, and results were inconsistent with the Harvard Six Cities and ACS studies. In a reanalysis (Abbey et al., 1999), the RRs were calculated for an increase of $PM_{2.5}$ concentration in the interquartile range (24.08 µg/m^3), for men and women separately. A positive association with PM was observed only for men (RR = 1.11). Results were not provided for both sexes together, and they could not be used in a pooled estimate.

In still another study, a national cohort of about 50 000 United States veterans, diagnosed as hypertensive in the mid-1970s, was followed up for 21 years (Lipfert, 2000): inconsistent and largely non-significant associations between PM exposure (to PM_{10}, $PM_{2.5}$, PM_{15} (particulate matter with an aerodynamic diameter smaller than 15 microns) and TSP, depending on availability) and mortality were reported.

Table 8. Adjusted mortality RRs associated with a 10-µg/m^3 change in $PM_{2.5}$

Cause of death	Adjusted RR (95% CI)[a]		
	1979-83	1999-2000	Average of the two periods
All causes	1.04 (95% CI: 1.01–1.08)	1.06 (95% CI: 1.02–1.10)	1.06 (95% CI: 1.02–1.11)
Cardiopulmonary	1.06 (95% CI: 1.02–1.10)	1.08 (95% CI: 1.02–1.14)	1.09 (95% CI: 1.03–1.16)
Lung cancer	1.08 (95% CI: 1.01–1.16)	1.13 (95% CI: 1.04–1.22)	1.14 (95% CI: 1.04–1.23)
All other causes	1.01 (95% CI: 0.97–1.05)	1.01 (95% CI: 0.97–1.06)	1.01 (95% CI: 0.95–1.06)

Source: Pope et al. (2002).

[a]Standardized by age, race, smoking, education, marital status, body mass, alcohol consumption, occupational exposure and diet.

2.9.1.2 Risk estimates used for the present study

In the present report, the average risk coefficient of the most recent Pope analysis has been applied, as suggested by the UNECE Task Force (UNECE, 2004) and as adopted by the CAFE programme (Holland et al., 2005). The risk coefficient for the average period was preferred and recommended as the most appropriate choice, because: (a) compared to the first period coefficient (RR=1.04), it reflects less random fluctuations; (b) recent exposures have probably caused more effects on mortality, as argued by Krewski (2000a); and (c) the average of the recent and older period characterizes the chronic exposure better.

For all the chronic effects under study, the observed average PM_{10} concentration has been converted to a $PM_{2.5}$ scale through the use of given coefficients (see Subsection 2.8.1).

In the present report, an RR of 1.06 (95% CI: 1.02–1.11) per 10-µg/m^3 increases of $PM_{2.5}$, derived from the ACS study, was used to assess the impact on mortality.

2.9.2 CHRONIC EFFECTS: LUNG CANCER

2.9.2.1 Scientific evidence

Recent prospective cohort studies (adjusted for tobacco smoking, occupation and other risk factors) have shown increases in the RR of lung cancer associated with exposure to particulate air pollutants. A review has recently synthesized the epidemiological evidence (Vineis et al., 2004).

The Harvard Six Cities Study (Dockery et al., 1993) estimated an RR of 1.19 per 10-µg/m^3 increase in $PM_{2.5}$ concentration. The Seventh Day Adventist study (Abbey et al., 1999) produced systematically higher RR estimates, for men and women separately (RR was 1.65 for men and 1.26, not statistically significant, for women), and was based on a very limited number of cases. Other risk estimates were produced by the update of the ACS study (Pope et al., 2002), and RR ranged from 1.08 to 1.14 per 10-µg/m^3 increase in $PM_{2.5}$ concentration.

2.9.2.2 Risk estimates used for the present study

The present report follows the UNECE (2004) recommendation, which is also consistent with EPA (2005a) recommendations. The risk coefficient of the first period of observation of the ACS study (Pope et al., 2002) was suggested as most appropriate, in consideration of the long latency time. As stated by the EPA (2004), the estimated effect of fine particles on mortality from lung cancer remained relatively stable even after adjustment for smoking status, although the estimated effect was larger and more significant for former and current smokers.

An RR of 1.08 (95% CI: 1.01–1.16) per 10-µg/m^3 increase in $PM_{2.5}$ concentration is used in the present report.

2.9.3 CHRONIC EFFECTS: INFARCTION

2.9.3.1 Scientific evidence

The follow-up of the ACS study carried out by Pope and colleagues (2002) used broader classifications for causes of death, because of concerns about potential cross-coding between cardiovascular and pulmonary deaths, and provided a set of RR values for cardiopulmonary mortality, ranging from 1.06 to 1.09 in the different time periods.

A more recent publication on the follow-up of the ACS study shows that long-term exposure to $PM_{2.5}$ increased the risk of several cardiovascular events: infarction, arrhythmias, heart failure and cardiac arrest (Pope et al., 2004). For these causes of death, a 10-µg/m^3 increase in fine PM concentration was associated with 8–18% increases in the risk of mortality (RR from 1.08 to 1.18), with comparable or larger risks observed for smokers relative to nonsmokers. In the same study, all the cardiovascular causes were analysed jointly with mortality from diabetes, giving a statistically significant RR of 1.12. Dysrhythmias, heart failure and cardiac arrest (plus cardiomyopathy, unspecified with arteriosclerosis, and related: ICD IX 420–429) gave a statistically significant RR of 1.13, while there was no significant association with hypertensive disease (ICD IX 401–405), other atherosclerosis and aortic aneurysm (ICD IX 440–441), diabetes (ICD IX 250) and all other cardiovascular diseases. Results for stroke are described separately in Subsection 2.9.4.

Only infarction (such as acute myocardial infarction, coronary atherosclerosis and other chronic ischaemic heart diseases: ICD IX 410–414) was considered in the present report

2.9.3.2 Risk estimates used for the present study

An RR of 1.18 (95% CI: 1.14–1.23) per 10-μg/m^3 increase in $PM_{2.5}$ concentrations was used as the risk coefficient of the average period.

2.9.4 CHRONIC EFFECTS: STROKE

2.9.4.1 Scientific evidence

As for infarction, the ACS study (Pope et al., 2004) is the only available study to date that provides risk estimates for stroke. A non-statistically significant RR of 1.02 (95% CI: 0.95–1.10) was estimated.

2.9.4.2 Risk estimates used for the present study

An RR of 1.02 (95% CI: 0.95–1.10) per 10-μg/m^3 increase in $PM_{2.5}$ concentration is used in the present study.

2.9.5 ACUTE EFFECTS: ALL CAUSES OF MORTALITY

2.9.5.1 Scientific evidence

As synthesized by Ostro (2004), recent studies on daily mortality and short-term exposure to PM are characterized by the use of high quality data and by advanced statistical techniques, to reduce potentially confounding influences on the results. In particular, he focuses on these aspects of the new studies:

- the use of Poisson regression models, since mortality is a rare event and can be described by a Poisson distribution;
- three or more years of daily pollution data in a given city or urban area;
- an accurate examination of the effects of day-of-the-week and daily changes in weather; and
- the use of general additive models with non-parametric smoothing, or general linear models with parametric splines, to control for time, season and weather.

Compared with a limited number of population analyses, several multi-city studies, studies of more than 100 cities and different meta-analytic studies have been carried out in recent years. Multi-city studies combine data from cities with various climates, air pollution sources or concentrations, and other risk factors. As summarized by the EPA (2005a), the advantages of multi-city studies include the following.

- The evaluation of associations in larger data sets can provide more precise estimates of effects than pooling results from separate studies.
- Consistency in handling data and specifying models can eliminate the variation due to study design.
- The modification of effects or confounding by co-pollutants can be evaluated by combining data from areas with different air pollutant combinations.
- Regional and geographical variations in effects can be evaluated.
- Publication biases or exclusion of reporting of negative or non-significant findings can be avoided.

In May 2002, United States National Morbidity Mortality and Air Pollution Study (NMMAPS) investigators discovered that part of the semi-parametric extension of the generalized linear model (GAM) programming in the S-Plus statistical software, used for time-series studies, was not entirely appropriate for this purpose. As a consequence, a wide set of studies was reanalysed with a more stringent convergence criteria (Katsouyanni et al., 2002; Samet et al., 2003): slightly smaller risk estimates were found, and the results were published in a special report by HEI (2003).

The most important recent multi-city studies and meta-analyses are reviewed in the present study. For detailed results on single-city studies, the EPA (2004, 2005a) recently carried out a complete. The adverse effects from PM for all ages were calculated by all the studies described in this section. Some of the risks estimated in the studies described in this section are shown in Fig. 3.

Evidence for the adverse effects of PM on children's health (0–4 years of age) is available, but limited: four of the six studies available were carried out in São Paulo (Saldiva et al., 1994; Pereira et al., 1998; Gouveia et al., 2000; Conceicao et al., 2001) one in Mexico City (Loomis et al., 1999) and one in Bangkok (Ostro et al., 1999). The present study does not extrapolate their risk functions to an European context because of differences in genetic factors, degree of urbanization, diet and distribution of wealth.

One of the first multi-city studies examined the Harvard Six Cities Study (Schwartz, Dockery & Neas, 1996). City-specific associations with each measure of particle pollution were estimated by a Poisson regression analysis adjusted for confounding factors, and estimates of combined effects were calculated as the inverse variance weighted mean of the city-specific estimates. Consistent associations were reported between daily mortality and daily exposures, both with PM_{10} and $PM_{2.5}$, and an RR of 1.008 (95% CI: 1.005–1.011) per 10-$\mu g/m^3$ increase in PM_{10} concentration was estimated. Klemm and colleagues (2000) replicated this study and observed the same results; the statistical reanalysis of the latter study showed lower risk estimates (Klemm & Mason, 2003).

The Air Pollution and Health: a European Approach 2 (APHEA 2) project, involving 29 European cities (Katsouyanni et al., 2001), analysed short-term effects of ambient particles on mortality, with emphasis on modifications of effects. Confounding from other pollutants, as well as from meteorological and chronological variables, were also considered. A wide set of variables that describe the city-specific pollution, climate, population and geography was utilized as potential effect modifiers. For the analyses of individual cities, generalized additive models, extending Poisson regression and using a smoothing function to control for seasonal patterns, were used. Second-stage regression models were also applied to provide quantitative

summaries of the results and to explain residual heterogeneity. The estimated increase in the daily number of deaths for all ages per 10-μg/m^3 increase in daily PM_{10} concentrations was 0.6% (95% CI: 0.4–0.8%). The recent reanalysis of the same data provided a reduced estimate of the risk (Katsouyanni et al., 2003).

In a study of the eight largest Canadian cities (Burnett et al., 2000), daily mortality rates and concurrent pollution data were analysed from 1986 to 1996. It was found that $PM_{2.5}$ was a stronger predictor of mortality than $PM_{10–2.5}$. For PM_{10}, a 10-μg/m^3 increase in daily concentration was associated with an RR of 1.007 (95% CI: 1.002–1.012).

Samet and colleagues (the NMMAPS study) analysed data (1987–1994) from a large database of the 88 largest cities in the United States (HEI, 2000a, b), applying a large set of statistical tools and sensitivity analyses. In a sub-sample of the 20 largest cities in the United States (Samet et al., 2000), an RR of 1.005 per 10-μg/m^3 increase in daily PM_{10} concentration was estimated. The same data were reanalysed with an alternative model in a more recent study (Dominici et al., 2002), and a lower risk estimate was observed (RR = 1.003).

Another multi-city study (Schwartz, 2003) used data from 10 United States cities, selected from the NMMAPS study, where PM_{10} daily data were available (in most of the cities monitoring was made on a 1-in-3- or 1-in-6-day basis). The authors reported a statistically significant association between PM_{10} and total mortality, with an RR of 1.007.

Taken together, multi-city studies in Canada, Europe and the United States reported a statistically significant association, with estimates of effects ranging from 1.003 to 1.007. Although the risk coefficients obtained by the different studies are quite consistent, recent methodological developments indicate that the magnitude of these risk estimates depends partially on the procedure adopted to model temporal patterns of exposure and confounders (HEI, 2003).

Earlier studies were summarized in two meta-analyses (Ostro, 1993; Dockery & Pope, 1994), and an approximate 0.8% mean change (ranging from 0.5% to 1.6%; RR = 1.008 (95% CI: 1.005–1.016)) in daily mortality associated with a one-day 10-μg/m^3 increase in PM_{10} concentration was estimated.

In an review on Asian studies (HEI, 2004), an RR from 1.004 to 1.005 per 10-μg/m^3 increase in daily PM_{10} concentration was estimated.

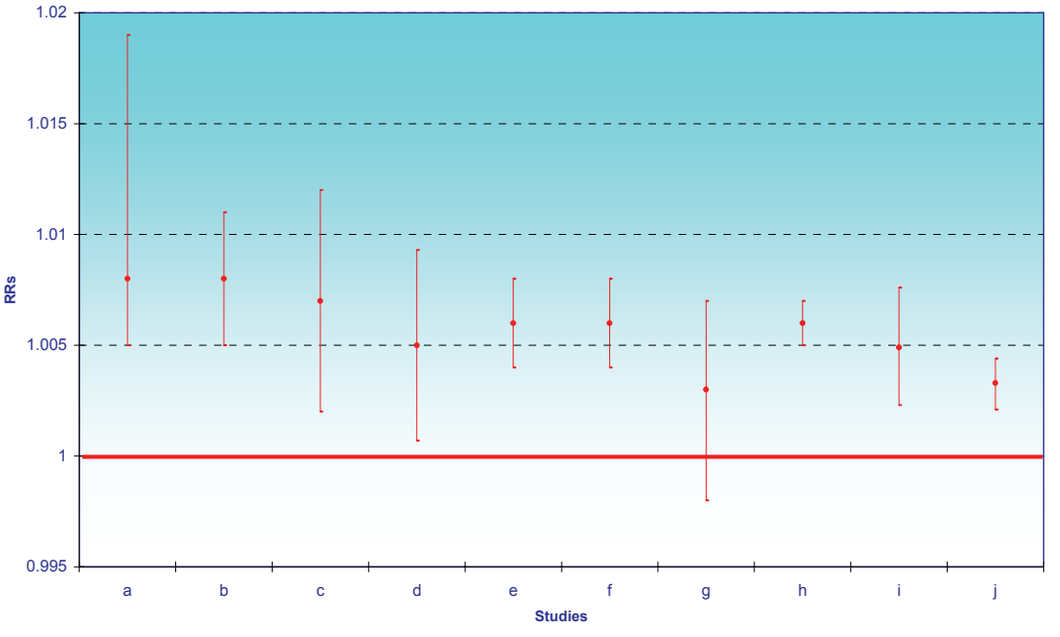
A recent review and meta-analysis of Latin American and Caribbean studies on the adverse effects on health of ambient air pollution was conducted by the Pan American Health Organization (PAHO) (2005). Quantitative summary estimates were calculated to assess the percentage increase in daily mortality associated with a 10-μg/m^3 increase in PM_{10} concentration. Based on 17 time-series studies carried out between 1994 and 2004 in four cities (Mexico City, São Paulo, Santiago and Rio de Janeiro, with a total population of more than 50 million inhabitants), an RR of 1.003 (95% CI: 1.002–1.004) from all causes was calculated from the fixed-effects model. Among the elderly, the summary estimates were higher than those for the overall population.

A WHO Regional Office for Europe meta-analysis (Anderson et al., 2004) based on 33 European studies, 21 of them already included in APHEA 2, provided an RR estimate for all causes of mortality of 1.006 (95% CI: 1.004–1.008) per 10-μg/m^3 increase in daily PM_{10} concentration.

A recent GBD study (Cohen et al., 2004) covered cities included in APHEA 2, NMMAPS (111 risk estimates) and 54 individual studies. The pooled estimate of RR = 1.006 (95% CI: 1.005–1.007) was derived and applied to all GBD baseline case-specific mortality (excluding accidents).

The MISA-2 study (Biggeri, Bellini & Terracini, 2004) carried out a meta-analysis of 15 Italian cities (a total of up to 9 million inhabitants), with data from 1996 to 2002. An increase in mortality of 0.31% (95% CI: -0.19–0.74%; RR = 1.003 (95% CI: 0.998–1.007)) per 10-$\mu g/m^3$ increase in daily PM_{10} concentration was found. Larger effects were reported for women and for the warmer seasons. Risk estimates were smaller when adjusted for nitrogen dioxide. City-specific risk estimates were also calculated for single cities, age groups and sex.

Fig. 3. Acute mortality and PM_{10}: results from multi-city studies and meta-analyses



Sources: [a]Dockery & Pope (1994); [b]Schwartz, Dockery & Neas (1996); [c]Burnett et al. (2000); [d]HEI (2000b); [e]Katsouyanni et al. (2003); [f]Anderson et al. (2004); [g]Biggeri, Bellini & Terracini (2004); [h]Cohen et al. (2004); [i]HEI (2004); [j]PAHO (2005).

2.9.5.2 Risk estimates used for the present study

Risk estimates derived from consolidated international studies (meta-analyses or multi-city studies) were favoured in the present study, and results from the WHO meta-analysis were chosen. However, several studies (on all causes of mortality, and cardiovascular and respiratory mortality) included in the meta-analysis were carried out before the HEI reanalysis was performed. An overestimate of the risk is therefore possible.

An RR of 1.006 (1.004–1.008) per 10-$\mu g/m^3$ increase in daily PM_{10} was used.

2.9.6 ACUTE EFFECTS: CARDIOVASCULAR CAUSES

2.9.6.1 Scientific evidence

Dominici, Samet & Zeger (2000) in their study of the 20 largest United States cities found an RR for cardiovascular mortality equal to 1.007 (95% CI: 1.002–1.011) per 10-µg/m^3 increase in daily PM_{10} levels.

Zanobetti and colleagues (2003) within the APHEA-2 project estimated a so-called very short-term risk of 1.007 per 10-µg/m^3 increases in daily PM_{10} concentration. An estimate of the so-called medium-term effect (deaths occurring up to 40 days after exposure) of 1.0197 was also provided. The European meta-analysis carried out by the WHO Regional Office for Europe (Anderson et al., 2004) on cardiovascular acute mortality was based on 17 estimates from studies made in France, Germany, Italy, the Netherlands, Spain and the United Kingdom. An RR estimate of 1.009 (95% CI: 1.005–1.013) per 10-µg/m^3 increase in daily PM_{10} concentration was calculated. Several Italian estimates included in the WHO Regional Office for Europe study were derived from the first edition of MISA, the Italian meta-analysis (Biggeri, Bellini & Terracini, 2001). Updated meta-analytic and city-specific estimates were calculated by MISA 2 (Biggeri, Bellini & Terracini, 2004), with an overall estimate of effect of 1.005 (95% CI: 1.000–1.010) per 10-µg/m^3 increase in daily PM_{10} concentration. A higher risk was estimated for women and for summer. Risk estimates were smaller when adjusted for nitrogen dioxide. Estimates were also available for single cities and age groups.

In the NMMAPS study, cardiovascular acute mortality was included in the wider group of cardiorespiratory mortality, and cause-specific risk estimates were not available for this study. However, the EPA (2005a) reported that risk estimates for cardiorespiratory and cardiovascular mortality in United States and Canadian cities fall in the range of 1.012–1.027 per 10-µg/m^3 increase in daily $PM_{2.5}$ concentration. The association between PM and acute mortality from cardiovascular causes appears to be generally higher than that observed for all causes of acute mortality. This difference is probably explained by the high susceptibility of people with chronic cardiovascular problems to exposure to PM.

2.9.6.2 Risk estimates used for the present study

An RR of 1.009 (1.005-1.013) per 10-µg/m^3 increase in daily PM_{10}, derived from the WHO meta-analysis, was used.

2.9.7 ACUTE EFFECTS: RESPIRATORY CAUSES

2.9.7.1 Scientific evidence

In a recent study (Pope et al., 2004) on cardiovascular mortality and long-term exposure to particulate air pollutants, none of several selected respiratory chronic effects (such as diseases of the respiratory system, COPD and allied conditions, pneumonia and influenza, and all the other respiratory diseases) were positively associated with exposure to PM (all the respiratory diseases and COPD were negatively associated with exposure to fine particulate air pollution). In contrast, evidence is more consolidated for acute respiratory effects: in numerous daily time-series studies, mortality from respiratory diseases was found to be associated consistently with daily changes in PM.

Zanobetti and colleagues (2003) calculated a very short-term effect and a medium-term effect (see Subsection 2.9.6), with RR equal to, respectively, 1.007 and 1.042.

The European meta-analysis on respiratory acute mortality and PM exposure, carried out by the WHO Regional Office for Europe (Anderson et al., 2004), was based on 18 estimates derived from French, German, Italian, Polish, Spanish and United Kingdom studies. It provided an RR estimate of 1.013 (95% CI: 1.005–1.020) per 10-μg/m^3 increase in daily PM_{10} concentration.

Several Italian estimates included in the WHO Regional Office for Europe study were derived from the first edition of MISA, the Italian meta-analysis (Biggeri, Bellini & Terracini, 2001). Updated meta-analytic and city-specific estimates were calculated by MISA 2 (Biggeri, Bellini & Terracini, 2004): an overall estimate of effect of 1.005 (95% CI: 0.991–1.017) was associated with an increment of 10 μg/m^3 in daily PM_{10} concentration. A higher risk was estimated for women and for summer. Results were smaller if adjusted for nitrogen dioxide. Estimates were also available for single cities and age groups.

In the NMMAPS study, respiratory acute mortality was included in the wider group of cardiorespiratory mortality, and the specific estimate for this cause was not available for the present study. The EPA (2005a) estimated that risk estimates for respiratory mortality fall in the range of 1.008–1.027 per 10-μg/m^3 increase in daily $PM_{2.5}$ concentration in United States and Canadian cities. As observed for acute cardiovascular mortality, the association between PM and acute mortality from respiratory causes appears to be generally stronger than that observed for all causes of acute mortality. This difference is probably explained by the greater susceptibility of people with chronic respiratory problems to exposure to PM. The estimates, however, are generally less precise, since respiratory deaths comprise a small proportion of total deaths.

2.9.7.2 Risk estimates used for the present study

An RR of 1.013 (95% CI: 1.005–1.020) per 10-μg/m^3 increase in daily PM_{10} concentration, derived from the WHO Regional Office for Europe meta-analysis, is used in the present study.

2.10 PM_{10}: health end-points – morbidity

The epidemiological evidence on the effects of PM on morbidity is based mostly on associations between short-term exposure to PM_{10}, $PM_{2.5}$, $PM_{10–2.5}$ and a wide range of outcomes (such as symptomatology, hospitalizations, and emergency department and health care visits), reflecting mainly both respiratory- and cardiovascular-related effects of morbidity.

These associations have been investigated mainly in population-based time-series analyses of changes in health outcomes, in relation to day-to-day variations in ambient PM concentrations and, more recently, through the use of case-crossover designs. This type of design is appropriate for the study of a transitory effect of an intermittent exposure on the subsequent risk of an acute adverse effect on health believed to occur shortly after exposure. Estimates of adverse effects derived from case-crossover studies are based on within-subject comparisons of exposures associated with incidence of disease events with exposures at times before the occurrence of disease.

A recent review of the adverse effects on health of PM, prepared by the EPA (2005a), summarizes the AQCD findings (EPA, 2004) on PM-related morbidity effects. The studies reviewed cover multi- and single-city analyses, numerous assessments using cardiovascular admissions/visits and evaluations of the effects of fine and thoracic coarse particles. Overall, the AQCD reports that the more precise estimates of adverse effects that result in hospitalization range from 2% to 6% and from 2% to 12% per 50 $\mu g/m^3$ increase in PM_{10} concentration, respectively, for cardiovascular and respiratory diseases. The EPA also observed that studies carried out after the previous version of the AQCD on PM (EPA, 1996), which reported associations between PM_{10} and medical consultations (that is, visits by the doctor) for respiratory diseases, offer a link between the most severe end-points (such as increased mortality and hospital admissions or emergency room visits for respiratory diseases) and less severe effects (such as respiratory symptoms and decreased lung function).

The EPA AQCD concluded that the epidemiological evidence continues to support probable causal associations between $PM_{2.5}$ and PM_{10} and both mortality and morbidity from cardiovascular and respiratory diseases, based on an assessment of the strength, robustness and consistency of results. Moreover, the AQCD stated that the new evidence from studies of mechanisms (suggesting plausible biological response pathways) and the extensive body of epidemiological evidence on associations between short- and long-term exposures to ambient PM_{10} and a range of adverse effects on health support the general conclusion that ambient thoracic particles – acting alone or in combination with gaseous co-pollutants, or both – are likely to be causally related to cardiovascular and respiratory mortality and morbidity.

Although the evaluation of epidemiological evidence is similar for the effects of PM exposure on both mortality and morbidity, quantification of the effects of short-term PM exposure on morbidity is expected to be less precise than for mortality for the following reasons.

- The database on concentration–response functions is limited.
- The way of registering hospital admissions varies across countries.
- Background country-specific rates are often difficult to obtain.

For these reasons, the present study has chosen country-specific estimates, if available, both for background rates and for concentration–response functions.

Another point to be considered, when quantifying the effects of morbidity at the local level, is whether the concentration–response functions to be used should be based on single- or multi-city studies. With respect to this, the EPA (2005a) reported that although concentration–response functions derived from multi-city studies may not accurately represent a specific assessment location, the use of functions from single-city studies does suffer the disadvantage of introducing possible publication biases. Also, single-city studies generally have lower precision. Moreover, recent multi-city studies adopt uniform methodologies to investigate the effects of PM on health, using data from multiple locations with varying climate and air pollution mixtures. These studies increase the understanding of the role of various potential confounders, including gaseous co-pollutants, on the associations observed.

The most important recent multi-city studies and meta-analyses are reviewed in the present report. For detailed results on single-city studies, the EPA (2005a) recently carried out a complete review.

Two approaches are generally adopted to estimate the impact of PM on morbidity. The first approach tries to quantify only the end-points for which strongly reliable data exist, both for concentration–response functions and for background rates. This approach – followed, for example, in the APHEIS study (Medina et al., 2005) – is biased in favour of specificity, but it entails an underestimation of the GBD attributable to air pollution, because it excludes uncertain health outcomes. In the present report, a criterion with high specificity was used for hospital admissions (see Subsections 2.10.1 and 2.10.2), because of the availability of precise city-specific background rates and national meta-analytical risk estimates.

The second approach favours sensitivity over specificity and quantifies all end-points for which air pollution likely plays a role, including those for which the uncertainty about the magnitude of the risks is large (Ostro & Chestnut, 1998). Large uncertainties about concentration–response coefficients (including lack of statistical significance) and background rates produce more uncertainties about the impact, but the overall estimate of the global health impact will be closer to reality and more suitable for cost–benefit considerations. The UNECE Task Force on Health (UNECE, 2004) stressed that, despite the imprecision of some of the risk estimates, it is appropriate to also include end-points for which evidence is not strong. Also, the CAFE programme group followed this approach. For the present report, the impact of several morbidity health outcomes was calculated, using impact functions estimated by Hurley and colleagues (2005) in the CAFE programme cost–benefit analysis. A synthesis of the impact estimates used is reported in Tables 9 and 10. Details of the derivation of the functions are described in Subsections 2.10.3–2.10.8 for PM_{10} and in Subsections 2.12.2–2.12.5 for ozone. It must be stressed that some of the impact functions derived by CAFE programme recommendations are based on a limited number of panel studies and carry some uncertainty.

Table 9. Summary of impact functions for selected morbidity outcomes due to PM_{10}–$PM_{2.5}$

Cause	Impact functions
Chronic bronchitis, adults	26.5 (95 % CI: -1.9–54.1) new cases per year per 100 000 adults ≥27 years of age per 10 μg/m^3 PM_{10} increment
Asthma (medication use), children	180 (95% CI: -690–1060) annual increase in days of bronchodilator usage per 1000 children 6–7 and 13–14 years of age per 10 μg/m^3 PM_{10} increment
Asthma (medication use), adults	912 (95% CI: -912–2774) annual increase in days of bronchodilator usage per 1000 adults ≥15 years of age per 10 μg/m^3 PM_{10} increment
RADs	902 (95% CI: 792–1013) RADs per year per 1000 adults 15–64 years of age per 10 μg/m^3 $PM_{2.5}$ increment
WLDs	City-specific impact functions (see Subsection 2.10.7.3)
MRADs	577 (95% CI: 468–686) MRADs per year per 1000 adults 18–64 years of age per 10 μg/m^3 $PM_{2.5}$ increment
LRS, children	1.86 (95% CI: 0.92–2.77) extra symptom days per child 5–14 years of age per 10 μg/m^3 PM_{10} increment
LRS, adults	1.30 (95% CI: 0.15–2.43) annual increase of symptom days per adult ≥ 15 years of age with chronic respiratory symptoms per 10 μg/m^3 PM_{10} increment

Source: Hurley et al. (2005).

2.10.1 CARDIAC-RELATED HOSPITAL ADMISSIONS

2.10.1.1 Scientific evidence

Many studies conducted in developed countries have repeatedly observed increased risks of hospital admissions for cardiovascular diseases (CVD) in association with PM air pollution. The majority of these studies have used PM_{10} as the main particle marker, due to the wider availability of data on this PM portion, as compared with monitoring data of smaller PM fractions. Also, a large number of relationships between air pollutants and adverse effects on health have been reported in locations that reflect a wide range of PM and gaseous co-pollutant concentrations. Of particular interest, as discussed in Subsection 2.9.5, are results from multi-city studies, due to their ability to aid in evaluating potential confounders or modifiers of adverse effects in a consistent way across areas that can vary for meteorological conditions, air pollutant levels and other risk factors. Numerous other studies carried out in a multiplicity of single urban sites present a more varied picture, particularly when gaseous co-pollutants have been considered in multi-pollutant models.

Ecological time-series studies (multi-city studies) on the risk of hospital admissions in relation to short-term PM exposure have been conducted independently in, Canada, European countries and the United States. The EPA (2004) has reviewed these studies.

Schwartz (1999) analysed eight United States metropolitan areas, by focusing the analysis on total hospital admissions for CVD among people older than 65 years of age. In a univariate regression analysis, consistent associations of PM_{10} with admissions for CVD were observed across the eight cities, with an increase in admissions from 3.6% to 8.6% associated with a 50-$\mu g/m^3$ increase in PM_{10} concentration (an equivalent range, for the RR, of 1.007–1.017 per 10-$\mu g/m^3$ increase in PM_{10} concentration). The pooled result of the univariate regression analysis of the adverse effects of PM_{10} was an RR of 1.01 (95% CI: 1.007–1.012), similar to the RR of 1.012 observed in a previous independent single-city study conducted in Tucson (Schwartz, 1997).

In another United States multi-city study, Samet et al. (HEI, 2000a, 2000b) analysed daily emergency-only hospital admissions for CVD, recorded from 1985 to 1994 among the elderly (older than 65 years of age), in relation to PM_{10} in 14 urban areas evaluated in the NMMAPS multi-city study. The mean risk estimate for the mean of 0–1-day lags (lags 0-1) was an RR of 1.012 (95% CI: 1.01–1.013) for CVD admissions associated with a PM_{10} increase in concentration of 10 $\mu g/m^3$. The city-specific risk estimates were not confounded by measures of socioeconomic status.

Data from the 14-city NMMAPS analysis of hospital admissions for CVD were reanalysed by Zanobetti & Schwartz (2003a), using three different methods to control for time, weather and other covariates. As compared with the original study results, the reanalysis did not find noticeable changes: the mean RR was 1.010 (95% CI: 1.008–1.012), 1.009 (95% CI: 1.007–1.012) and 1.010 (95% CI: 1.008–1.012) when reanalysed by, respectively, a semi-parametric extension of the generalized linear model (GAM) with stringent convergence criteria, a generalized linear model (GLM) with natural spline and a GLM with penalized spline.

In another reanalysis of the original study, Zanobetti et al. (2000) considered a subset of 10 cities among the 14 evaluated by the NMMAPS report, obtaining the same basic pattern of results, with strongest PM_{10} associations on lag 0 day, smaller effects on lag 1 and lag 2, and none at longer lags.

Janssen et al. (2002), in a further reanalysis (based on a GAM function) of the NMMAPS multi-city study, found that the positive association between PM_{10} and CVD-related hospitalization admissions decreased significantly with increasing percentage of homes with central air conditioning (possibly an indicator of increased exposure to ambient pollutants) (EPA, 2005a) and increased significantly with increasing contribution of PM_{10} from vehicular emissions and oil combustion. Zanobetti & Schwartz (2003a) confirmed the latter results, using alternative methods (a GLM with natural splines and a GLM with penalized splines) to control for time and weather covariates.

In the Canadian multi-city studies, the RRs of hospital admissions for CVD were almost all positive and generally statistically significant for pollutants analysed in univariate regression analyses. However, the use of estimates of the coefficient of haze (CoH), TSPs and sulfates, rather than measures of PM components, limits the ability to interpret and compare results. Although the studies by Burnett and colleagues (1995, 1997a, 1997b, 1999) examine the adverse effects of PM in conjunction with multiple gaseous pollutants, the inconsistent use of alternative PM metrics in the various analyses confuses the overall picture. As a general observation, the association between cardiovascular outcomes and PM in multi-pollutant analyses lacks robustness. For instance, in the Toronto summer analysis, gravimetric PM variables were not robust predictors (Burnett et al., 1997a), while CoH did perhaps reflect the influence of primary motor vehicle emissions. On the other hand, this interpretation differs noticeably with the lack of robustness for CoH in the 10 city analysis (Burnett et al., 1997b).

The APHEA-2 European multi-city study (Le Tertre et al., 2002a) analysed the association between airborne particles and hospital admissions for cardiac causes (ICD IX 390–429) in eight European cities with an overall pooled population of about 38 million people. Control for long-term trend, season, influenza epidemics and meteorology was made to assess the short-term effects of PM in each city. Significant or nearly significant adverse effects due to PM_{10} (fixed-term estimates) for several cardiovascular admission outcomes were observed: hospital admissions for cardiac problems in all ages (RR = 1.005 (95% CI: 1.002–1.008)) and for people older than 65 years of age (RR = 1.007 (95% CI: 1.004–1.01)); admissions for ischaemic heart disease in people younger than 65 years of age (RR = 1.003 (95% CI: 0.999–1.006)) and people older than 65 years of age (RR = 1.006 (95% CI: 1.003–1.008)). Non-significant excesses were reported for stroke admissions of people older than 65 years of age. However, compared with PM_{10}, black smoke was associated more robustly with hospital admissions for CVD when a control for co-pollutants was performed. For this reason, the authors of the APHEA-2 study (Le Tertre et al., 2002a) suggest that PM from the combustion of diesel fuel may be particularly important. The estimated effect of PM_{10} on hospital admissions did not seem to be strongly confounded by ozone or sulfur dioxide, although it was reduced (by carbon monoxide) and eliminated (by nitrogen dioxide) when other traffic-related pollutants were incorporated in the regression model. Reanalysis of the APHEA-2 study (Le Tertre et al., 2003), using both GAM with stringent convergence criteria and GLM with either natural or penalized splines, did not find noticeable changes from the original results.

Meta-analytic risk estimates for cardiac hospital admissions were provided by the MISA-2 study (Biggeri, Bellini & Terracini, 2004), and an RR of 1.003 (95% CI: 1.000–1.006) per 10-$\mu g/m^3$ increase in daily PM_{10} concentration was reported. A higher risk (1.005) was estimated for men and for the age group of people 65–74 years old. Risk estimates were smaller when adjusted for nitrogen dioxide. Non-significant estimates were reported for cerebrovascular-related hospital admissions. Estimates were also available for individual cities and age groups. A recent multi-city case-crossover study (Zanobetti & Schwartz, 2005) of residents of 21 United States cities examined the risk of emergency hospitalization associated with PM_{10} for about 300 000 elderly subjects with a primary diagnosis of myocardial infarction, between

1986 and 1999. To control for possible residual confounding by weather conditions, time–stratified controls matched by day of the week or by temperature were used. Overall, an increased risk of hospitalization of 0.65% (95% CI: 0.3–1.0%) for myocardial infarction was observed per 10-μg/m^3 increase in ambient concentration (an equivalent RR of 1.0065 (95% CI: 1.003–1.010)).

A pooled analysis of 12 hospital admission studies (Morris, 2001) showed significant increases in admission rates of 0.8% and 0.7% (RRs of 1.007 and 1.008) for heart failure and ischaemic heart disease, respectively, per 10-μg/m^3 increase in PM_{10} concentration. However, the adjustment for co-pollutants consistently reduced the effects of PM_{10}, with reductions ranging from 10% to 320% across studies.

The issue of potential confounding by gaseous co-pollutants of the association between CVD-related hospital admissions and exposure to PM is discussed in the EPA review of standards for particulate matter (EPA, 2005a). In particular, the addition of gaseous co-pollutants is considered to have little influence on most of the observed associations with PM, although a substantial reduction in associations with PM could be seen in some cases where gaseous pollutants were added to the model. In other studies, the effects of PM were substantially reduced when controlled for co-pollutants; but this effect is believed to be due, in part, to collinearity between PM indices and co-pollutants, which have independent effects on cardiovascular function. The EPA review concludes that, although the role of CVD-related hospitalization is considered likely to be causally related to exposure to PM, the extent to which PM affects this outcome, independent of (or together with) other co-pollutants (such as carbon monoxide), is uncertain.

To date, data from time-series or individual based studies are considered insufficient to provide clear information about which PM components are specifically associated with CVD-related effects. When multiple metrics are available for epidemiological analyses, they are often highly correlated or are inadequate, because the number of monitoring sites or the monitoring frequency (or both) is different for the different PM fractions.

The consistency of results observed in multi-city time-series studies is reinforced by results obtained in individual based studies of physiological measures of cardiac function or biochemical measures in blood (or both) in relation to PM pollution. In particular, although the findings of some of these studies are conflicting, the overall evidence can be considered to suggest possible mechanisms that underlie PM-related cardiovascular effects. For instance, there is a hypothesis about the association between ambient PM indices and increased blood viscosity, increased serum C-reactive protein and increased blood fibrinogen – all biological markers related to increased risks of serious cardiac events (EPA, 2005a).

The association between daily or hourly changes in PM and the incidence of myocardial infarction provide another example of a possible link between the findings of epidemiological and toxicological studies of cardiovascular effects due to PM. Experimental evidence has revealed plausible biological mechanisms through which PM has the potential to cause and exacerbate CVD. A recent paper on air pollution and CVD (Brook et al., 2004) describes possible pathways. One of these pathways involves the initiation of pulmonary and systemic oxidative stress and inflammation by components of PM. A subsequent cascade of physiological responses that are able of activate cardiovascular events may follow. These possible events, also summarized by the EPA (2005a), include alterations in blood that favour thrombosis, cardiac dysrhythmias, acute vascular dysfunction, plaque instability and the long-term development of atherosclerosis. Additional pathways are also suggested, involving changes in autonomic balance via lung neural reflex arcs or by PM (or certain components) reaching the circulation and beyond, or both.

These direct effects of air pollution may represent a plausible explanation for the occurrence of rapid (within a few hours) cardiovascular responses, such as an increasing number of myocardial infarctions, while less acute (from several hours to days) and chronic indirect effects may occur via pulmonary oxidative stress/inflammation induced by inhaled pollutants (Brook et al., 2004). At present, however, it is unclear which PM components are responsible for mediating these effects and what roles are played by the various gaseous co-pollutants. Direct effects may occur via agents that readily cross the pulmonary epithelium into the circulatory system, agents such as gases and possibly ultrafine particles and soluble constituents of $PM_{2.5}$ (such as transition metals).

The epidemiological evidence gives conflicting results on whether an increased risk of myocardial infarction is related to increases in exposure to PM immediately before the adverse event. In a large study conducted in Boston, based on 772 patients with acute myocardial infarction, Peters and colleagues (2001) linked (2- and 24-hour) ambient $PM_{2.5}$ and PM_{10} concentrations with the increased risk of myocardial infarction in the subsequent two hours and one day. A more recent study, not available at the last EPA review, was conducted in Augsburg, southern Germany (Peters et al., 2005); it included 851 subjects with non-fatal myocardial infarction. In this study by Peters and colleagues, the hypothesis that levels of ultrafine or fine particles up to two hours before the event would be associated with the induction of myocardial infarction was not confirmed. The two studies, however, reported similar magnitudes of increased RR of myocardial infarction per unit increase in $PM_{2.5}$ levels 24–48 hours before the onset of myocardial infarction. This finding suggests that exposure to $PM_{2.5}$ may play a role in the acute induction of myocardial infarction, although the differences in the estimate of the effect between the two studies need to be resolved by additional studies.

Recently, Sullivan et al. (2005) reported the results of a case–crossover study of the onset of myocardial infarction in over 3000 patients living in the Seattle area. The analytical approach taken was to use a case and control selection strategy similar to the one used by Peters and colleagues (2005). A small but non-significant increase in the onset of myocardial infarction was associated with levels of $PM_{2.5}$ at either 2 or 24 hours before onset of the event, in individuals with or without pre-existing CVD.

In a recently published European multi-centre cohort study (von Klot et al., 2005), cardiac-related hospital readmissions increased significantly in association with the same day level of PM_{10} (RR = 1.021 (95% CI: 1.005–1.039)) and the level of estimated particle number concentrations (RR = 1.026 (95% CI: 1.005–1.048)). Two studies conducted in Seattle, using a case–crossover approach, did not find an association between high levels of PM_{10} and events of primary cardiac arrest that occurred outside of the hospital in healthy adults (Sullivan et al., 2003) or in subjects already affected by underlying heart disease (Levy D et al., 2001). A possible explanation for the lack of consistency between results of studies conducted with the same design could be differences in the characteristics of the pollution mix among different study areas.

The EPA (2005a) concluded that, although the new epidemiological findings for physiological changes suggest links to mechanistic pathways that could result in observed cardiovascular morbidity or mortality, there are limitations to be considered in the interpretation of these studies. While many research questions remain open, the convergence of evidence from epidemiological and toxicological studies that relates to cardiac health indicates both coherence and plausibility in this body of evidence.

2.10.1.2 Risk estimates used for the present study

In light of the current evidence, cardiac-related hospital admissions (ICD IX 390–429) were included as an outcome for impact assessment. Also, risk estimates provided by the MISA-2 study were used in the present report. Since hospital admission rates are sensitive to large random fluctuations, local factors and differences in the criteria used in data registration, meta-analytical national risk coefficients were preferred to both international meta-analytical risk coefficients (APHEA 2) and city-specific risk estimates.

An RR of 1.003 (95% CI: 1.000–1.006) per 10-µg/m^3 increase in daily PM_{10} concentration was used in the present study.

2.10.2 RESPIRATORY-RELATED HOSPITAL ADMISSIONS

2.10.2.1 Scientific evidence

The most recent EPA (2005a) review of air quality standards for PM, in summarizing salient findings on acute PM exposure and respiratory-related hospital admissions and medical visits during the last decade, concluded that the results of new studies "are generally consistent with and supportive of findings presented in the 1996 PM AQCD (EPA, 1996), with regard to ambient PM associations of short-term exposures with respiratory-related hospital admissions/medical visits".

In summary, the EPA concluded that, in considering results from studies conducted both within and outside Canada and the United States, there is a pattern of positive and often statistically significant associations across studies between respiratory health outcomes – including mortality and hospitalization and medical visits for respiratory diseases – and PM_{10} and $PM_{2.5}$.

With regard to potential confounding by co-pollutants, the EPA reported that the estimates of the effects of PM on respiratory-related mortality and morbidity were little changed in multi-pollutant models, as compared with single-pollutant models. Therefore, estimates of adverse effects from single-pollutant models can be used to represent the magnitude of a concentration–response relationship, though some uncertainty remains with regard to the potential effect from other pollutants.

The results of 17 single-city studies and one multi-city study from Canada and the United States were analysed in the EPA review. These 18 studies were selected for comparison because they were all characterized by: (a) having provided single-pollutant model (PM only) results and (b) having considered major categories of respiratory-related morbidity outcomes commonly used in PM time-series studies (such as hospitalization and medical visits). Moreover, all studies used one or more of the three major PM mass indicators (PM_{10}, $PM_{2.5}$, or $PM_{10–2.5}$) and either did not use GAM or were reanalysed with corrected GAM models.

Results of this EPA analysis showed that all associations between PM_{10} and hospitalization for respiratory diseases were positive and that most were statistically significant. Almost all PM_{10} associations with visits to emergency departments for respiratory diseases were positive, and most were statistically significant. The most precise estimates of adverse effects ranged from about 2% to 12% increased risk per 50-µg/m^3 increase in PM_{10} concentration (an equivalent RR ranking from 1.004 to 1.023 for a 10-µg/m^3 increase in PM_{10} concentration) for respiratory diseases, with some estimates of adverse effects for respiratory medical visits up to about 30% per 50-µg/m^3 increase in PM_{10} concentration (RR = 1.054). Similar results were

produced for hospitalization and medical visits for CVD. The EPA has emphasized that results of multi-city studies reflect more accurately the magnitude of the associations between PM and health, and their precision is greater than that for single-city studies. In particular, the NMMAPS 14-city analysis reported a statistically significant increased risk of about 6% and 8% per 50-μg/m^3 increase in PM_{10} concentration for, respectively, pneumonia- and COPD-related hospital admissions for the elderly (RRs, equivalent to, respectively, 1.012 and 1.016 per 10-μg/m^3 increase in PM_{10} concentration) (HEI, 2000a, b; Zanobetti et al., 2003). The authors of these studies a lack of confounding by co-pollutants. Two other important multi-city studies, not considered in the EPA review, are the APHEIS-3 European project (with a risk estimate provided in the appendix of the report) (Medina et al., 2005) and the MISA-2 study (Biggeri, Bellini & Terracini, 2004), which provide further evidence of the association between exposure to PM_{10} and respiratory-related hospital admissions.

The APHEIS-3 project analysed daily counts of hospital respiratory admissions for nine European cities (of the original 26 under study) with a combined population of about 25 million people. City-specific risk estimates were made for respiratory-related hospital admissions (across all ages), in relation to exposure to PM_{10}. The single-pollutant model summary estimates (fixed and random) were, respectively, 1.010 (95% CI: 1.008–1.013) and 1.011 (95% CI: 1.006–1.017) for a 10-μg/m^3 increase in PM_{10} concentration, and they were robust to the inclusion of ozone in the model. A city-specific risk estimate is also available for Rome.

The MISA-2 study provided city-specific and meta-analytic risk estimates for respiratory-related hospital admissions. It reported an overall RR of 1.006 (95% CI: 1.002–1.011) per 10-μg/m^3 increase in PM_{10} concentration for 15 cities. No differences were observed by sex. Risk estimates for PM_{10} were smaller when adjusted for nitrogen dioxide and higher when controlling for ozone. City-specific risk estimates were also provided.

In the WHO Regional Office for Europe meta-analysis of time-series and panel studies of PM and ozone (Anderson et al., 2004), the summary estimate of RR for respiratory admissions in the age group of people older than 65 years for a 10-μg/m^3 increase in PM_{10} concentration was 1.007 (95% CI: 1.002–1.013) and was based on eight estimates. Six of these eight estimates were provided by the APHEA-2 project (Atkinson et al., 2001). The evidence available from epidemiological studies on the association between ambient PM concentrations and increased respiratory-related hospital admissions and emergency department and other medical visits is reinforced by results that show several different effects, such as exacerbation of asthma, increased incidence of other respiratory symptoms and decrements in pulmonary function. Moreover, new findings are emerging, indicating increased occurrence of chronic bronchitis in association with exposure to PM. The biological mechanisms underlying such effects may involve inflammatory responses, increased airway reactions or altered responses to infectious agents (EPA, 2005a). Results from the main studies on specific respiratory effects of exposure to PM are described in Subsections 2.10.3–2.10.8.

2.10.2.2 Risk estimates used for the present study

Baseline data on respiratory hospital admissions were derived from the recently published average annual number of hospital admissions for each selected city (Biggeri, Bellini & Terracini, 2004). All respiratory diseases were included (ICD IX 460–519), while scheduled admissions were excluded. The MISA-2 meta-analytical risk estimates were chosen.

An RR of 1.006 (95% CI: 1.002–1.011) per 10-μg/m^3 increase in daily PM_{10} concentration was used in the present study.

2.10.3 CHRONIC BRONCHITIS IN ADULTS

2.10.3.1 Scientific evidence

The evidence from epidemiological studies, conducted before the 1996 AQCD on PM (EPA, 1996), on the association between respiratory disease and long-term exposure to PM was considered limited. More recent studies on respiratory morbidity, included in the last review by the EPA (2005a), report positive and statistically significant associations between fine particles (or fine particle components) and decreased lung function or chronic respiratory diseases, such as chronic bronchitis. Among several long-term studies of respiratory effects from non-North-American countries, many report significant associations between indices on long-term exposure to PM and either decreases in lung function or increased prevalence of respiratory disease.

Chronic bronchitis is the most prevalent COPD-related illness and is characterized by pathological airway inflammation and epithelial damage, mucus cell hyperplasia and hypersecretion, airway obstruction and, in advanced cases, airway fibrosis. The EPA (2005a), in its review, emphasized that people with chronic bronchitis (or asthma or acute lung infections) are likely to have increased deposition and retention of inhaled particles and to be at increased risk for adverse effects from exposure to ambient PM. For these reasons, such individuals may plausibly be expected to be at even greater risk when inhaling ambient PM under conditions of high humidity (with increased delivery of peroxides, sulfur dioxide and other noxious agents into the deep lung). It has to be stressed, however, that relatively few studies have been carried out on the association between air pollution and the development of chronic bronchitis.

A study conducted in Tokyo (Ye et al., 2001) examined records of hospital emergency transports for cardiovascular and respiratory diseases in adults (older than 65 years of age) during the months of July and August, from 1980 to 1995. Both PM_{10} and nitrogen dioxide levels were associated significantly with daily hospital admissions for angina, cardiac insufficiency, myocardial infarction, acute and chronic bronchitis and pneumonia. Except for pneumonia, daily maximum temperatures were not associated with hospital emergency transports. The severity of the effects of pollutants was generally found to be greater in men than in women, except those for angina and acute bronchitis, which were the same across sexes.

The Seventh Day Adventist study is considered the only major study to date to quantify the effects of PM on the increase of new cases of chronic bronchitis in adults (Hurley et al., 2005). This is a cohort study of 3914 adults 27 years of age or older at enrolment in 1977 (Abbey et al., 1993, 1995a, 1995b). Chronic bronchitis in individuals was defined as their having chronic cough or sputum, on most days, for at least three months per year, for at least two years. New cases of chronic bronchitis were defined as those that met the criteria in 1987/1988, but not in 1977, whereas cases of remission were those that met the criteria in 1977, but not 10 years later. The number of new cases of chronic bronchitis over a 10-year period was analysed in relation to air pollution, including estimates of lifetime exposure to PM_{10}, derived from data on TSPs. Results were adjusted for such covariates as age, sex and education and for respiratory symptoms in 1977. The extremely low prevalence of smoking and the relatively healthy dietary patterns among the study subjects almost eliminate the potential for confounding by these factors. An RR of 1.15 for development of airway obstructive disease, for an increase of 20 $\mu g/m^3$ in PM_{10} concentrations (equivalent to an RR of 1.07 per 10 $\mu g/m^3$), was observed.

2.10.3.2 Risk estimates used for the present study

An RR of 1.15 for an increase of 20 µg/m^3 in PM_{10} concentrations (equivalent to 1.07 per 10 µg/m^3) and background rates (234 new cases of chronic bronchitis in 1987 (compared with 1977) out of the 3310 not classified as having bronchitis at the start of the period – an attack rate of 0.707%) was taken from two studies by Abbey and colleagues (1993, 1995a). The remission rate was estimated to be 46.6% (Abt Associates, 2000). The net incidence rate is equal to 0.00707*(1/0.466) = 0.378% or 3.78 new annual cases per 1000 adults at risk. As in the study by Hurley and colleagues (2005), an impact function was applied to the population older than 27 years of age who did not have chronic bronchitis.

The number of new cases of chronic bronchitis per year per 100 000 adults older than 27 years of age is 26.5 (95% CI: -1.9–54.1) per 10-µg/m^3 increase in PM_{10} concentration.

It must be stressed that problems of transferability of this impact function are noteworthy: the estimates have been derived from a single subnational study, referring to California only and to a population (the Seventh Day Adventists) with a lifestyle that may be quite different from those found in other parts of the United States. Also, it must be stressed that the risk estimate applied is not statistically significant, as reflected by the width of the confidence interval. It has been decided to use this kind of estimate in the present report, as anticipated earlier, in line with the approach followed by Ostro and by the CAFE programme group in their cost–benefit quantification of adverse effects on health.

2.10.4 ACUTE BRONCHITIS IN CHILDREN

2.10.4.1 Scientific evidence

Several studies indicate an association between annual exposure to PM and the likelihood of bronchitis in children. For example, Dockery and colleagues (1989, 1996) analysed data from 6 and 24 United States cities, respectively. Each child, prior to exposure to PM, was matched with a survey questionnaire that asked parents whether their children of 8–12 years of age had had bronchitis during the preceding 12 months. The first study used PM_{15} data for the years 1980 and 1981 and found an RR of 1.6 (95% CI: 1.1–2.5) per 20-µg/m^3 increase in PM_{10} concentration; the second study used PM_{10} data for 1988–1991 and found an RR of 1.60 (95% CI: 0.92–2.78).

In the morbidity studies of the cohort of Southern California children, which included 3676 subjects, no associations were reported for all children (Peters et al., 1999). However, restricting the analysis to children with asthma (McConnell et al., 1999), a statistically significant increased risk for bronchitis was observed in relation to an increase of 20 µg/m^3 in PM_{10} concentration (RR = 1.4 (95% CI: 1.1–1.8)). McConnell and colleagues noted that, as PM_{10} increased across communities, the risk of bronchitis per interquartile range also increased. These results are consistent with those reported by Dockery and colleagues (1996). However, in the study by McConnell and colleagues, the high correlation of PM_{10}, acid and nitrogen dioxide precludes clear attribution of the findings on the effects of bronchitis specifically to PM alone.

In the first cross-sectional assessment of the Swiss Surveillance Program of Childhood Allergy and Respiratory Symptoms with Respect to Air Pollution and Climate (SCARPOL) in 1992/1993 (Braun-Fahrlander et al., 1997), rates of respiratory symptoms and diseases, adjusted for individual risk factors, were positively associated with PM_{10}, nitrogen dioxide

and sulfur dioxide in children living in 10 urban, suburban, rural and alpine areas of Switzerland. Among children 6–15 years old, PM_{10} was significantly associated with bronchitis. These results were confirmed in a recent update (Bayer-Oglesby et al., 2005) of the study, which included 9591 children from nine Swiss communities who participated in cross-sectional health assessments between 1992 and 2001. In particular, declining PM_{10} concentrations were associated with a declining prevalence of bronchitis (odds ratio (OR) = 0.66 (95% CI: 0.55–0.80)), after adjustment for socioeconomic, health-related and indoor factors.

2.10.4.2 Risk estimates used for the present study

The joint estimate from the three above-mentioned studies (Dockery et al., 1989, 1996; Braun-Fahrlander et al., 1997), as in the first WHO Regional Office for Europe report (Martuzzi et al., 2002), derived by Künzli (1999), was applied to the present study. The same background rates were used also, derived from the first phase of the SIDRIA study: a 10.6% prevalence of doctor-diagnosed bronchitis in the last year in a sample of 10 147 children (SIDRIA, unpublished data) was applied.

An RR of 1.306 (95% CI: 1.135–1.502) per 10-μg/m^3 increase in PM_{10} concentration was used in the present study.

2.10.5 ASTHMA IN CHILDREN

2.10.5.1 Scientific evidence

In a recent WHO Regional Office for Europe monograph on atmospheric pollution and children's health (WHO Regional Office for Europe, 2005a), it was stated that "there is substantial literature on the health effects of air pollution on children in general, and on children within certain subgroups of susceptibility, particularly those with asthma".

In general, the evidence that children are adversely affected by air pollution is considered conclusive, but some distinctions must be made for different health end-points. In particular, the available evidence is judged sufficient to establish a causal relationship between exposure to air pollution and aggravation of asthma (mainly due to exposure to PM and ozone). In contrast, the evidence for a causal relationship between the occurrence of new cases of asthma and air pollution is not conclusive. However, data suggest a causal relationship between the prevalence/incidence of asthma symptoms and living in close proximity to traffic (WHO Regional Office for Europe, 2005a).

Several specific characteristics – such as the active processes of lung growth and development, incomplete metabolic systems, immature host defenses, high levels of activity and ventilation that enhance exposure (and lung doses) to air pollution, and the high rates of infection due to respiratory pathogens – are all considered potential determinants of susceptibility of children to inhaled pollutants (Schwartz, 2004; EPA, 2005a; WHO Regional Office for Europe, 2005a).

In addition to that, the WHO Regional Office for Europe has suggested that the large proportion of children with principal chronic lung disease (with particular emphasis on asthma) may be at greater risk than children without this condition. Moreover, childhood asthma is a heterogeneous clinical condition, and some evidence indicates that sensitivity to environmental agents may also vary among asthmatic children. Another consideration is the

patterns of exposure to indoor pollutants: children with higher indoor exposures – for instance, to environmental cigarette smoke – may be at greater risk of being affected by outdoor pollutants.

Most of the population-level time-series studies on hospital admissions, emergency department visits and calls to doctors for asthma in children, published during the period 1990–2003, have been reviewed by the WHO Regional Office for Europe, (2005a). In at least eight studies, including the APHEA-2 study, results show a statistically significant adverse effect of PM on health.

A recent reanalysis of the APHEA-2 study (Atkinson & APHEA 2 Project, 2004), which evaluated the association between PM and hospital admissions for respiratory symptoms in eight European cities, reported an increase of 1.5% in asthma admissions in children per 10-$\mu g/m^3$ increase in PM_{10} concentration.

The WHO Regional Office for Europe reviewed panel time-series studies that considered asthmatic children at the individual-level. A total of 21 studies were available for PM_{10} (10 in Europe and 11 elsewhere), and the majority of them found a positive association between asthma and PM. The number of reviews of studies on the use of medications was more limited, but the overall picture was considered very similar to that reported on the aggravation of symptoms.

The concluding remarks of the WHO Regional Office for Europe review (2005a) stated that, although the relative risk estimates for the health outcomes in children are generally small, the risk attributable to air pollution among European children is high, due to the widespread nature of atmospheric pollutants and the relatively high incidence of many relevant effects.

The EPA (2004, 2005a) also recently reviewed numerous epidemiological studies, and most of them reported associations between PM and emergency hospital admissions for respiratory illness and asthma-related symptoms in children. Results from panel studies for asthma symptoms in groups of schoolchildren showed that children are susceptible to PM-related effects. These effects were studied by monitoring subjects over generally short periods of time in relation to changes in ambient PM_{10}, and $PM_{10–2.5}$ or $PM_{2.5}$ (or all three), or other airborne particles (ultrafine PM, TSPs, black smoke and the sulfate fraction of ambient PM). The respiratory symptoms considered in these studies are cough, phlegm, difficulty breathing, wheeze and use of a bronchodilator.

As in the WHO Regional Office for Europe monograph (2005a), EPA documents (EPA, 2004, 2005a) stated that combining the small risk estimates and small changes in PM concentration observed with large groups of United States populations, such as young children that are considered likely to be susceptible to the effects of exposure to PM, results in large public health impacts. In fact, in 2000, about 11% of the children in the United States were diagnosed as having asthma and about 26% of the United States population was under 18 years of age. Therefore, even a small percentage reduction in the effects of PM on cardiorespiratory disease would result in a large number of cases avoided.

In the previous WHO Regional Office for Europe report (Martuzzi et al., 2002), the health impact of PM_{10} on asthma was measured by using the exacerbation of asthma attacks in children and adults as the measure. Use of medication – specifically bronchodilator usage – was examined more recently by the WHO Regional Office for Europe meta-analysis of acute effects found in panel studies (Anderson et al., 2004). These studies are based on the hypothesis that daily variations in the level of air pollution will cause a supplementary use of medication to control asthma attacks, because well-treated asthmatics are able to manage their

attacks through an increased use of medication. For this reason, both in children and in adults, bronchodilator usage is thought to be a more reliable measure of the frequency of asthma, rather than the occurrence of self-reported asthma symptoms or attacks (Just et al., 2002).

In relation to exposure to PM_{10}, bronchodilator use or use of ß agonists in symptomatic children has been analysed by Anderson and colleagues (2004) in a WHO Regional Office for Europe meta-analysis based on 17 studies, covering 31 different locations, including 27 estimates from the Pollution Effects on Asthmatic Children in Europe (PEACE) studies. The meta-analysis provided a pooled estimate of risk, widely representative of the European cities, of 1.005 (95% CI: 0.981–1.029) per 10-µg/m^3 increase in PM_{10} concentration.

Results from the SIDRIA-2 project, on the prevalence of respiratory and allergic disorders and their geographical and temporal variations, carried out in the context of the International Study of Asthma and Allergies in Childhood (ISAAC) study, were also published recently. The SIDRIA-2 study was designed to assess the prevalence of a multiplicity of environmental, social, behavioural and familiar risk factors and to evaluate the relationship between these factors and several respiratory disorders in children. The study, carried out in 2002, covered 13 Italian areas and included 20 016 children (6–7 years old) and 16 175 adolescents (13–14 years old), with a response rate of 89% and 93%, respectively (Galassi, De Sario & Forastiere, 2005). As stated by the childrens' parents, during the 12 months preceding the study, asthma with symptoms, cough or phlegm were reported in 13.2% of children 6-7 years of age and in 9.9% of adolescents 13-14 years of age.

2.10.5.2 Risk estimates used for the present study

A pooled estimate of the OR of 1.005 (95% CI: 0.981–1.029) per 10-µg/m^3 increase in PM_{10} concentration, not statistically significant and approximately equal to a percentage increase of 0.5%, was derived by the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004). The prevalence rates for asthma were derived from the SIDRIA-2 study (Galassi et al., 2005) for selected age groups (6–7 and 13—14 years of age). City-specific rates were available for Turin, Milan and Rome; the Florence-Prato rate was applied to Florence, and the Emilia-Romagna rate to Bologna. The prevalence rates for the remaining cities were assumed equal to the SIDRIA-2 overall rate (13.2% for children 6–7 years of age and 9.9% for children 13–14 years of age) (Forastiere et al., 2005). An estimate of 10% mean daily prevalence of bronchodilator usage among schoolchildren was chosen, as suggested by Hurley and colleagues (2005).

The present report uses the following annual increase of bronchodilator usage in children 6–7 years of age and 13–14 years of age: 180 (95 % CI: -690–1060) annual increase in days of bronchodilator usage per 1000 children, per 10-µg/m^3 increase in PM_{10} concentration.

2.10.6 ASTHMA IN ADULTS

2.10.6.1 Scientific evidence

As indicated in the EPA review (2004) of panel studies on asthma symptoms, both the elderly and children are susceptible subpopulations for PM-related effects. In addition, recent epidemiological studies have shown associations between increased non-hospital medical visits (physician visits) and effects on asthma. These findings suggest large additional effects on health and costs to society due to ambient PM, besides those due to hospital

admissions/visits or mortality, or both (EPA, 2005a). For respiratory conditions, about 9% of United States adults have been diagnosed as having asthma (11% among children); even small risks and small changes in PM concentrations would therefore result in large public health impacts, in children and adults (EPA, 2004).

Among United States studies that found associations between respiratory-related hospital admissions (or medical visits) and PM_{10}, the excess risk estimates most consistently fall in the range of 5–25% per 50-µg/m^3 increment in PM_{10} concentration (that is, from 1% to 5% per 10-µg/m^3 increment in PM_{10} concentration); the risks of asthma visits and hospital admissions are generally higher than those of COPD and pneumonia admissions.

Studies published after the 1996 AQCD on PM (EPA, 1996) have examined various admission categories, including total respiratory admissions and asthma admissions, for all ages, and by age for COPD admissions (usually for patients older than 64 years of age). Overall, PM_{10} and $PM_{2.5}$ both appear to affect lung function in asthmatics, but there is only limited evidence for a stronger effect of fine versus coarse fraction particles; and ultrafine particles do not appear to have any noticeably stronger effect than other larger-diameter fine particles. The effects of PM_{10} on respiratory symptoms in asthmatics tended to be positive, although they were somewhat less consistent than the effects of PM_{10} on lung function.

As observed in children, the quantification of bronchodilator usage is considered to be preferred to asthma symptoms or attacks in estimating the relationship between air pollution and asthma in adults (Just et al., 2002). A pooled OR estimate of 1.010 (95% CI: 0.990–1.031)) per 10-µg/m^3 increase in PM_{10} concentration, relating medication use (bronchodilator use or specific use of ß agonists) in symptomatic adults to exposure to PM_{10}, was calculated by Anderson et al. (2004), based on three studies that comprised 138, 128 and 32 subjects followed for about three months.

2.10.6.2 Risk estimates used for the present study

An OR has been derived by the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004) that provides a result, per 10-µg/m^3 PM_{10} increment, that is not statistically significant (1.010 (95% CI: 0.990–1.031)). The mean daily prevalence of the use of bronchodilators by people with asthma (0.5) and the background rates (4.5) – that is, the percentage of adults with asthma, of a severity comparable to that of the panel studies analysed in the meta-analysis – have been derived, as in the study by Hurley and colleagues (2005). Combining these three elements, an impact function has been derived.

The annual increase in days of bronchodilator usage per 1000 adults 15 years of age and older is 912 (95% CI: -912–2774) per 10-µg/m^3 PM_{10} increment.

2.10.7 RESTRICTION IN ACTIVITY

An RAD is defined as a day when a person is forced to alter his/her normal activity, for health-related reasons. This type of day is classified according to degrees of severity in three mutually exclusive categories, as follows (Portney & Mullahy, 1986):

- bed disability days: days when a person needs to stay in bed;
- work or school loss days (WLDs): days when a person stays away from work or school, but does not need to stay in bed; and

- minor restricted activity days (MRADs): these do not involve work loss or bed disability, but include less serious restrictions on normal activity.

As stated in the report by Hurley and colleagues (2005), these categories of health end-points are culturally specific, and the uncertainty of transferring rates from place to place will be large.

2.10.7.1 Scientific evidence

Most of the air pollution health impact assessment studies that provide estimates of days in which normal activities are restricted were included in the present study. Several morbidity health outcomes (such as RADs, MRADs and WLDs) were studied using concentration–response functions estimated in two United States studies (Ostro, 1987; Ostro & Rothschild, 1989).

In the first study, Ostro (1987) studied RADs among adults 18–64 years of age in separate analyses for each of the six years 1976–1981 the Health Interview Study (HIS) was carried out annually by the United States National Center for Health Statistics (NCHS). The HIS, a large cross-sectional database, is a multi-stage probability sample of 50 000 households from United States metropolitan areas and regions. The large majority (85–95%) of subjects reported no RADs. Results for RADs, based on about 12 000 subjects per year from 68 metropolitan areas, showed a consistent relationship with $PM_{2.5}$, estimated from airport visibility data. Confounders such as race, sex, temperature during the two relevant weeks, education and income were included. The concentration–response coefficients were all positive and highly significant statistically ($p<0.01$).

In the second study, Ostro & Rothschild (1989) considered the same six years of the HIS (1976–1981) and focused on MRADs and respiratory RADs. Only current workers, residents in urban areas, were included in the study. As in the previous study, analyses of the data were performed separately, year by year. The $PM_{2.5}$ data were the same as in the first study (Ostro, 1987). The relationship between $PM_{2.5}$ and respiratory RADs was clear and consistent, with regression coefficients positive, consistent and statistically significant for all six years.

More recently, Stieb and colleagues (2002) reported findings on air pollution and disability days from Canada's National Population Health Survey (NPHS). The design of this study is similar to the United States HIS, but was carried out every two years. Stieb and colleagues studied data from three periods (1994/1995, 1996/1997 and 1998/1999) of the Canadian NPHS, with particular attention on Toronto. Disability days were defined as days spent in bed or days when the respondent decreased usual activities, during the two weeks prior to interview. This definition was similar to that used by Ostro (1987). Based on 5309 interviews, the average number of disability days in the previous 14 days was 0.73, entailing on average 19 disability days per subject per year (the same baseline incidence rate as Ostro (1987) for RADs). Two-week averages of daily pollution concentrations were available for PM_{10}, $PM_{2.5}$, coarse particles, nitrogen dioxide, sulfur dioxide and carbon monoxide.

In the last EPA review on PM (EPA, 2005a), epidemiological evidence indicated that exposure to PM was associated with an increased risk for various cardiopulmonary effects, including school absences, WLDs and RADs. For chronic respiratory health diseases, it was estimated that 700 million RADs per year are due to respiratory conditions (Adams, Hendershot & Marano, 1999).

As to WLDs, in the Ostro study (1987), a unique sample of about 7000 employed adults (15–64 years of age) was available for each of the six years 1976–1981 of the HIS. The year-by-year coefficients estimated for the relationship between WLDs and $PM_{2.5}$ were more variable than those between RADs and $PM_{2.5}$. Nevertheless, four of the six coefficients were positive and statistically significant (one was negative and statistically significant), and for three of the years the estimate was practically the same. As to the number of WLDs per subject in employment, Bergendorff (2003) compared rates of absence due to sickness in eight European countries (Denmark, Finland, France, Germany, the Netherlands, Norway, Sweden and the United Kingdom), based on data provided by the appropriate Labour Force Surveys (LFS) in each country. The LFS obtained data by asking each participant about absences from work for the whole reference week considered by the survey. A limitation of this study is therefore the failure in accounting for absences shorter than one week or for absences spread across consecutive weeks. The study showed a variation in the number of WLDs per subject (crude absence rates), from 1.4% in Germany to 4.2% in Sweden, with an eight-country average of 2.1%. Assuming 228 working days per year, this implies an average of 4.8 WLDs per person per year, attributable to absences of at least one-week duration. Bliksvaer & Helliesen (1997) reported figures that were broadly similar.

2.10.7.2 Risk estimates used for the present study: RADs

An estimated OR of 0.475% (95% CI: 0.417–0.533%) per 10-µg/m^3 increment in $PM_{2.5}$ concentration was positive and highly significant statistically ($p<0.01$), and a background rate of 19 RADs per person per year, equivalent to a prevalence of 5.2%, was derived from the study by Ostro (1987).

As in the study by Hurley and colleagues (2005), linking this background rate with the percentage increase of 0.475% per 10-µg/m^3 increment in $PM_{2.5}$ concentration provides the impact function used in the present report.

The increase in RADs per 1000 adults 15–64 years of age per year is 902 (95% CI: 792–1013) per 10-µg/m^3 increment in $PM_{2.5}$ concentration.

2.10.7.3 Risk estimates used for the present study: WLDs

An estimated OR of 0.46% (95% CI: 0.39–0.53%) per 10-µg/m^3 increment in $PM_{2.5}$ concentration was derived from the study by Ostro (1987).

The background rate was divided in two components.

1. The first component was based on the average WLDs per individual (15–64 years of age) in employment.

2. The second component, based on an Italian city-specific employment rate, is defined as employed people 15–64 years of age per total population of the same age.

The first component was derived from the Bergendorff study (2003) on sickness absence in Europe. The study provided an average of 4.8 WLDs (per person per year) attributable to absences of at least one-week duration. Because of shorter duration absences, that average

was multiplied by 1.5, to provide an estimate of 7.5 total annual WLDs (per person), as summarized by Hurley and colleagues (2005).

The second component, because of the availability of more detailed socioeconomic data (employment rates (ISTAT, 2005)), has been calculated for each province of residence, and the employment rates obtained were extrapolated.

Therefore, combining the city-specific employment rates with the average number of WLDs (7.2), the following estimates of background rates were obtained:

- Turin: 4.3 WLDs per year per person 15–64 years of age (general population);
- Genoa: 4.1 WLDs per year per person 15–64 years of age (general population);
- Milan: 4.5 WLDs per year per person 15–64 years of age (general population);
- Padua: 4.4 WLDs per year per person 15–64 years of age (general population);
- Verona: 4.5 WLDs per year per person 15–64 years of age (general population);
- Venice-Mestre: 4.4 WLDs per year per person 15–64 years of age (general population);
- Trieste: 4.3 WLDs per year per person 15–64 years of age (general population);
- Bologna: 4.8 WLDs per year per person 15–64 years of age (general population);
- Florence: 4.5 WLDs per year per person 15–64 years of age (general population);
- Rome: 3.9 WLDs per year per person 15–64 years of age (general population);
- Naples: 2.7 WLDs per year per person 15–64 years of age (general population);
- Catania: 2.9 WLDs per year per person 15–64 years of age (general population); and
- Palermo: 2.8 WLDs per year per person 15–64 years of age (general population).

City-specific impact functions were derived per 1000 people 15–64 years of age in the general population:

- Turin: 197 WLDs (95% CI: 167–227) per 10-μg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-μg/m^3 increment in $PM_{2.5}$ concentration);
- Genoa (187, 95% CI: 158–215) per 10-μg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-μg/m^3 increment in $PM_{2.5}$ concentration);
- Milan: 206 WLDs (95% CI: 175–238) per 10-μg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-μg/m^3 increment in $PM_{2.5}$ concentration);
- Padua: 202 WLDs (171–233) per 10-μg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-μg/m^3 increment in $PM_{2.5}$ concentration);

- Verona: 207 WLDs (95% CI: 175–238) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration);
- Venice-Mestre: 201 WLDs (95% CI: 170–232) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration);
- Trieste: 198 WLDs (95% CI: 168–228) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration);
- Bologna: 222 WLDs (95% CI: 188–256) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration);
- Florence: 205 WLDs (95% CI: 174–237) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration);
- Rome: 181 WLDs (95% CI: 154–209) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration);
- Naples: 126 WLDs (95% CI: 107–145) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration);
- Catania: 134 WLDs (95% CI: 114–154) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration); and
- Palermo: 127 WLDs (95% CI: 108–147) per 10-µg/m^3 increment in $PM_{2.5}$ concentration, calculated as baseline rate (4.3 WLDs per person per year) per change per 10-µg/m^3 increment in $PM_{2.5}$ concentration).

2.10.7.4 Risk estimates used for the present study: MRADs

An estimated OR of 0.74% (95% CI: 0.60–0.88%) per 10-µg/m^3 increase in $PM_{2.5}$ concentration and a baseline rate of 7.8 MRADs per year, among people of employment age (18–64 years), were derived from the report by Ostro & Rothschild (1989).

As in the report by Hurley and colleagues (2005), the impact function used in the present report was obtained by linking this background rate with the percentage increase of 0.474% per 10-µg/m^3 increase in $PM_{2.5}$ concentration.

The increase in MRADs per 1000 adults aged 18–64 years per year used in the present study is 577 MRADs (95% CI: 468–686) per 10-µg/m^3 increment in $PM_{2.5}$ concentration.

2.10.8 LRS

LRS are not defined in a consistent way across studies, but they generally include wheezing, tightness of chest, shortness of breath and cough.

Consistent with the 1996 review on PM (EPA, 1996), some significant associations between increased respiratory symptoms and decreased lung function and short-term exposures to PM are also reported in the last EPA reports (EPA, 2004, 2005a). However, although most studies showed increases in cough, phlegm, difficulty in breathing and bronchodilator use, these were generally not statistically significant for PM_{10}.

Among asthmatic subjects, associations have been described between PM_{10} and $PM_{2.5}$ and decreases in lung function measures, but not all of the relationships were statistically significant. In addition, positive associations have been reported between PM_{10} and $PM_{2.5}$ and one or more of a variety of respiratory symptoms, including, cough, wheeze and shortness of breath. The findings, however, were less consistent than those observed for lung function. In studies of non-asthmatic subjects, though inconsistent results were described for changes in lung function, there were generally positive relationships between PM_{10} and $PM_{2.5}$ and respiratory symptoms, but they were generally not statistically significant.

2.10.8.1 Scientific evidence: children

The WHO Regional Office for Europe review on the effects of air pollution on children's health (WHO Regional Office for Europe, 2005a) reported that a significant body of evidence suggests a causal relationship between exposure to ambient air pollution (including PM_{10}, $PM_{2.5}$, nitrogen dioxide, sulfur dioxide and ozone) and an increased incidence of upper and lower respiratory symptoms. In particular, it stated that many of upper and lower respiratory symptoms in children are likely to be related to infections, providing evidence for possible mechanisms of interaction of air pollutants with infections. This evaluation confirms that reducing pollutants could improve the health of children and that, although the estimates are mainly for a small effect, the risks attributable to the population would be high. Further studies are considered necessary to improve the understanding of the mechanisms of interaction (WHO Regional Office for Europe, 2005a).

Few studies of respiratory symptoms and lung function in children were found to have included data on both $PM_{2.5}$ and $PM_{10–2.5}$. The AQCD for PM (EPA, 2004) summarizes results from the Harvard Six Cities Study analysis (Schwartz & Neas, 2000), a study carried out in Philadelphia (Neas, Schwartz & Dockery, 1999) and a Finnish study (Tiittanen et al., 1999). The findings of these studies suggest a role for both $PM_{2.5}$ and $PM_{10–2.5}$ in reducing lung function and in increasing the incidence of respiratory symptoms.

In a panel study of 156 normal children who attended summer camps in the Greater Philadelphia area in 1993 (Neas, Schwartz & Dockery, 1999), subjects were followed for at most 54 days. Morning and evening deviations of each child's peak expiratory flow (PEF) were analysed, using a mixed-effects model adjusted for autocorrelations. Covariates included time trend and temperature, and negative, but non-significant results were obtained for PEF.

Tiittanen et al. (1999), in a six-week panel study of 49 children with chronic respiratory disease followed in spring 1995, showed significant effects of $PM_{2.5}$ on cough for a four-day average level of $PM_{2.5}$.

In the Six Cities Study (Schwartz & Neas, 2000), which reported on the analysis of 1844 school children who lived in Boston, St. Louis, Knoxville, Topeka, Portage and Steubenville, LRS were found to have increased significantly with $PM_{2.5}$, but not with $PM_{10–2.5}$, while the opposite results were observed for cough.

In the WHO Regional Office for Europe meta-analysis of time-series studies and panel studies of PM and ozone (Anderson et al., 2004), 34 point estimates from European panel studies were available for investigating the relationship between PM_{10} and cough in symptomatic children. Many of these estimates came from the PEACE study, a multi-city panel study conducted in 14 centres, using a common protocol. The pooled estimated OR was close to 1.0 and was non-statistically significant (OR= 0.999 (95% CI: 0.987–1.011)).

A recent review of particulate air pollution and panel studies in children (Ward & Ayres, 2004) identified 22 studies. Almost all (but two) studies reported a 24-hour mean level of PM_{10} higher than 50 $\mu g/m^3$. The ages of the children extended from 6 to 11 years. The majority of studies (15) considered panels of children either diagnosed with asthma or with reported pre-existing respiratory symptoms. In seven studies, both symptomatic and non-symptomatic subjects were enrolled. Reported effects of PM_{10} on PEF and symptoms were spread widely and were smaller than those observed for $PM_{2.5}$. The authors did not exclude the possibility of an interaction between PM and ozone. In particular, estimates of pooled random effects (larger than the fixed ones) for cough alone were 1.010 (95% CI: 1.005–1.016) per 10-$\mu g/m^3$ increment in $PM_{2.5}$ concentration and 1.004 (95% CI: 1.002–1.006) per 10-$\mu g/m^3$ increment in PM_{10} concentration. The results for lower respiratory tract symptoms (excluding cough) were 1.009 (95% CI: 1.002–1.016) for $PM_{2.5}$ and 1.004 (95% CI: 1.002–1.005) for PM_{10}. Due to considerable heterogeneity and evidence for publication bias, the authors stated that limited confidence may be placed on summary estimates of adverse effects.

The methodology paper (Vol. 2) of the CAFE programme (Hurley et al., 2005) describes in detail the reasons why, when in carrying out cost–benefit analyses for air-quality-related issues, the estimates provided by Ward & Ayres (2004) are to be preferred to the pooled estimates published in the WHO Regional Office for Europe meta-analysis. In particular, the dominance of the results from the PEACE study, which was a study of winter-time pollution, in the WHO Regional Office for Europe meta-analysisis, is likely to have reduced the potential for heterogeneity: the exposure levels of the panels would have been higher if the data had been gathered for different years or seasons. Also, the concurrent influenza epidemic was not taken into account in the analysis, and the study period was considered too short to control adequately for time trends (Hurley et al., 2005). On the other hand, unlike the WHO Regional Office for Europe meta-analysis, the meta-analysis of Ward & Ayres was not restricted to European studies, and estimates of adverse effects in published United States studies seemed generally higher than those for Europe. Moreover, Ward & Ayres included the generally negative overall results from the PEACE study as overall results from a single study only (the WHO meta-analysis included 27 separate point estimates). Finally, Ward & Ayres included both general population panels and panels of symptomatic children and found that there were also adverse effects in panels of children from the general population. Hurley and colleagues concluded that the review by Ward & Ayres strongly suggests that the effects of PM on respiratory symptoms should be considered for children in general and not be restricted to children with chronic symptoms.

A study on the decline of ambient air pollution levels and improved respiratory health (Bayer-Oglesby et al., 2005) was recently carried out in nine Swiss communities, covering a broad range of urbanization, air pollution levels and climatic conditions. A total of 9591 children participated in cross-sectional health assessments, between 1992 and 2001. Each child was assigned an estimate of regional PM_{10} level, and changes in mean PM_{10} concentrations were estimated since the first survey. Declining levels of PM_{10} during the study period were associated with a statistically significant decrease in the prevalence of chronic cough, bronchitis, common cold, nocturnal dry cough and symptoms of conjunctivitis. Risk estimates

were adjusted for socioeconomic, health-related and indoor factors. No reduction was observed for the prevalence of sneezing during pollen season, asthma and hay fever.

2.10.8.2 Risk estimates used for the present study: children

An OR of 1.04 (95% CI: 1.02–1.06) per a 10-µg/m^3 increase in PM_{10} concentration was derived from Ward & Ayres (2004) meta-analysis; a background mean daily prevalence rate of LRS (including cough) equal to 15% (in children 5–14 years of age) was calculated as an average from two Dutch studies (Hoek & Brunekreef, 1995; van der Zee et al., 1999).

As in the methodology report of Hurley and colleagues (2005), combining the OR with the background prevalence rate, an estimated new rate of 15.51% (95% CI: 15.25–15.76%), equivalent to an increase of 0.0051 (95% CI: 0.0025–0.0076) in the probability of daily average occurrence of LRS (including cough), was obtained and used in an impact function.

The increase of extra symptoms days per year per child 5–14 years of age is: 1.86 (95% CI: 0.92–2.77) per 10-µg/m^3 increase in PM_{10} concentration.

2.10.8.3 Scientific evidence: adults

The recent WHO Regional Office for Europe meta-analysis of time-series and panel studies of PM and ozone (Anderson et al., 2004) identified six panels in Europe that examined cough (or nocturnal cough or cough and phlegm) in adults, in association with exposure to PM_{10}. The results from three of these studies (two from the Netherlands (Dusseldorp et al., 1995; Boezen et al., 1998)) and one from Paris (Neukirch et al., 1998)) were selected for the meta-analysis, providing an estimated OR of 1.043 (95% CI: 1.005–1.084) per 10-µg/m^3 PM_{10} increment. None of the other three studies was included in the meta-analysis, because (a) one had presented RR rather than OR (Hiltermann et al., 1998) and (b) the other two (two different Dutch settings (van der Zee et al., 2000) had presented results by simply quoting them as "not significant" in the text.

Dusseldorp and colleagues (1995) studied 32 adults (16 years of age and older) who lived near a steel mill and had moderate to severe symptoms. Mean daily prevalence rates were 18.6% for cough, 17.4% for shortness of breath and 8.1% for wheezing. Very similar ORs were observed for shortness of breath and for wheeze after a 2-day lag (1.46 and 1.49, respectively, per 100 µg/m^3 PM_{10}). Over a 6-month period, Neukirch and colleagues (1998) studied 40 nonsmoking adult (16–70 years of age) outpatients in Paris who had mild to moderate asthma. Results were reported both for incidence and for prevalence. The background daily mean prevalence rate of wheeze was 15.1%, and for shortness of breath and nocturnal cough it was 45.7% and 20.1%, respectively. The OR for prevalence of wheeze, in relation to PM, was estimated as 1.059 (95% CI: 0.998–1.123) per 10-µg/m^3 increment in PM_{10} concentration, while the OR for incidence (used in the WHO meta-analysis) was 1.116 (95% CI: 1.052–1.183) per 10-µg/m^3 increment in PM_{10} concentration.

Also, Boezen and colleagues (1998) carried out, for three months during the winter of 1993/1994, a study of 75 symptomatic and asymptomatic adults near Amsterdam. No relationship was found between pulmonary function and PM, while a significant association between cough and PM_{10} was observed (OR = 1.021 (95% CI: 1.001–1.041)). Hiltermann and colleagues (1998) studied 60 adults (18–55 years of age) in the Netherlands who had intermittent to severe asthma (85% used bronchodilators). A mean daily prevalence of 43% and 34.5% was reported, respectively, for shortness of breath and for cough or phlegm (or both) during three summer months. The associations between daily PM_{10} and daily presence

or absence – that is, prevalence – of shortness of breath were reported as RRs, and they were not significant. Van der Zee and colleagues (2000) studied an urban and a non-urban panel of adults (50–70 years of age) who lived in the Netherlands. Respiratory symptoms and PEF were measured for three winters, starting in 1992/1993. Different adults were included each year. Overall, the analyses of the urban symptomatic panel were based on 138 subjects, and the analyses of the non-urban panel on 128 subjects. The researchers did not find consistent associations between daily respiratory symptoms in winter and the air pollutants studied, including PM_{10}, in either the urban or the non-urban areas. However, detailed results for LRS were not given.

Based on the five studies described in the preceding paragraphs, with the exception of the one by Boezen and colleagues (1998), which was considered to be a subset of the data studied by van der Zee and colleagues (2000), the CAFE programme (Hurley et al., 2005) considered other respiratory symptoms in these same panels, focusing on the following symptoms and estimates of effects.

- Dusseldorp and colleagues (1995) estimated ORs of 1.46 and 1.49, respectively, per 100-µg/m^3 increment in PM_{10} concentration, for shortness of breath and for wheeze (2-day lag). By using the concentration–response function for shortness of breath, which was statistically significant, and applying it to LRS generally, Hurley and colleagues estimated an effect of 1.038 (95% CI: 1.010–1.068) per 10-µg/m^3 increment in PM_{10} concentration.
- Neukirch and colleagues (1998) estimated the OR for wheeze in relation to PM was 1.059 (95% CI: 0.998–1.123) per 10-µg/m^3 increment in PM_{10} concentration.
- Hiltermann and colleagues (1998) estimated the relationship between daily PM_{10} and daily prevalence of shortness of breath as an RR of 1.032 (95% CI: 1.006–1.060) per 10-µg/m^3 increment in PM_{10} concentration.
- Van der Zee and colleagues (2000) studied an urban and a non-urban panel of adults with at least one of eleven chronic respiratory symptoms. The estimated RRs were: urban (2-day lag): 1.002 (95% CI: 0.985–1.020) per 10-µg/m^3 increment in PM_{10} concentration; non-urban (1-day lag): 1.005 (95% CI: 0.995–1.015) per 10-µg/m^3 increment in PM_{10} concentration.

A random effects meta-analysis based on the risk estimates selected from the five panels gave an overall estimated RR of 1.017 (95% CI: 1.002–1.032) per 10-µg/m^3 increment in PM_{10} concentration (Hurley et al., 2005).

2.10.8.4 Risk estimates used for the present study: adults

An OR of 1.017 (95% CI: 1.002–1.0032) per 10-µg/m^3 increment in PM_{10} concentration and a background rate – that is, LRS mean daily prevalence, including cough, among symptomatic adults – of 30% were used in the present report, as derived from a meta-analysis on five panel studies (Hurley et al., 2005). Combining the estimated ORs with the mean daily prevalence (see Subsection 2.13.2 for more details on calculations), an impact function was calculated.

The annual increase of symptom days per adult with chronic respiratory symptoms was 1.30 (95% CI: 0.15–2.43) per 10-µg/m^3 increment in PM_{10} concentration.

2.11 Ozone: health end-points – mortality

In the last few years, several meta-analyses on acute mortality and ozone have been published, and their results show (quite consistently) significant increases in risks. Most of the time, however, different metrics were used for the calculation of the pooled estimates. To allow comparisons among the studies, the following conversion relationship was applied, as shown in a recent study (Thurston & Ito, 2001):

20 :15: 8, respectively, for 1-hour maximum : 8-hour maximum : daily average.

2.11.1 Acute effects: all causes of mortality

2.11.1.1 Scientific evidence

Thurston & Ito (2001) reported an RR of 1.001 per 10-µg/m^3 ozone increment, with a very narrow CI. The subset of studies that specified the nonlinear relationship between temperature and mortality yielded a combined estimate about 2% higher, indicating an underestimation of the adverse effects on health obtained by hypothesizing a linear relationship for the concentration–response function.

Stieb, Judek & Burnett (2002) estimated an RR equal to 1.001 (95% CI: 1.000–1.002), based on 109 studies from around the world. A similar RR of 1.001 (95% CI: 1.000–1.001) was estimated by HEI (2000b) and the same results were obtained by Dominici in an HEI report (HEI, 2003) through a reanalysis of 80 United States cities.

An RR of 1.0009 (95% CI: 1.0005–1.0015) in daily mortality was calculated using NMMAPS data (Bell et al., 2004) for 95 large United States urbanized areas (from 1987 to 2000), including 40% of the total United States population. The same study analysed daily mortality as a function of the previous week's ozone levels and provided an RR of 1.002 (95% CI: 1.001–1.003).

The WHO Regional Office for Europe meta-analysis (Anderson et al., 2004) selected 15 European studies, carried out in: France (Le Havre, Lyon, Paris, Rouen, Strasbourg and Tolouse) (Le Tertre et al., 2002b)); Italy (Rome (Michelozzi et al., 1998) and Turin (Cadum et al., 1999)); the Netherlands (the Netherlands (Hoek et al., 2000) and Amsterdam (Roemer & van Wijnen, 2001)); Spain (Barcelona, Madrid and Valencia) (Saez et al., 2002)); and the United Kingdom (West Midlands (Anderson et al., 2001) and London (Bremner et al., 1999)). The combined RR per 10-µg/m^3 increment in ozone concentration was 1.003 (95% CI: 1.001–1.004).

The APHEA-2 study (Gryparis et al., 2004), based on data from 23 European cities/areas for at least three years since 1990, provided no evidence for a relationship between ozone and mortality for the whole year and during the winter months, while an RR of 1.003 (95% CI: 1.002–1.005) was calculated for the summer months. A similar seasonal estimate of an RR of 1.003 (95% CI: 0.997–1.007) was provided by the MISA-2 Italian project for the period from May to September (Biggeri, Bellini & Terracini, 2004).

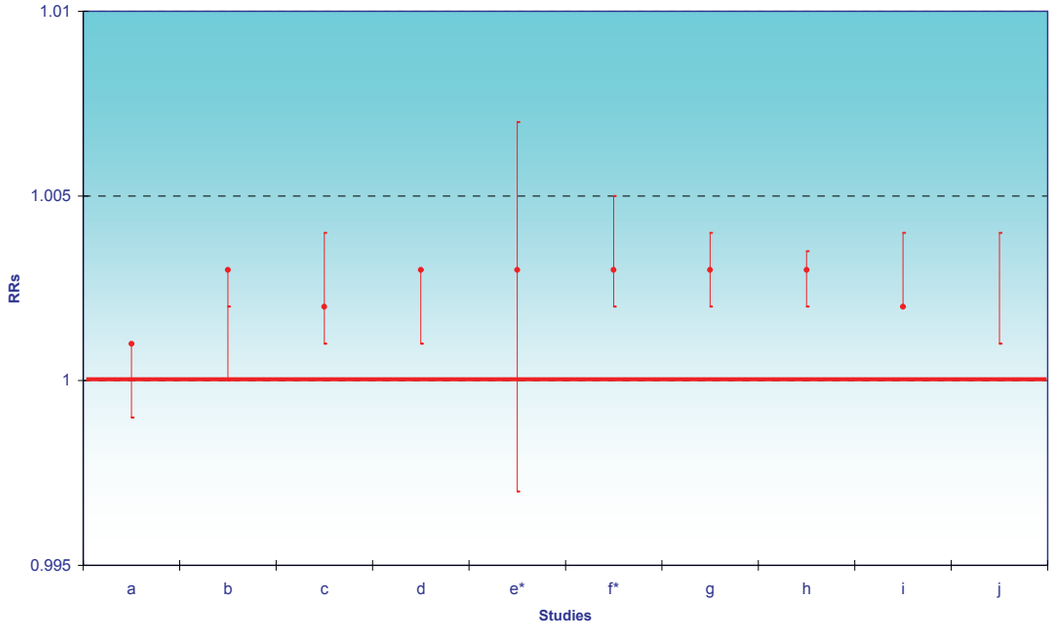
The EPA funded three research teams to carry out independent meta-analyses (Bell, Dominici & Samet, 2005; Ito, De Leon & Lippmann, 2005; Levy, Chemerynski & Sarnat, 2005) on the same working database of studies from the EPA. "The goal was to see whether differences in analytical methods or subjective decisions by the researchers would lead to similar or different conclusions" (Editors, 2005). Levy and colleagues, using data from 14 United States

cities, 13 Canadian cities and 21 European cities, and excluding data from the NMMAPS study and from Mexico City, found an RR of 1.003 (95% CI: 1.002–1.004). Bell and colleagues used 144 estimates of effects from 39 time-series from studies in the United States and other countries; they found an RR of 1.003 (95% CI: 1.002–1.004) and agreed that United States data and non-United States data were similar. Ito and colleagues used 43 estimates from 38 studies carried out in United States and other countries and reported an RR of 1.003 (95% CI: 1.002–1.003).

A study based on more than a million deaths in 14 United States cities was carried out by Schwartz (2005), who estimated an RR of 1.002 (95% CI: 1.001–1.004).

The studies listed above undoubtedly answer the question about whether ambient ozone levels are positively associated with increases in daily mortality: they reported consistent and robust associations, independent of the action of other pollutants. The majority of the studies were conducted for the whole year: analyses carried out by season found larger estimates of effects for the warm months, with the strongest associations found between mortality and exposure to ozone on the same day or the previous day (EPA, 2005b). The results of Bell and colleagues (2004) suggest more delayed adverse effects on health.

Fig. 4. Acute mortality and ozone: results from multi-city studies and meta-analyses



Sources: [a]HEI (2000b); [b]Stieb, Judek & Burnett (2002); [c]Anderson et al. (2004); [d]Bell et al. (2004) ; [e]Biggeri, Bellini & Terracini (2004); [f]Gryparis et al. (2004); [g]Bell, Dominici & Samet (2005); [h]Ito, De Leon & Lippmann (2005); [i]Levy, Chemerynski & Sarnat (2005); [j]Schwartz (2005).

*Summer only.

2.11.1.2 Risk estimates used for the present study

The differences in the RR estimates analysed above are small. All of them range from 1.001 to 1.004. For the present study, it was decided to use the combined estimate calculated by the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004).

An RR of 1.003 (95% CI: 1.001–1.004) per 10-µg/m^3 increment in ozone concentration was used in the present study.

2.11.2 ACUTE EFFECTS: CARDIOVASCULAR CAUSES

2.11.2.1 Scientific evidence

The WHO Regional Office for Europe meta-analysis (Anderson et al., 2004) was carried out before the results of the APHEA-2 project were published. Thirteen studies from France, Germany (rural), Italy, the Netherlands, Spain, Switzerland and the United Kingdom were available, and an RR of 1.004 (95% CI: 1.003–1.005) per 10-µg/m^3 increment in ozone concentration was calculated for cardiovascular mortality.

In the APHEA-2 project (Gryparis et al., 2004), no significant effects of ambient ozone concentrations on cause-specific acute mortality were observed during the cold seasons. For the warm seasons, an increase in the 8-hour ozone concentration by 10 µg/m^3 was associated with a 0.49% (95% CI: 0.34–0.64%) increase in the number of cardiovascular deaths (equivalent to an RR of 1.004 (95% CI: 1.003–1.006) – fixed effect estimate).

Among the three meta-analyses funded by the EPA, only the study by Bell and colleagues (2005) analysed cardiovascular mortality, reporting an RR of 1.004 (95% CI: 1.003–1.006).

The MISA-2 study (Biggeri, Bellini & Terracini, 2004) provided an estimated RR of 1.002 (95% CI: 0.997–1.007) for the relationship between ozone and cardiovascular mortality in the warm period from May to September.

2.11.2.2 Risk estimates used for the present study

Although the APHEA-2 and MISA-2 estimates are more recent, they are based on seasonal risk estimates. The WHO Regional Office for Europe meta-analysis (Anderson et al., 2004) combined RR has been chosen for the present study.

An RR of 1.004 (95% CI: 1.003–1.005) per 10-µg/m^3 increment in ozone concentration was used in the present study.

2.12 Ozone: health end-points – morbidity

In the WHO Regional Office for Europe meta-analysis of European time-series and panel studies of PM and ozone (Anderson et al., 2004), pooled estimated ORs were calculated for the relationship between exposure to ozone and respiratory hospital admissions in adults and in the elderly (65 years of age and older). No summary estimates however, were given for respiratory hospital admissions for children, cardiovascular hospital admissions, and other morbidity outcomes, since the number of studies available for a meta-analysis was insufficient (a minimum of four estimates was required).

More recently, the methodology paper (Vol. 2) of the CAFE programme (Hurley et al., 2005) provided summary risk estimates and impact functions – not restricted to European studies and therefore based on a higher number of published studies, also for other morbidity outcomes not analysed in the WHO meta-analysis.

The health impact assessment for the effects of ozone on morbidity was carried out in the present study through the adoption of the impact functions provided by Hurley and colleagues (2005) for medication use in asthmatic children and adults, for MRADs and for LRS in children (Table 10). The impact assessment for respiratory hospital admissions in adults was based on the WHO Regional Office for Europe combined risk estimate (Anderson et al., 2004). Background rates of each morbidity outcome selected and included in the impact functions are described in the relevant sections.

Table 10. Summary of impact functions for selected morbidity due to ozone

Cause	Impact functions
Asthma (medication use), children	310 (95% CI: 44–569) annual increase in days of bronchodilator usage per 1000 children 6–7 and 13–14 years of age per 10 µg/m^3 increment
Asthma (medication use), adults	730 (95% CI: -225–1570) annual increase in days of bronchodilator usage per 1000 adults ≥ 15 years of age per 10 µg/m^3 increment
MRADs	115 (95% CI: 44–186) MRADsper 1000 adults 18–64 years of age per 10 µg/m^3 increment
LRS, children	0.16 (95% CI: -0.43–0.81) increase of days of LRS per child 5–14 years of age per 10 µg/m^3 increment

Source: Hurley et al. (2005).

2.12.1 RESPIRATORY HOSPITAL ADMISSIONS IN ADULTS OLDER THAN 65 YEARS OF AGE

2.12.1.1 Scientific evidence

To analyse the relationship between ozone and hospital admissions in the elderly, some risk estimates from European time-series studies have been included in the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004). In particular, the meta-analysis was based on results from the APHEA project (Spix et al., 1998) and from a study carried out in West Midlands, United Kingdom (Anderson et al., 2001). A summary RR of 1.005 (95% CI: 0.998–1.012) for a 10-µg/m^3 increase in ozone levels was provided by the WHO Regional Office for Europe meta-analysis, which combined the results from these studies.

In the first study (Spix et al., 1998), the authors reported the results obtained from the quantitative pooling of city-specific analyses. The air pollutants studied were sulfur dioxide, PM, ozone and nitrogen dioxide. The most consistent and strongest finding was a significant increase in daily admissions for respiratory diseases (adults and elderly) due to elevated levels of ozone. Moreover, the elderly were affected more during the warm season, and the authors reported that the results for ozone were in good agreement with the findings of similar United States studies. A summary estimate based on four cities (Amsterdam, London, Paris and Rotterdam) was used in the WHO Regional Office for Europe meta-analysis.

In the second study (Anderson et al., 2001), time-series of health outcomes and environmental data were obtained for the period 1994–1996. The RRs of hospital admissions were estimated, controlling for long-term time trends, seasonal patterns, influenza epidemics, effects due to day of the week, temperature and humidity. The percentage change in daily hospital admissions among the elderly was 0.2% (95% CI: -4.1–4.8%) per ozone increment from the 10th to 90th percentile.

Associations between daily admissions and ozone levels were analysed for respiratory causes in the elderly (older than 65 years of age) in Hong Kong and in London (Wong et al., 2002). The RR for a 10-μg/m^3 increase in ozone concentration (single-pollutant analysis) for mean 0–1-day lag was very similar in the two cities: 0.8 (95% CI: 0.3–1.3) in Hong Kong and 0.8 (95% CI: 0.2–1.4) in London. These associations tended to be stronger for shorter lags in Hong Kong and for longer lags in London. Associations were stronger in the colder seasons in Hong Kong and in the warmer seasons in London, periods during which the levels of humidity are at their lowest in each city. The authors concluded that air pollution has remarkably similar associations with daily cardiorespiratory admissions in both cities, in spite of considerable differences between the social, lifestyle and environmental factors in these cities.

The impact of ozone on daily respiratory admissions, specifically in the elderly, was analysed in a few other recent studies. The greater Vancouver, British Columbia study (Yang et al., 2003) included adults 65 years of age and older who had acute hospital admissions for any respiratory disease during the 13-year period 1986–1998. Respiratory admissions were associated with ozone levels 2–5 days prior to admission, with the strongest association observed at a 4-day lag. An OR for hospital admission of 1.13 (95% CI: 1.09–1.18) per interquartile range was observed for ozone. The results were not attenuated after adjustment for other pollutants and for socioeconomic status.

The MISA-2 study (Biggeri, Bellini & Terracini, 2004) reported age-group-specific percentage increases in the risk of respiratory-related hospital admissions due to a 10-μg/m^3 increase in daily ozone concentration. The estimates varied: 1.10% (95% CI: -0.14–2.33%), in the age group of 65–74 year olds; 0.32% (95% CI: -0.90–1.52%), among subjects 75 years of age and older; and 1.20% (95% CI: -1.04–3.13%), in the very elderly (85 years of age and older). These estimates, however, referred to the summer period only (from May to September), and they could not be adopted in the present study.

2.12.1.2 Risk estimates used for the present study

The estimates of the MISA-2 study are available for the warm season only, so the WHO Regional Office for Europe combined risk for adults over 65 years of age was chosen for the present report.

An RR of 1.005 (95% CI: 0.998–1.012) per 10-μg/m^3 increment in ozone concentration was used for the present study.

Background rates were estimated from the MISA-2 study city-specific respiratory-related hospital admissions, through the use of provincial age-specific rates (Health for All Italia, 2005), because admissions were not disaggregated into age groups. Background rates for Padua were selected with the MISA-2 protocol but were not available from the MISA-2 publications; instead they were retrieved from the web (Department of Environmental Medicine and Public Health, University of Padua – Office of Hygiene, ARPAV – Padua Department, and Local Health Authority No. 16 of Padua, 2005).

2.12.2 ASTHMA IN CHILDREN

2.12.2.1 Scientific evidence

Ozone is considered a more potent oxidant than nitrogen dioxide, clearly causing acute exacerbations of asthma, by impairing lung growth and inducing a greater decline in lung function over time, especially in children of low birth weight (Mortimer et al., 2000). In the WHO Regional Office for Europe review of the effect of air pollution on children's health (WHO Regional Office for Europe, 2005a), the overall available evidence was judged sufficient to assume a causal relationship between exposure to ozone and aggravation of asthma in children.

In particular, in the WHO Regional Office for Europe review of the effects of air pollution on children's health and development, Weiland & Forastiere (2005) reviewed studies on aggravation of childhood asthma (short-term effects). Among the 20 studies that evaluated the effect of ozone on hospital admissions, emergency department visits and calls to doctors for asthma in children, some studies in Europe were paradoxically found to show a protective effect for this pollutant. On the other hand, studies performed outside of Europe tended to show an increase in hospital admissions for asthma, and larger estimates of effects were found for the warmer seasons.

In a panel study carried out in the Netherlands (Gielen et al., 1997), the effect of summer air pollution in Amsterdam was evaluated in a group of 61 asthmatic children (7–13 years of age), of whom 77% were taking asthma medication. The PEF was measured twice daily, the occurrence of acute respiratory symptoms and the use of medication was registered daily, and exposure to ozone and other pollutants was estimated from ambient concentrations. The associations were evaluated using time-series analysis. After adjusting for pollen, time trend and day of the week, black smoke (in particular) was associated with acute respiratory symptoms and with the use of medication. Weaker associations were found for PM_{10} and ozone. The mean daily prevalence rate of bronchodilator usage, based on 61 subjects with completed data for at least 60% of the days, was 40%. This estimate was adopted by the methodology paper (Vol. 2) of the CAFE programme (Hurley et al., 2005), to calculate the impact function for ozone and bronchodilator use (β2 agonist).

In the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004), only one European panel study was considered to evaluate the association between use of medication and exposure to ozone in symptomatic children. This study (Just et al., 2002) was carried out in Paris on 82 children (7–15 years of age) medically diagnosed with asthma and followed for three months, during spring and early summer. Outcomes included the incidence and prevalence of asthma attacks, nocturnal cough, supplementary use of β2 agonists, symptoms of airway irritation, and PEF value. Ozone levels had a great effect on supplementary bronchodilator use, corresponding to an OR of 1.410 (95% CI: 1.050–1.890), adjusted for the lack of independence between daily health outcomes, temporal trends, pollen and weather conditions. This estimate was found when analyses were restricted to days on which the children used no corticosteroids.

2.12.2.2 Risk estimates used for the present study

The OR for supplementary use of a bronchodilator was derived from the only relevant study selected by the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004), which provided a positive and significant risk estimate (RR of 1.41 (95% CI: 1.05–1.89)). As in the analysis by Hurley and colleagues (2005), the background rate (mean daily prevalence)

on days at risk (1% of person–days of all children aged 5–14 years in countries of western Europe (Hurley et al., 2005)) was 40% and was derived from the study by Gielen and colleagues (1997).

Combining all these elements, as described in detail in Subsection 2.13.2, an impact function can be derived and applied in the present study.

The annual increase of bronchodilator usage in children of 310 (95% CI: 44–569) days per 1000 children per 10-µg/m^3 ozone increment was used in the present study.

2.12.3 ASTHMA IN ADULTS

2.12.3.1 Scientific evidence

The WHO Regional Office for Europe meta-analysis (Anderson et al., 2004) identified two relevant European panel studies on medication use and ozone in symptomatic adults. The first study (Higgins et al., 1995) was performed in the United Kingdom on 75 subjects (16 years of age and older) with asthma or COPD, who were followed for a month. Results from multi-pollutant models showed a highly statistically significant relationship between (24-hour average) ozone and daily bronchodilator usage: an OR of 1.44 (95% CI: 1.14–1.82) per 10-µg/m^3 increase in ozone level was reported.

The second study was carried out in the Netherlands (Hiltermann et al., 1998) on 60 subjects (18–55 years of age) with intermittent to severe persistent asthma, who were followed for three months. The ozone concentration was represented by the daily maximum 8-hour moving average. A positive, although non-significant increase in daily prevalence of bronchodilator usage per 10-µg/m^3 increase in ozone level (RR = 1.009 (95% CI: 0.997–1.020)) at 1-day lag was reported.

More recently, Hurley and colleagues (2005), to define the concentration–response function to be adopted for the impact function of ozone on adult asthma and to be consistent with the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004), indicated that results from analyses based on daily 8-hour ozone concentrations and adjusted for climate and other confounders (when available) had to be chosen. Thus, the present study preferred results from Hiltermann and colleagues (1998).

2.12.3.2 Risk estimates used for the present study

The statistically non-significant OR of 1.009 (95% CI: 0.997–1.020) per 10-µg/m^3 increase in ozone concentration was obtained from Hiltermann and colleagues (1998), one of the two relevant studies selected by the WHO Regional Office for Europe meta-analysis (Anderson et al., 2004). A background rate (mean daily prevalence in bronchodilator use) of 32% was derived, as in the analysis by Hurley and colleagues (2005), from the study by Hiltermann and colleagues (1998). Combining this information with the percentage of adults with persistent asthma (4.5%, as already described in Subsection 2.10.6), an impact function was derived and applied in the present study.

The value used in the present study for the annual increase of bronchodilator usage in adults 20 years of age and older is 730 (95% CI: -225–1570) days per 1000 adults per 10-µg/m^3 increase in ozone level.

2.12.4 MRADs

2.12.4.1 Scientific evidence

No clear or consistent relationship, linking ozone with respiratory-related RADs, was reported by the methodology paper (Vol. 2) of the CAFE programme (Hurley et al., 2005). As to MRADs, only one study (Ostro et al., 1989) was considered relevant. In this study, RADs were determined between 1976 and 1981 from the annual HIS, a nationally representative cross-sectional sample of 50 000 households. Restricted activities were taken from 2-week recall surveys of working adults, and a regression analysis was performed for $PM_{2.5}$ and ozone each year. Ozone concentrations were highly correlated with temperature. For MRADs in multi-pollutant models, the authors reported that an inverse-variance weighting yielded a 0.2% increase per 1-µg/m^3 increase in 2-week average 1-hour maximum ozone concentrations. No uncertainty bounds were reported, but a simple variance estimate based on reported standard deviations by year yielded a 95% CI of 0.1–0.3%.

Hurley and colleagues (2005) derived a percentage increase in RR of 1.48% (95% CI: 0.57–2.38%) per 10-µg/m^3 increase in ozone level (daily 8-hour average), using a conversion factor of 1.33, based on a study by Schwartz (1997).

2.12.4.2 Risk estimates used for the present study

An estimate of the OR of 1.48% (95% CI: 0.57–2.38%) per 10-µg/m^3 increase in ozone concentration and a baseline rate of 7.8 MRADs per year, among people in employment (18–64 years of age) were applied in the present study, following Ostro & Rothschild (1989). As in the analysis by Hurley and colleagues (2005), linking this background rate with a percentage increase of 1.48% per 10-µg/m^3 increase in ozone concentration, an annual increase was obtained and inserted in an impact function.

The yearly increase in MRADs per 1000 adults 18–64 years of age was 115 MRADs (95% CI: 44–186 MRADs) per 10-µg/m^3 increase in ozone concentration.

2.12.5 LRS IN CHILDREN

2.12.5.1 Scientific evidence

The methodology paper for the cost–benefit analysis of the CAFE programme (Hurley et al., 2005) reported that:

> … there is convincing evidence that daily variations in ozone are associated with lower respiratory symptoms, including cough… [and] effects on LRS/cough/phlegm are not restricted to people with chronic respiratory symptoms, e.g. asthma; indeed, there is no strong evidence that relative risks (in practice, odds ratios) are higher among people with chronic respiratory disease than among the general population.

In agreement with Hurley and colleagues (2005), the WHO Regional Office for Europe review on the effect air pollution on children's health (WHO Regional Office for Europe, 2005a) concluded that a significant body of evidence suggests a causal relationship between exposure to ambient air pollution (including ozone) and an increased incidence of upper and lower respiratory symptoms.

The WHO Regional Office for Europe meta-analysis (Anderson et al., 2004) identified only one relevant European panel study on cough and ozone in symptomatic children. This study (Just et al., 2002) included 82 Parisian asthmatic children (7–15 years of age) who were followed for three months. Ozone levels had a positive, although non-significant effect on nocturnal cough, with an OR of 1.040 (95% CI: 0.920–1.176).

In a recent review on particulate air pollution and panel studies in children (Ward & Ayres, 2004), the impact of PM_{10} on both cough and LRS was larger for studies conducted under conditions of relatively high-concentration levels of ozone.

The results from the paper by Declercq & Macquet (2000) were adopted by the methodology paper for the health impact assessment of the CAFE programme (Hurley et al., 2005), to quantify the respiratory effects among children in the general population in Europe. This study examined the prevalence of symptoms, in relation to 8-hour daily ozone concentrations, in 91 10-year-old schoolchildren (including 7 who were asthmatic) from the general population in Armentieres, northern France, in the early summer (April to June). Measurements of ambient ozone concentrations were obtained from a continuous fixed monitor, located 850 meters from the school. Although ozone concentrations remained moderate (1-hour level <180 $\mu g/m^3$) during the study period, a 30-$\mu g/m^3$ increase in the daily maximum 8-hour mean level of ozone was associated with an increased daily prevalence of cough and PEF.

2.12.5.2 Risk estimates used for the present study

The OR of 1.03 (95% CI: 0.92–1.15) per 10-$\mu g/m^3$ increase in ozone concentration was applied to LRS (excluding cough), as derived from a French study (Declercq & Macquet, 2000). A background OR of 0.0523, derived from a mean daily prevalence rate of LRS of 1.5%, was used, as in the analysis by Hurley and colleagues (2005). Combining this information, an impact function was estimated and applied in the present study.

The increase of days of LRS per year per child 5–14 years of age was 0.16 (95% CI: -0.43–0.81) per 10-$\mu g/m^3$ increase in ozone concentration.

2.13 Methods for quantification

As discussed in Sections 2.9 and 2.10, several impact estimates for large Italian cities have been calculated recently by studies carried out in Italy and Europe. The published results vary across studies, depending on the methods used in the analyses, notably the choice of counterfactual factors and of concentration–response coefficients.

The first WHO Regional Office for Europe report on the health impact assessment of PM_{10} in the eight major Italian cities (Martuzzi et al., 2002) calculated impact estimates for chronic mortality and for a wide set of acute morbidity outcomes, extrapolating RRs from the most recent scientific literature available at the time.

The European multi-centre study, APHEA 2 (Katsouyanni et al., 2001), estimated RRs for acute mortality and hospital admissions from exposure to PM_{10} and ozone: Turin, Milan and Rome were included in the analysis. Health impact assessment methods were subsequently developed by the APHEIS-2 study (Medina et al., 2005), which calculated impact estimates for mortality (chronic and acute) and hospital admissions. Only Rome was included in this study and a city-specific impact estimate is available.

The Italian study, MISA 2 (Biggeri, Bellini & Terracini, 2004), a time-series study, calculated risk estimates from exposure to PM_{10} and ozone and impact estimates (from PM_{10} only) for acute mortality and hospital admissions in 15 large Italian cities. Meta-analytic estimates and city-specific estimates were calculated.

The CAFE Working Group calculated a national impact estimate of several pollutants (PM_{10} and ozone included), in terms of YLL, under the hypothesis of different possible future scenarios (Amann et al., 2005). This study further developed the methods used in the first WHO Regional Office for Europe report and updated the health end-point analysed. As seen in previous sections of the present report, almost all risk coefficients for the present study were updated according to the technical specifications of recognized international organizations, task forces or working groups. Impact estimates for chronic and acute mortality and for morbidity end-points have been calculated by city, sex and age group, using the most recent scientific evidence. In addition, YLL was also considered (see Subsection 2.13.4), using methods for estimating the mortality component of GBD.

2.13.1 Number of cases associated with a given counterfactual factor

As in the previous WHO Regional Office for Europe publication (Martuzzi et al., 2002), a simple algorithm was used to calculate the number of cases (such as deaths and hospital admissions) associated with a given counterfactual factor (see Section 2.7), exposed population (see Section 2.1), specific mortality (morbidity) rate and RR estimate. For each of the health end-points selected, an estimate of RR was obtained or calculated from the literature, as described previously (Section 2.8). The RR is the increase in the probability of occurrence of the adverse effect on health associated with a given change in exposure level (typically 10 $\mu g/m^3$ for PM_{10} and ozone).

The number of cases attributable to an air pollution concentration over a given counterfactual factor, E, is given by the following equation:

$$E = A * B * (C/10) * P, \qquad \text{(Equation 8)}$$

where

- P = the population exposed, obtained from census data;
- C = the relevant change in concentration (difference between the observed concentration and the counterfactual level), obtained from monitoring networks in each city; and
- A = the proportion of effect on health attributable to air pollution, which can be calculated as follows:

$$A = \frac{(RR - 1)}{RR}. \qquad \text{(Equation 9)}$$

B_0 is the mortality (morbidity) rate of the given health end-point that would be observed at the given counterfactual level and that can be calculated as:

$$B_0 = \frac{B}{[1+(RR-1)*(C/10)]}, \qquad \text{(Equation 10)}$$

where B is the observed mortality (morbidity) rate of the adverse effect on health under the current exposure obtained from available health statistics.

The C term is somewhat different for PM_{10} and ozone: for PM_{10}, C is the difference in concentration, while for ozone calculations there is no explicit counterfactual factor, because it is already included in the SOMO35 indicator (see Subsection 2.6.2, Equation 5). For this reason, for an ozone impact assessment, the following formulation was used:

$$C = SOMO\,35\ (or\ SOMO\,0). \qquad \text{(Equation 11)}$$

As an example of these calculations, using data presented in the earlier WHO Regional Office for Europe publication (Martuzzi et al., 2002), Equation 8 allows one to calculate how many deaths would have been saved in the city of Turin if the observed PM_{10} concentration could have been reduced to the given counterfactual level.

Assuming that:

- P, the exposed population (adults over 30 years of age in Turin) = 642 260;
- $C = 23.8$ (53.8 µg/m^3 minus 30 µg/m^3 – that is, the observed PM_{10} concentration minus the counterfactual factor);
- RR = 1.026, the relative risk for a 10-µg/m^3 change in concentration, for all causes of chronic mortality (excluding accidents) in adults over 30 years of age;
- $A = \frac{(1.026-1)}{1.026} = 0.0253$, the attributable proportion of all causes mortality from PM_{10};
- $B = 0.0115$; and
- $B_0 = 0.0115/[1+(1.026-1)*(23.8/10)] = 0.0108,$

it follows that:

E = (0.0253) * (0.0108) * (23.8/10) * (642 260)= 420 yearly extra deaths due to a PM_{10} level exceeding the counterfactual factor of 30 µg/m^3.

In the present report, impacts are also given disaggregated by sex and age classes, to provide details on the distribution of cases in the population attributable to exposure to PM. Breakdown by age and sex is also used for the calculation of YLL (see Subsection 2.13.4).

The number of cases attributable to exposure to PM is obtained by summing the number of cases in the 13 cities.

Credibility intervals (CrI) were calculated using the software WinBugs (Spiegelhalter, Thomas & Best, 1999) through an algorithm, known as Gibbs sampling (Gelfand, Hills & Racine-Poon, 1990), belonging to the family of Monte Carlo simulation iterative methods.

2.13.2 IMPACT FUNCTIONS FOR MORBIDITY END-POINTS

For some of the morbidity outcomes selected, concentration–response functions and background rates (such as incidence and prevalence) were combined to derive impact functions. These functions express the number of cases attributable to exposure to PM per year, per unit population (for example, per 1000 people at risk), per unit exposure (for example, 10 µg/m^3).

Impact functions can be based on risks of occurrence of binary events estimated by logistic regression. In these cases, the algorithms involve the OR *o*:

$$o = \frac{p}{1-p} \qquad \text{(Equation 12)}$$

where *p* is equal to the probability of occurrence of the event. If *p* is small, the OR can be expressed as a percentage change; otherwise, more complicated calculations are needed, as illustrated in the following example from Hurley and colleagues (2005).

For bronchodilator usage in children with asthma, the logistic regression analysis suggested a risk of 1.41 per 10-µg/m^3 increase in ozone concentration, with a mean daily prevalence of bronchodilator usage of 40% – that is, $p = 0.4$. This prevalence translates to an odds of 0.4/(1.0–0.4) = 0.666. The OR and the odds, 1.41 and 0.666, respectively, can be combined multiplicatively, giving a new odds figure equal to 1.41*0.666 = 0.94. This odds figure is converted back to a probability using the inverse of Equation 12 ($p=o/1+o$): 0.94/1.94 = 0.485. So, an increase in exposure of 10 µg/m^3 of ozone in a population with a background prevalence of 40% produces a prevalence of 48.5%. The difference between the two, 0.085, is the extra daily probability of bronchodilator usage per 10-µg/m^3 increment in ozone concentration. On a yearly scale, assuming that a child was at risk for the whole year, this is equivalent to 0.085*365 = 31 extra usage days per year. Since the proportion of days at risk was estimated as 1% of person–days of all children between 5 and 14 years of age, the following impact function was derived:

The annual increase in days of bronchodilator usage per 1000 children 5–14 years of age is 310 (95%CI: 44–569) per 10-µg/m^3 increment in ozone concentration.

The CI is derived by making the same calculation for the 95% confidence bounds. Final impact estimates can be obtained by applying the result of the impact function described to the exposed population and to the change in exposure observed.

2.13.3 LIFE TABLES AND LIFE EXPECTANCIES

For the mortality end-points used to calculate the number of YLL, life expectancies disaggregated by age groups and sex were calculated. Complete life tables (for every single year of age) and abridged life tables (for age groupings, based on the assumption that death rates are similar at neighbouring ages and, hence, death rates calculated from groups of ages can be used) were constructed (Mathers et al., 2001). Two vectors of data are needed for the calculation of a complete life table: a population vector N_x (at the start of the year x) and a death vector D_x (for deaths during the year x). The probability q_x of dying in year x is estimated, as follows:

$$q_x = \frac{D_x}{N_x}. \qquad \text{(Equation 13)}$$

Usually, in life tables, the mid-year population (P_x) is used, so q_x has to be calculated in an indirect way, through the age-specific death rates M_x, which are defined as follows:

$$M_x = \frac{D_x}{P_x}. \qquad \text{(Equation 14)}$$

Under the assumption that people who died in the year x lived half a year, we have:

$$N_x = P_x + 0.5 * D_x \qquad \text{(Equation 15)}$$

and from Equations 13-15 we obtain:

$$q_x = \frac{M_x}{1 + (1 - a_x) * M_x}, \qquad \text{(Equation 16)}$$

where the fraction of a year lived ($a_x = 0.5$) is assumed to be equal to 0.1 for the first age class. Using Equation 16, the number of survivors at exact age x from an initial population of 100 000 (l_x) is estimated by:

$$l_x = l_{x-1} * q_{x-1}, \qquad \text{(Equation 17)}$$

where l_x is needed to calculate life expectancy. For the last open class (for example, the class of people older than 85 years of age (≥85)), one assumes $D_{\geq 85} = l_{85}$.

For the calculation of the life expectancies, L_x (the total number of person–years lived between the exact ages of x and x+1) and T_x (representing the total number of person–years lived after the age of x) are also needed, where:

$$L_x = l_{x+1} + a_x * D_x \qquad \text{(Equation 18)}$$

and

$$T_x = T_{x+1} + L_x. \qquad \text{(Equation 19)}$$

The life expectancy at age x, e_x, is defined as the expected (average) number of years of life left to a person aged x, and is given by the following expression:

$$e_x = \frac{T_x}{l_x}. \qquad \text{(Equation 20)}$$

Using local demographic and health data, life expectancies at age x were calculated for each city in the present study, by sex and one-year age group. From these complete life tables, abridged life tables and life expectancies (with age groups 0–1, 1–4, then 5-year age groups, until the group of people older than 85 years of age) were calculated with an automatic procedure and used for the calculation of YLL.

City-specific life expectancies, calculated for 2001, are reported in the Annex (Tables 10 and 11).

2.13.4 Burden of disease: YLL

To further characterize the impacts on mortality due to air pollution, proportions and number of deaths attributable to exposure to PM were complemented by the number of YLL due to premature mortality. The methodology for calculating YLL is that used for disability-adjusted life-years (DALYs), introduced by WHO in 1996 (Murray & Lopez). DALYs include a second component, years of life lived with disability (YLDs), which cannot be estimated with the kind of data available to a study like the present one. Therefore, only the YLL component was estimated.

The calculation of YLL by age class and sex is as follows:

$$YLL_{x,sex} = E_{x,sex} * e_{x,sex}, \qquad \text{(Equation 21)}$$

where $E_{x,sex}$ are the deaths attributable to exposure to PM by age class x and sex (see Equation 8) and $e_{x,\,sex}$ are the life expectancies (described in Subsection 2.13.3) for the same subgroups.

As for the calculation of cases attributable to exposure to PM, CrI were calculated using the software WinBugs (Spiegelhalter, Thomas & Best, 1999) through an algorithm, known as Gibbs sampling (Gelfand, Hills & Racine-Poon, 1990).

Chapter 3. Results

This chapter begins by describing city-specific concentration data for 2002–2004 (Section 3.1). Health impact assessment results for all 13 cities combined are given in the sections that follow. The health impact of PM_{10} on mortality (in terms of deaths attributable to PM_{10} and YLL) for the four counterfactuals is reported in Section 3.2. Cases of morbidity attributable to PM_{10} (such as hospital admissions and RADs) for the four counterfactuals are reported in Section 3.3. The impact of ozone on mortality and morbidity for SOMO35 and SOMO0 are presented in Section 3.4. More detailed results on deaths attributable to these two air pollutants and YLL by sex and age group are given in the Annex.

3.1 Environmental exposure

3.1.1 PM_{10}

Annual mean concentrations of PM_{10} for the triennium 2002–2004 in the cities under study are given in Table 11. Yearly averages are based on data available from selected traffic and background monitoring stations, described in detail in the Annex (Table 7). Triennium average PM_{10} concentrations ranged from 26.3 µg/m^3 (Trieste) to 61.1 µg/m^3 (Verona) with a population weighted mean of 45.3 µg/m^3.

Table 11. Annual and triennium average concentration of PM_{10} (µg/m^3)

City	Annual concentration by year			Average
	2002	2003	2004	
Turin	51.4	53.0	54.0	52.8
Genoa	48.9	49.1	40.6	46.2
Milan	60.6	56.7	55.2	57.5
Padua	57.9	60.0	57.2	58.4
Verona	53.2	63.5	66.5	61.1
Venice-Mestre	46.0	51.0	46.5	47.8
Trieste	33.6	28.8	16.6	26.3
Bologna	45.3	44.7	38.5	42.8
Florence	43.4	43.2	43.6	43.4
Rome	44.1	42.0	42.1	42.7
Naples	36.5	38.9	33.1	36.2
Catania	32.6	25.7	31.5	29.9
Palermo	41.7	38.9	39.0	39.9

Source: Annex Table 7.

3.1.2 OZONE

Annual ozone values of SOMO35 for the triennium 2002–2004 in the cities under study are given in Table 12. Yearly averages are based on data available from selected urban background monitoring stations, described in detail in the Annex (Table 8). Triennium average SOMO35 values ranged from 3.1 µg/m^3 (Catania) to 33.6 µg/m^3 (Palermo). Summary values for SOMO0 are reported in the Annex (Table 12).

Table 12. Annual and triennium average levels of ozone: SOMO35 (µg/m^3)

City	Annual concentration by year			Average
	2002	2003	2004	
Turin	25.0	36.4	24.2	28.5
Genoa	19.8	24.2	14.7	19.6
Milan	12.7	19.3	9.6	13.8
Padua	12.9	32.0	20.0	21.6
Verona	0.8	32.8	22.7	18.8
Venice-Mestre	3.8	28.2	14.6	15.5
Trieste	4.9	22.1	23.9	16.9
Bologna	10.3	22.5	18.0	16.9
Florence	21.3	19.9	10.5	17.2
Rome	8.2	27.0	20.0	18.4
Naples	19.1	9.8	33.1	20.7
Catania	6.6	1.5	1.1	3.1
Palermo	37.3	43.0	20.5	33.6

Source: Annex Table 8.

3.2 PM_{10}: mortality

Several tables describe the number of deaths that could be prevented if mortality rates predicted at PM_{10} concentrations of 20, 30 and 40 µg/m^3 and reduced by 10% prevailed, in place of the observed mortality rates. Mortality results are described by sex and health outcome in Tables 13–16. The same outcomes, in terms of YLL, are reported in Tables 17-20.

3.2.1 DEATHS ATTRIBUTABLE TO PM_{10} REDUCED TO 20 µg/m^3

For all causes of chronic mortality (Table 13), excluding accidental causes, in adults older than 30 years of age, 8220 deaths (9.0% of all deaths) are attributable to the level of PM_{10} exceeding 20 µg/m^3. This percentage rises to 11.6% for lung cancer and to 19.8% for infarction. The lowest percentage attributable is observed for stroke (3.3% of cases). For all causes of acute mortality, excluding accidental causes, for all ages, 1372 deaths (1.5 % of all deaths) are attributable to PM_{10} levels exceeding 20 µg/m^3, and 2.1% of acute cardiovascular deaths and 3.1% of acute respiratory causes are attributable to the level of PM_{10} exceeding 20 µg/m^3. All estimates are statistically significant at the 95% CI level.

Table 13. Deaths attributable to levels of PM_{10} exceeding 20 µg/m^3

Causes of death	Males		Females		Total			
	No.	95% CrI	No.	95% CrI	No.	95% CrI	% attr cases	95% CrI
Chronic effects[a]								
All causes of mortality (excluding accidents)	3909	2996–4827	4311	3315–5310	8220	6308–10140	9.0	6.9–11.1
Lung cancer	551	392–711	191	137–245	742	530–956	11.6	8.3–14.9
Infarction	1293	1220–1367	1269	1198–1341	2562	2418–2707	19.8	18.7–21
Stroke	126	79–174	203	132–275	329	207–452	3.3	2.1–4.6
Acute effects[b]								
All causes of mortality (excluding accidents)	654	574–735	718	631–806	1372	1204–1540	1.5	1.3–1.7
Cardiovascular causes	362	303–421	481	404–558	843	706–980	2.1	1.8–2.5
Respiratory causes	99	77–121	86	67–106	186	145–227	3.1	2.4–3.8

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.2.2 Deaths attributable to PM_{10} reduced to 30 µg/m^3

Obviously, the percentages of deaths attributable to levels of PM_{10} exceeding 30 µg/m^3 (Table 14) are lower: 5196 deaths for chronic mortality (5.7% of the total deaths in people older than 30 years of age, 478 for lung cancer (7.5%), 1684 for infarction (13%) and 203 for stroke (2.1%) are attributable to PM_{10} pollution. Lower estimates are observed for acute effects on health as well: 844 cases of acute mortality (0.9%), 516 deaths from cardiovascular causes (1.3%) and 115 deaths from respiratory causes (1.9%). Cities with average concentrations below 30 µg/m^3 do not contribute to these impact estimates.

Table 14. Deaths attributable to levels of PM_{10} exceeding 30 µg/m^3

Causes of death	Males		Females		Total			
	No.	95% CrI	No.	95% CrI	No.	95% CrI	% attr cases	95% CrI
Chronic effects[a]								
All causes of mortality (excluding accidents)	2465	1857–3078	2731	2059–3405	5196	4028–6357	5.7	4.4–7
Lung cancer	354	250–457	124	88–160	478	339–617	7.5	5.3–9.6
Infarction	851	796–905	833	781–886	1684	1577–1791	13.0	12.2.–13.9
Stroke	78	47–108	125	76–175	203	123–284	2.1	1.2–2.9
Acute effects[b]								
All causes of mortality (excluding accidents)	401	350–453	443	386–500	844	735–953	0.9	0.8–1
Cardiovascular causes	221	183–260	295	245–345	516	429–604	1.3	1.1–1.5
Respiratory causes	61	47–76	54	41–68	115	88–143	1.9	1.5–2.4

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.2.3 Deaths attributable to PM_{10} reduced to 40 µg/m^3

Deaths attributable to levels of PM_{10} exceeding 40 µg/m^3 are described in Table 15. There are 2270 deaths from chronic mortality (2.5% of the total deaths in people older than 30 years of age), 214 deaths from lung cancer (3.3%), 749 from infarction (5.8%) and 88 from stroke (0.9%) attributable to air pollution. Lower estimates can be observed for acute effects on health as well: 361 cases of acute mortality (0.4%), 218 deaths from cardiovascular causes (0.6%) and 51 deaths from respiratory causes (0.8%). Cities with average concentrations below 40 µg/m^3 do not contribute to these impact estimates.

Table 15. Deaths attributable to levels of PM_{10} exceeding 40 µg/m^3

Causes of death	Males		Females		Total			
	No.	95% CrI	No.	95% CrI	No.	95% CrI	% attr cases	95% CrI
Chronic effects[a]								
All causes of mortality (excluding accidents)	1068	751–1384	1202	844–1558	2270	1595–2941	2.5	1.8–3.2
Lung cancer	157	108–207	56	39–74	214	147–280	3.3	2.3–4.4
Infarction	376	348–405	373	344–401	749	692–805	5.8	5.4–6.2
Stroke	33	17–49	55	28–81	88	45–131	0.9	0.5–1.3
Acute effects[b]								
All causes of mortality (excluding accidents)	170	144–196	191	161–220	361	305–416	0.4	0.3–0.5
Cardiovascular causes	93	74–112	125	99–151	218	164–262	0.6	0.4–0.7
Respiratory causes	26	18–34	25	17–32	51	36–66	0.8	0.6–1.1

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.2.4 DEATHS ATTRIBUTABLE TO PM_{10} REDUCED BY 10%

By reducing the concentration of PM_{10} by 10% (Table 16) in every city, a total of 1610 deaths from chronic mortality could be avoided (1.8% of all deaths in people older than 30 years of age). Also, 149 cases of lung cancer (2.3%), 586 cases of infarction (4.5%), 61 cases of stroke (0.6%), 258 cases of acute mortality for all ages (0.3%), 154 deaths for cardiovascular causes (0.4%) and 34 for respiratory causes (0.6%) could be prevented.

Table 16. Deaths attributable to levels of PM_{10} being reduced by 10%

Causes of death	Males		Females		Total			
	No.	95% CrI	No.	95% CrI	No.	95% CrI	% attr cases	95% CrI
Chronic effects[a]								
All causes of mortality (excluding accidents)	766	584–949	843	646–1041	1610	1232–1989	1.8	1.4–2.2
Lung cancer	111	77–144	38	26–50	149	104–194	2.3	1.6–3
Infarction	295	274–316	291	271–311	586	547–627	4.5	4.2–4.9
Stroke	23	14–33	38	24–51	61	39–83	0.6	0.4–0.8
Acute effects[b]								
All causes of mortality (excluding accidents)	123	109–137	135	119–151	258	227–288	0.3	0.2–0.3
Cardiovascular causes	66	55–78	88	74–102	154	130–179	0.4	0.3–0.5
Respiratory causes	18	14–23	16	10–21	34	27–41	0.6	0.4–0.7

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.2.5 YLL: PM_{10} REDUCED TO 20 µg/m^3

A total of 90 151 YLL for all causes of chronic mortality are attributable to PM_{10} levels exceeding 20 µg/m^3 (Table 17). Lung cancer accounts for 10 305 YLL, infarction for 24 718 YLL and stroke for 2832 YLL. For acute effects, a total of 15 764 YLL for all causes of mortality are attributable to PM_{10} levels exceeding 20 µg/m^3, with 7749 YLL due to cardiovascular causes and 1641 YLL to respiratory causes.

Table 17. Years of life lost: PM_{10} reduced to 20 µg/m^3

Causes of death	Males		Females		Total	
	No.	95% CrI	No.	95% CrI	No.	95% CrI
Chronic effects[a]						
Mortality (excluding accidents)	45 311	34 650–56 030	44 840	34 470–55 240	90 151	69 130–111 200
Lung cancer	7 339	5 199–9 486	2 966	2 111–3 825	10 305	7 317–13 310
Infarction	14 062	13 250–14 870	10 656	10 060–11 250	24 718	23 310–26 130
Stroke	1 176	738–1 618	1 655	1 050–2 264	2 832	1 784–3 883
Acute effects[b]						
Acute mortality (excluding accidents)	7 995	7 008–8 984	7 770	6 828–8 715	15 764	13 830–17 700
Cardiovascular diseases	3 782	3 159–4 407	3 967	3 332–4 604	7 749	6 491–9 010
Respiratory diseases	904	702–1 105	737	574–902	1 641	1 276–2 007

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.2.6 YLL: PM_{10} REDUCED TO 30 µg/m^3

A total of 56 980 YLL for all causes of chronic mortality are attributable to PM_{10} levels exceeding 30 µg/m^3 (Table 18). Also, 6627 YLL are due to lung cancer, 16 231 YLL to infarction and 1743 YLL to stroke. For acute effects, a total of 9670 YLL for all causes of mortality are attributable to PM_{10} levels exceeding 30 µg/m^3, 4737 YLL are due to cardiovascular causes and 1018 YLL are due to respiratory causes.

Table 18. Years of life lost: PM_{10} reduced to 30 µg/m^3

Causes of death	Males		Females		Total	
	No.	95% CrI	No.	95% CrI	No.	95% CrI
Chronic effects[a]						
Mortality (excluding accidents)	28 587	21 460–35 740	28 393	21 410–35 390	56 980	42 870–71 130
Lung cancer	4 703	3 316–6 092	1 924	1 364–2 485	6 627	4 687–8 575
Infarction	9 258	8 660–9 857	6 973	6 534–7 412	16 231	15 200–17 270
Stroke	723	438–1 010	1 020	623–1 419	1 743	1 061–2 429
Acute effects[b]						
Acute mortality (excluding accidents)	4 894	4 528–5 535	4 776	4 165–5 389	9 670	8 423–10 920
Cardiovascular diseases	2 313	1 914–2 714	2 424	2 016–2 833	4 737	3 930–5 546
Respiratory diseases	557	425–691	461	350–571	1 018	775–1 262

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.2.7 YLL: PM_{10} REDUCED TO 40 µg/m^3

A total of 24 856 YLL for all causes of chronic mortality are attributable to PM_{10} levels exceeding 40 µg/m^3 (Table 19). Also, 2949 YLL are due to lung cancer, 7150 YLL to infarction and 752 YLL to stroke. For acute effects, a total of 4107 YLL, for all causes of mortality, are attributable to PM_{10} levels exceeding 40 µg/m^3, 1991 YLL are due to cardiovascular causes and 444 YLL are due to respiratory causes.

Table 19. Years of life lost: PM_{10} reduced to 40 µg/m^3

Causes of death	Males		Females		Total	
	No.	95% CrI	No.	95% CrI	No.	95% CrI
Chronic effects[a]						
Mortality (excluding accidents)	12 376	8 677–16 050	12 480	8 778–16 160	24 856	17 460–32 200
Lung cancer	2 079	1 426–2 731	870	600–1 139	2 949	2 028–3 867
Infarction	4 068	3 756–4 377	3 083	2 855–3 308	7 150	6 612–7 684
Stroke	310	161–457	443	233–652	752	395–1 111
Acute effects[b]						
Acute mortality (excluding accidents)	2 064	1 742–2 384	2 043	1 728–2 354	4 107	3 471–4 739
Cardiovascular diseases	968	770–1 165	1 022	819–1 225	1 991	1 588–2 391
Respiratory diseases	239	168–310	205	144–266	444	312–575

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.2.8 YLL: PM_{10} REDUCED BY 10%

By reducing the mean concentration of PM_{10} by 10% in every city, a total of 17 646 years of life for all causes of chronic mortality can be saved (Table 20). The same reduction would lead to a gain of 2064 years of life for lung cancer, 5647 for infarction and 524 for stroke. For acute effects, the total gain would be 2961 years for all causes of mortality, 1421 years for cardiovascular causes and 302 years for respiratory causes.

Table 20. Years of life lost: PM_{10} reduced by 10%

Causes of death	Males		Females		Total	
	No.	95% CrI	No.	95% CrI	No.	95% CrI
Chronic effects[a]						
Mortality (excluding accidents)	8 876	6 340–11 400	8 769	5 606–1 1950	17 646	11 210–24 110
Lung cancer	1 473	1 022–1 926	591	409–773	2 064	1 432–2 699
Infarction	3 205	2 976–3 433	2 442	2 274–2 611	5 647	5 250–6 044
Stroke	218	139–297	306	198–415	524	337–712
Acute effects[b]						
Acute mortality (excluding accidents)	1 503	1 323–1 683	1 458	1 286–1 629	2 961	2 609–3 312
Cardiovascular diseases	693	580–807	728	613–844	1 421	1 192–1 652
Respiratory diseases	167	130–203	135	106–164	302	236–368

[a]Adults ≥30 years of age, risk based on $PM_{2.5}$ estimates;
[b]all ages.

3.3 PM_{10}: morbidity

3.3.1 CASES ATTRIBUTABLE TO PM_{10} REDUCED TO 20 µg/m^3

The reduction of PM_{10} concentration to 20 µg/m^3 could prevent 0.7% of the observable cardiac-related hospital admissions (809 cases), 1.3% of respiratory-related hospital admissions (990 cases), 31.7% of acute bronchitis cases in children under 15 years of age (38 342 cases) and 1.7% of chronic bronchitis cases in people older than 27 years of age (4321 cases). Also, 1259 days of bronchodilator usage in children and more than 800 000 in adults could be prevented, as could more than 500 000 extra days of LRS in children and almost 8 million in adults. Moreover, almost half a million RADs, 6 million MRADs and the loss of almost 2 million working days could be avoided (Table 21).

Table 21. Cases of morbidity attributable to levels of PM_{10} exceeding 20 µg/m^3

Causes of morbidity	Exposed population	Number of cases	Prevalence rate (%)	Attributable cases		Attributable proportion	
				No.	95% CrI	%	95% CrI
All ages							
Cardiac-related hospital admissions	8 950 587	113 772	1.3	809	472–1 143	0.7	0.4–1.0
Respiratory-related hospital admissions	8 950 587	69 630	0.8	990	728–1 252	1.4	1.0–1.8
Children up to 15 years of age							
Acute bronchitis	1 139 660	120 804	10.6	38 342	33 440–43 230	31.7	27.7–35–8
Asthma, 6–7 and 13–14 years of age[a]	26 567	2 833	10.9[b] 10.3[c]	1 259	837–1 685	–	–
LRS, 5–14 years of age[d]	762 522	117 639	15.0	512 680	414 400–611 100	–	–
Adults ≥15 years of age							
Chronic bronchitis, ≥27years of age	6 638 581	250 938	0.4	4 321	2 676–5 967	1.7	1.1–2.4
Asthma[a]	7 810 927	351 492	4.5	814 756	504 500–1 126 000	–	–
RADs 15–64 years og age[e]	5 968 996	317 578	5.2	495 067	471 900–495 000	–	–
WLDs, 15–64 years og age[e]	5 968 996	–	–	1 961 060	1 845 000–2 078 000	–	–
MRADs, 18–64 years of age[e]	5 734 129	–	–	5 863 881	5 439 000–6 289 000	–	–
LRS[d]	7 810 927	2 394 599	30.0	7 742 560	5 134 000–10 360 000	–	–

[a]Attributable cases are expressed in terms of days of bronchodilator usage;
[b]for 6–7 year olds, derived from aggregation of city-specific results;
[c]for 13–14 year olds, derived from aggregation of city-specific results;
[d]attributable cases are expressed in terms of days of extra symptoms;
[e]$PM_{2.5}$.

3.3.2 CASES ATTRIBUTABLE TO PM_{10} REDUCED TO 30 µg/m^3

The reduction of PM_{10} concentration to 30 µg/m^3 could prevent 0.4% of the observable cardiac-related hospital admissions (476 cases), 0.8% of respiratory-related hospital admissions (588 cases), 21.8% of acute bronchitis cases in children under 15 years of age (26 375 cases) and 1.1% of chronic bronchitis cases in people older than 27 years of age (2644 cases). Also, 753 days of bronchodilator usage in children and almost 500 000 in adults could be prevented, as could more than 300 000 extra days of LRS in children and around 4.7 million in adults. Moreover, about 300 000 RADs, 3.5 million MRADs and the loss of about a million working days could be avoided (Table 22).

Table 22. Cases of morbidity attributable to levels of PM_{10} exceeding 30 µg/m^3

Causes of morbidity	Exposed population	Number of cases	Prevalence rate (%)	Attributable cases		Attributable proportion	
				No.	95% CrI	%	95% CrI
All ages							
Cardiac-related hospital admissions	8 950 587	113 772	1.3	476	264–676	0.4	0.2–0.6
Respiratory-related hospital admissions	8 950 587	69 630	0.8	588	428–747	0.8	0.6–1.0
Children up to 15 years of age							
Acute bronchitis	1 139 660	120 804	10.6	26 375	22 450–30 290	21.8	18.6–25.1
Asthma, 6–7 and 13–14 years of age[a]	26 567	2 833	10.9[b] 10.3[c]	753	484–1 023	–	–
LRS, 5–14 years of age[d]	762 522	117 639	15.0	301 438	241 900–361 100	–	–
Adults ≥15 years of age							
Chronic bronchitis, ≥27years of age	6 638 581	250 938	0.4	2 644	1 600–3 694	1.1	0.6–1.5
Asthma[a]	7 810 927	351 492	4.5	497 114	300 000–695 400	–	–
RADs 15–64 years og age[e]	5 968 996	317 578	5.2	300 752	286 200–315 400	–	–
WLDs, 15–64 years og age[e]	5 968 996	–	–	1 220 027	1 145 000–1 295 000	–	–
MRADs, 18–64 years of age[e]	5 734 129	–	–	3 567 704	3 229 000–3 838 000	–	–
LRS[d]	7 810 927	2 394 599	30.0	4 724 037	3 067 000–6 391 000	–	–

[a]Attributable cases are expressed in terms of days of bronchodilator usage;
[b]for 6–7 year olds, derived from aggregation of city-specific results;
[c]for 13–14 year olds, derived from aggregation of city-specific results;
[d]attributable cases are expressed in terms of days of extra symptoms;
[e]$PM_{2.5}$.

3.3.3 CASES ATTRIBUTABLE TO PM_{10} REDUCED TO 40 µg/m^3

The reduction of PM_{10} concentration to 40 µg/m^3 could prevent 0.2% of the observable cardiac-related hospital admissions (176 cases), 0.3% of respiratory-related hospital admissions (225), 9.5% of acute bronchitis cases in children under 15 years of age (11 463) and 0.4% of chronic bronchitis cases for people over 27 years of age (1 114); 312 days of bronchodilator usage in children and more than 200 000 in adults, more than 100 000 extra days of LRS in children and almost 2 million in adults could be prevented; more than 100 000 of RADs, almost 1.5 million MRADs and the loss of half a million working days could be avoided (Table 23).

3.3.4 CASES ATTRIBUTABLE TO PM_{10} REDUCED BY 10%

The reduction of PM_{10} concentration by 10% in every city under study could prevent 0.1% of the observable cardiac-related hospital admissions (149 cases), 0.3% of respiratory-related hospital admissions (183 cases), 9.1% of acute bronchitis cases in children under 15 years of age (11 002 cases) and 0.3% of chronic bronchitis cases in people older than 27 years of age (771 cases). Also, 228 days of bronchodilator usage in children and almost 150 000 days in adults could be prevented, as could almost 100 000 extra days of LRS in children and almost 1.4 million days in adults. Moreover, almost 90 000 RADs, about a million MRADs and the loss of almost 350 000 working days could be avoided (Table 24).

Table 23. Cases of morbidity attributable to levels of PM_{10} exceeding 40 µg/m^3

Causes of morbidity	Exposed population	Number of cases	Prevalence rate (%)	Attributable cases		Attributable proportion	
				No.	95% CrI	%	95% CrI
All ages							
Cardiac-related hospital admissions	8 950 587	113 772	1.3	176	96–256	0.2	0.1–0.2
Respiratory-related hospital admissions	8 950 587	69 630	0.8	225	153–296	0.3	0.2–0.4
Children up to 15 years of age							
Acute bronchitis	1 139 660	120 804	10.6	11 463	9 449–13 460	9.5	7.8–11.1
Asthma, 6–7 and 13–14 years of age[a]	26 567	2 833	10.9[b] 10.3[c]	312	169–454	–	–
LRS, 5–14 years of age[d]	762 522	117 639	15.0	115 758	88 850–124 400	–	–
Adults ≥15 years of age							
Chronic bronchitis, ≥27years of age	6 638 581	250 938	0.4	1 114	595–1 629	0.4	0.2–0.6
Asthma[a]	7 810 927	351 492	4.5	208 493	110 400–305 900	–	–
RADs 15–64 years og age[e]	5 968 996	317 578	5.2	124 689	117 500–131 800	–	–
WLDs, 15–64 years og age[e]	5 968 996	–	–	532 252	493 700–570 600	–	–
MRADs, 18–64 years of age[e]	5 734 129	–	–	1 484 094	1 352 000–1 615 000	–	–
LRS[d]	7 810 927	2 394 599	30.0	1 981 293	1 156 000–2 800 000	–	–

[a]Attributable cases are expressed in terms of days of bronchodilator usage;
[b]for 6–7 year olds, derived from aggregation of city-specific results;
[c]for 13–14 year olds, derived from aggregation of city-specific results;
[d]attributable cases are expressed in terms of days of extra symptoms;
[e]$PM_{2.5}$.

Table 24. Cases of morbidity attributable to levels of PM_{10} being reduced by 10%

Causes of morbidity	Exposed population	Number of cases	Prevalence rate (%)	Attributable cases		Attributable proportion	
				No.	95% CrI	%	95% CrI
All ages							
Cardiac-related hospital admissions	8 950 587	113 772	1.3	149	87–211	0.1	0.1–0.2
Respiratory-related hospital admissions	8 950 587	69 630	0.8	183	135–231	0.3	0.2–0.3
Children up to 15 years of age							
Acute bronchitis	1 139 660	120 804	10.6	11 002	9 098–12 900	9.1	7.5–10.7
Asthma, 6–7 and 13–14 years of age[a]	26 567	2 833	10.9[b] 10.3[c]	228	153–303	–	–
LRS, 5–14 years of age[d]	762 522	117 639	15.0	93 817	75 850–111 800	–	–
Adults ≥15 years of age							
Chronic bronchitis, ≥27years of age	6 638 581	250 938	0.4	771	482–1 060	0.3	
Asthma[a]	7 810 927	351 492	4.5	145 588	91 030–200 400	–	–
RADs 15–64 years og age[e]	5 968 996	317 578	5.2	88 702	84 610–92 800	–	–
WLDs, 15–64 years og age[e]	5 968 996	–	–	345 703	32 550–365 900	–	–
MRADs, 18–64 years of age[e]	5 734 129	–	–	1 049 591	974 600–1 125 000	–	–
LRS[d]	7 810 927	2 394 599	30.0	1 383 508	925 400–1 843 000	–	–

[a]Attributable cases are expressed in terms of days of bronchodilator usage;
[b]for 6–7 year olds, derived from aggregation of city-specific results;
[c]for 13–14 year olds, derived from aggregation of city-specific results;
[d]attributable cases are expressed in terms of days of extra symptoms;
[e]$PM_{2.5}$.

3.4 Ozone: mortality

3.4.1 Attributable deaths and YLL

A total of 516 premature deaths from all causes (0.6% of total acute mortality), excluding accidents, equivalent to almost 6000 YLL, are attributable every year to ozone, as measured with the SOMO35 indicator (this is roughly equivalent to concentrations in excess of 70 µg/m^3). Also, 303 acute cardiovascular deaths (0.8%), equivalent to almost 3000 YLL, are attributable to the same levels of exposure (Tables 25 and 26).

As described in Chapter 2, a sensitivity analysis was carried out, using the SOMO0 indicator. An upper estimate was made of 1897 deaths (2.1%) for all causes, corresponding to more than 20 000 YLL, and 1112 acute cardiovascular deaths (2.8%), corresponding to more than 10 000 YLL (Tables 25 and 26).

Table 25. Deaths attributable to ozone: SOMO35 and SOMO0

Causes of death	Males		Females		Total			
	No.	95% CrI	No.	95% CrI	No.	95% CrI	Attributable proportion	95% CrI
SOMO35								
Acute mortality (excluding accidents)	247	186–308	269	205–334	516	390–641	0.6	0.4–0.7
Acute mortality, cardiovascular causes	132	118–145	172	154–189	303	273–334	0.8	0.7–0.9
SOMO0								
Acute mortality (excluding accidents)	908	686–1 130	989	754–1 225	1897	1440–2354	2.1	1.6–2.6
Acute mortality, cardiovascular causes	482	434–531	630	569–692	1112	1002–1223	2.8	2.6–3.1

Table 26. Years of life lost attributable to ozone: SOMO35 and SOMO0

Causes of death	Mlaes		Females		Total	
	No.	95% CrI	No.	95% CrI	No.	95% CrI
SOMO35						
Acute mortality (excluding accidents)	3 023	2 269–3 776	2 921	2 207–3 636	5 944	4 478–7 411
Acute mortality, cardiovascular causes	1 386	1 239–1 533	1 450	1 300–1 600	2 835	2 539–3 133
SOMO0						
Acute mortality (excluding accidents)	11 123	8 369–13 880	10 731	8 142–13 320	21 854	16 510–27 200
Acute mortality, cardiovascular causes	5 067	4 540–5 597	5 324	4 783–5 868	10 391	9 325–11 460

3.5 Ozone: morbidity

3.5.1 Cases attributable to ozone

A total of 1710 days of bronchodilator usage for asthma in children between 6–7 and 13–14 years of age and more than half a million days in adults are attributable to levels of ozone exceeding 70 µg/m^3 (SOMO35 indicator), as are more than 35 000 extra days of LRS in children, 228 respiratory-related hospital admissions for people older than 65 years of age (1% of the total cases) and more than a million MRADs in people 18–64 years of age (Table 27).

The same health outcomes were studied with a sensitivity analysis that used the SOMO0 indicator, and the following adverse health outcomes were estimated: 6364 days of bronchodilator usage for asthma in children between 6–7 and 13–14 years of age and almost two million days in adults, more than 135 000 extra days of LRS in children, 827 respiratory-related hospital admissions for people older than 65 years of age (3.5% of the total cases) and almost five million MRADs in people aged 18–64 years (Table 28).

Table 27. Cases of morbidity attributable to ozone: SOMO35

Causes of morbidity	Exposed population	Number of cases	Prevalence rate (%)	Attributable cases		Attributable proportion	
				SOMO35			
				No.	95% CrI	%	95% CrI
Children 5–14 years of age							
Asthma[a]	26 567	2 833	10.9[b] 10.3[c]	1 710	1 207–2 212	–	–
LRS[d]	762 522	117 639	15.0	36 580	22 160–51 070	–	–
Adults ≥ 15 years of age							
Hospital admission for respiratory diseases[e]	1 842 941	23 832	1.3	228	142–313	1.0	0.6–1.3
Asthma	7 810 927	351 492	4.5	500 443	311 800–689 100	–	–
MRADs[f]	5 734 129	–	–	1 290 504	987 500–1 595 000	–	–

[a]Attributable cases are expressed in terms of days of bronchodilator usage in children 6–7 and 13–14 years of age;
[b]for 6–7 year olds, derived from aggregation of city-specific results;
[c]for 13–14 year olds, derived from aggregation of city-specific results;
[d]attributable cases are expressed in terms of days of extra symptoms;
[e]for adults ≥ 65 years of age;
[f]for adults 18–64 years of age.

Table 28. Cases of morbidity attributable to ozone: SOMO0

Causes of morbidity	Exposed population	Number of cases	Prevalence rate (%)	Attributable cases		Attributable proportion	
				SOMO0			
				No.	95% CrI	%	95% CrI
Children 5–14 years of age							
Asthma[a]	26 567	2 833	10.9[b] 10.3[c]	6 364	4 554–8 173	–	–
LRS[d]	762 522	117 639	15.0	136 253	82 880–189 700	–	–
Adults ≥ 15 years of age							
Hospital admission for respiratory diseases[e]	1 842 941	23 832	1.3	827	517–1 135	3.5	2.2–4.8
Asthma	7 810 927	351 492	4.5	1 859 579	1 159 000–2 559 000	–	–
MRADs[f]	5 734 129	–	–	4 792 243	3 662 000–5 920 000	–	–

[a]Attributable cases are expressed in terms of days of bronchodilator usage in children 6–7 and 13–14 years of age;
[b]for 6–7 year olds, derived from aggregation of city-specific results;
[c]for 13–14 year olds, derived from aggregation of city-specific results;
[d]attributable cases are expressed in terms of days of extra symptoms;
[e]for adults ≥ 65 years of age;
[f]for adults 18–64 years of age.

Chapter 4. Conclusions

This chapter is divided into the following sections: Section 4.1 contains a short summary of the results of the health impact analyses; Section 4.2 interprets the results; Section 4.3 compares them with previous health impact estimates; Section 4.4 illustrates problems that arise from so-called vulnerable groups; Section 4.5 describes data quality and analytical uncertainties; and Section 4.6 illustrates policy implications.

4.1 Air pollution in 13 Italian cities: a large health impact

Air pollution has a large impact on health in Italian cities. During the period 2002–2004, 8220 deaths a year, on average, were attributable to PM_{10} concentrations above 20 $\mu g/m^3$. This is 9% of the mortality for all causes, excluding accidents, in the population older than 30 years of age. This figure is estimated by taking into account long-term effects. With the effects on mortality that take place in the short-term (within a week after exposure), the impact on mortality, again for PM_{10} above 20 $\mu g/m^3$, was 1372 deaths, 1.5% of the total mortality in the whole population (see Table 13).

The greater detail now available in the literature on the effects on mortality of PM allows a breakdown, by causes of death, of the impact on mortality (above 20 $\mu g/m^3$). The causes of death included in the long-term effects are lung cancer (742 cases a year), infarction (2562 cases a year) and stroke (329 cases a year). Acute effects include cardiovascular causes (843 cases a year) and respiratory causes (186 cases a year) (see Table 13).

Although mortality is the most severe outcome, large numbers of cases attributable to PM were estimated for many other outcomes of different severities. These include morbidity in children and adults (such as bronchitis, asthma and respiratory symptoms), hospital admissions for cardiac disease and respiratory conditions, and ill health that results in restricted activity and also in a loss of working days. For Italian cities, these effects are sizeable, and the results are in line with those obtained in analogous impact assessments in Europe and the Americas. Concentrations measured in Italian cities during the years 2002–2004 were above the European average (Medina et al., 2005) and so were, proportionately, the effects on health.

The impacts of PM and ozone on all the health outcomes represent important public health issues. The burden of disease is great at the individual and family level, among adults and children, and includes premature death, and chronic and acute disease, such as cancer, bronchitis, asthma and the prevalence of respiratory symptoms. The burden on society is also great: loss of life and a significant erosion of life expectancy, and the loss of economic productivity due to mild and severe impairments. Finally, it is a great burden on health care systems, in terms of the costs of thousands of hospital admissions.

As the implications for health are considerable, adequate countermeasures are warranted, as discussed in Section 4.6. The next sections discuss the strength and weaknesses of the findings of the present study.

4.2 Interpreting "impacts"

What is the correct interpretation of the notion of health impact? Strictly speaking, an impact estimate consists of the difference between the observed number of deaths and the number of deaths that would be observed if the mortality rates predicted by the concentration–response models at selected counterfactuals prevailed. However, the impact is more commonly described as the mortality that would be avoided if concentrations were reduced to the counterfactual. The equivalence of the two definitions depends mainly on the relationship between cause and effect (causality) and the relationship with time (temporality).

The causality of the association observed between air pollution and adverse effects on health is one of the main underlying assumptions used in estimating impacts. Epidemiological studies show a consistently positive relationship between exposure to outdoor air pollution and adverse effects on health, but they alone do not prove conclusively the causality of the association. Results from toxicological studies, however, corroborate the epidemiological evidence and, although the underlying biological mechanisms of PM-related effects are not fully understood, there is now strong support for the biological plausibility of the associations observed. For these reasons, the causality of changes in ambient air pollution and the adverse effects on health can be linked confidently.

Temporality is another important factor. Having established the causal association between air pollution and adverse effects on health, it is correct to assume that reductions in average concentrations produce the estimated gains in health. However, it is not possible to determine exactly the period of time over which these gains are achieved. This is especially uncertain for long-term mortality, because the induction time is not known. It is likely, however, that at least part of long-term excess mortality involves long-latency associations – for example, in the case of mortality from lung cancer. For this part, health benefits would follow reductions in concentrations with a substantial delay, while they would be almost immediate for acute effects, which represent the majority of the effects on health considered. In any case, observations confirm that health gains do take place as a result of air-pollution abatement: in a recent study, the reduction in fine PM achieved in six American cities was followed by a reduction in mortality, by an amount even greater than would be expected by impact-assessment predictions (Laden et al., 2006).

These health gains have a bearing on long-term mortality, which accounts for the largest share of the impact on mortality. In parallel with the issue of long-term mortality, the issue of the health benefits linked to short-term effects raises some questions, and the scientific debate about the prediction of mortality due to the effects of PM_{10} is intense (Dominici, 2004). In particular, harvesting or mortality displacement, might play a role in determining the real public health significance of the effects of air pollution on mortality. Harvesting is a short advancement of the death of the frailest individuals, who would have died within a short period of time, such as a few days, of a peak episode of air pollution. If this is the case, mortality rates should increase immediately after a day with a high level of pollution and decrease thereafter, as the pool of the susceptible people replenishes. This consideration, however, is inconsistent with the increasing amount of evidence emerging from recent time-scale analyses, which found air pollution associated more strongly with longer-term variations in mortality rates (Zeger, Dominici & Samet, 1999; Schwartz, 2000, 2001; Dominici et al., 2003b; Fung et al., 2003; Zanobetti & Schwartz, 2003b). Thus, while

harvesting cannot be ruled out totally, its importance is probably limited. For the same reason – that is, for the partial, if limited, overlap between the short-term and long-term effects – it is incorrect to add long-term and short-term effects on mortality (Künzli et al., 2001; Martuzzi, 2001; Crosignani et al., 2003; Hurley et al., 2005). A model for estimating acute and chronic effects jointly has been proposed recently by Eftim & Dominici (2005).

The counterfactual levels chosen for the comparison – that is, the reference levels chosen – are also important when evaluating effects on health. The results in the present report include impacts estimated with different counterfactuals – 20, 30 and 40 µg/m^3 of PM_{10} – to provide, respectively, scenarios of compliance with the European Union limits to be reached by 2010, with the reference level used in the previous impact assessment (for comparison), and with the European Union limits to have been reached by 2005. In addition, a proportional reduction of 10% of the average value of PM_{10} concentration in every city was chosen as a policy-based scenario, to evaluate the health gain of limited, more easily achievable pollution reductions. The impacts vary proportionately, due to the linear association between concentrations and adverse effects on health. In any case, as no threshold is known for the adverse effects on health of PM_{10}, any additional reduction in concentrations, down to the natural background level of 6–7 µg/m^3 of PM_{10}, would result in health gains.

4.3 Comparison with previous impact estimates

The results from Chapter 3 update those of the impact assessment previously carried out for the eight largest Italian cities (Martuzzi et al., 2002). For 1998, 3472 deaths in the population older than 30 years of age were estimated as attributable to PM_{10} above the level of 30 µg/m^3. The present study covers 13 cities, but the equivalent figure for the same eight cities for the period 2002–2004 is 4514 deaths. The comparison of the two estimates must be done by taking the following three factors into consideration.

1. The mortality and population profiles were similar, and they do not influence differences in the final result.

2. An updated, considerably larger and more reliable RR was applied in the analyses (RR =1.060 per 10-µg/m^3 increase in PM_{10} concentration, compared with RR = 1.026 per 10-µg/m^3 increase in PM_{10} concentration of the previous study), which strongly increases the impact estimates.

3. The concentrations recorded in the two periods are difficult to compare: first, corrections for gravimetric monitoring stations were needed; second, data from some stations used in 1998 were not available. Restricting the data to concentration measurements from the most comparable stations, there is a decrease of about 6.5 µg/m^3 from 1998 to the period 2002–2004.

This last factor (in contrast with the previous one) decreases the impact estimate, although the comparability of the concentrations in the two periods is highly uncertain. Thus, the increase in the estimates of the impact on mortality from 1998 to the period 2002–2004 is due to the stronger evidence available now on the adverse effects on health; this alone outweighs the decrease in concentration of PM_{10} due to the use of more reliable air quality data from the national monitoring network.

The pollutant of choice for assessing the health impact of air pollution is still PM_{10}: epidemiological evidence continues to grow, with new studies using PM_{10} as the exposure indicator for PM, and most monitoring data are presently based on measurements of PM_{10}. Given the very high correlation between PM_{10} and other air pollutants, including finer particles, PM_{10} is considered a good measure of the complex mix of particles and dust that originate from fuel combustion in vehicles and power generators. The health impact of PM_{10} therefore reflects possible effects due to other correlated pollutants or their interactive effects. However, it is desirable to have systematic measures of the concentrations of finer particles, because the effects on health of $PM_{2.5}$ are presently well known, and fine particles can be more easily traced in terms of emission sources: $PM_{2.5}$, for example, correlates more closely with motor vehicle traffic than does PM_{10}. It is not by chance that $PM_{2.5}$ has been routinely monitored in several European and North American countries in recent years.

Unlike the previous assessment, the present one takes the impact of ozone into consideration. Ozone is a pollutant of growing concern, especially in southern European countries. Observed concentrations of this pollutant are on the increase, and its adverse effects on health are being more firmly established. Using the standard SOMO35 indicator for concentrations, described in Chapter 2, an estimated 516 extra deaths yearly in Italian cities (0.6% of the total) are attributed to ozone, with a loss of 5944 years of life. Though its impact is smaller than that of PM, the impacts of the two pollutants can be summed, because they are uncorrelated and can be used as independent indicators of air quality. Thus, it can be assumed that policies that result in the abatement of PM and ozone would produce a gain estimated by the sum of their respective health impacts.

4.4 Vulnerable subgroups

When evaluating the impacts, particular attention should be paid to vulnerability to air pollution. Current scientific evidence indicates that within any large population there is a wide range of susceptibility, and some subjects are more vulnerable than others to PM_{10}. This susceptibility entails an increased risk of mortality and morbidity for people with pre-existing heart and lung disease, especially among the elderly and very young (EPA, 2005a). Children with asthma are also more susceptible to ambient PM_{10} (Pope & Dockery, 1992; Boezen et al., 1999; EPA, 2005a; WHO Regional Office for Europe, 2005a).

To some extent, the special vulnerability of certain groups is taken into account in assessing the impact of air pollution: age groups where risks are known to differ from those of the general population are included as end-points of interest. The impact on bronchitis, for example, is estimated for acute bronchitis in those younger than 15 years of age, and for chronic bronchitis it is estimated in those older than 27 years of age; also, specific age groups are used to estimate asthma and respiratory symptoms. The likely presence of more vulnerable subgroups, however, is not captured by available concentration–response coefficients, which in many cases apply to the average population. In these cases, the overall impact of air pollution may be slightly underestimated but, more important, the impact within vulnerable subgroups will be seriously underestimated. Apart from the inaccurate estimation, the failure to recognize adverse effects in vulnerable subgroups is unfortunate, as it prevents the adoption of targeted protective action.

Socioeconomic status is another important characteristic that may determine differential susceptibility. Socioeconomically disadvantaged communities and individuals are exposed to higher levels of environmental risk factors than the average population. Similarly, socioeconomically disadvantaged communities do not have the same degree of protection

from environmental and health hazards as do average populations. Also, socioeconomically deprived people may be at greater risk because of more limited access to health care.

A recent review by O'Neill and colleagues (2003) suggested that socioeconomic status and its association with poor health may be partially explained by related differences in exposure. Using traffic density on the level of a census block group in California, Gunier and colleagues (2003) reported that low-income people and non-white children may experience greater exposure to vehicle emissions than the average population. Possible reasons for the role of socioeconomic status in different distributions of air pollution include poorer housing quality, ethnic discrimination and class bias in land-use decisions (O'Neill et al., 2003). Proximity to areas of dense traffic depresses property values, and the lower costs of dwellings are likely to attract people of lower socioeconomic status who, as a consequence, frequently have higher levels of exposure. Moreover, socioeconomic status probably plays a role in the kind of transport people use, as stated in a recent WHO Regional Office for Europe report (Krzyzanowski, Kuna-Dibbert & Schneider, 2005).

Recent evidence suggests that people with lower socioeconomic status are more susceptible to the effects of PM_{10} on mortality, and this evidence is growing (O'Neill et al., 2003). Positive associations were shown in cohort studies (Krewski et al., 2000b, 2003; Pope et al., 2002) and to a lesser extent in time-series studies (Hoek et al., 2002; Jerrett et al., 2004). Results from the reanalysis of the Harvard Six Cities Study (Dockery et al., 1993) and the ACS study (Pope et al., 1995) of particulate air pollution and mortality showed a gradient of risks that decrease as the level of education increases (Krewski et al., 2000a) (Table 29). As to all causes of mortality in the long term, the present report applied the average risk coefficient (RR = 1.06; 95% CI: 1.02–1.11) of the most recent Pope analysis, as suggested by the UNECE Task Force on Health (UNECE, 2004) and as adopted by the CAFE programme (Holland et al., 2005). This may cause the impacts on long-term mortality to be underestimated, since the population followed by Pope and colleagues had an educational level above average (Krewski et al., 2000a), so the risks estimated may be smaller than those borne by the average population.

Table 29. Relative risk by educational status per 10-μg/m^3 increase in $PM_{2.5}$ concentration

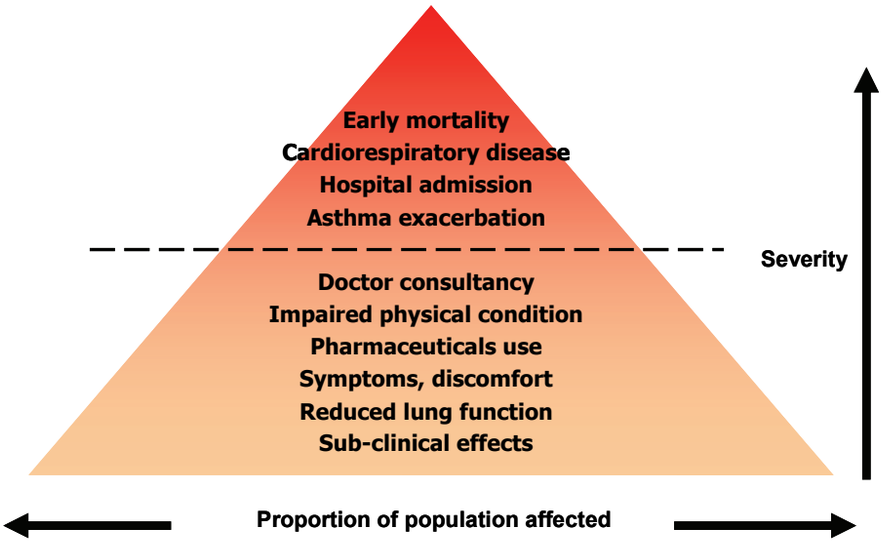
Cause	Education		
	Less than High School	High School	More than High School
All causes of mortality	1.13 (95% CI: 1.07–1.2)	1.09 (95% CI: 1.03–1.15)	1.02 (95% CI: 0.71–1.07)
Cardiopulmonary disease	1.17 (95% CI: 1.08–1.27)	1.13 (95% CI: 1.04–1.22)	1.05 (95% CI: 0.8–1.13)
Cardiovascular disease	1.17 (95% CI: 1.07–1.28)	1.14 (95% CI: 1.05–1.25)	1.09 (95% CI: 1.02–1.17)
Respiratory disease	1.13 (95% CI: 0.48–1.41)	1.06 (95% CI: 0.62–1.31)	0.65 (95% CI: 0.2–1.01)
Lung cancer	1.15 (95% CI: 0.94–1.4)	1.14 (95% CI: 0.77–1.37)	0.65 (95% CI: 0.23–0.71)

Source: Data elaborated from Krewski et al. (2000a).

The estimated impacts are likely to provide an incomplete picture of the total burden of disease. Other health end-points are also affected that are not included in the assessment, because the risks are not estimated reliably. Infant mortality, for example, in not included, due to the difficulties of extrapolating risks estimated in studies carried out in Latin America and Asia (the question of extrapolation is discussed in Section 4.5). Also, other health end-points are mild, difficult to measure and with positive, but unquantified risks. As shown in Fig. 5, the hierarchical adverse effects of air pollution on health can be described by a pyramid. As effects decrease in severity, the proportion of the population affected increases. The available evidence allows the estimation of the impact of many severe or relatively severe outcomes, but not of those at the bottom of the pyramid. Though it is not included in the estimates, the

impact of air pollution, in terms of mild symptoms or sub-clinical effects, is probably non-negligible, given that it affects large portions of the population.

Fig. 5. Pyramid of the adverse effects on health attributable to air pollution



4.5 Data quality and analytical uncertainties

Italian data on demographics, health statistics and air quality were used in the present report. Mortality data were coded centrally by ISTAT, procedures were standardized nationally, and completeness and accuracy were acceptable . Hospital personnel (usually the doctor in charge of the patient) carried out the coding of hospital admissions records; the reliability of the coding is thus lower. For example, the rates of infarction given by all ICD codes 410–414 are comparable between Turin and Milan, but in Milan the use of ICD code 410 for infarction (almost all acute) is almost double that of Turin, where codes ranging from 411 to 414 (subacute or chronic) are more frequent (Cadum, 2006).

The availability of environmental data was mixed. While the overall quality was very good for ozone, and hourly records were available for all the cities in the study, the same was not the case for PM_{10}. The validation process, described in Subsection 2.6.1, could be only partially applied to PM_{10}: when hourly data were not available, annual averages were extrapolated from air quality reports written by regional environmental authorities, and they were calculated by a different validation process. An additional source of uncertainty is the application of conversion and correction coefficients used for homogeneous concentration values – that is, gravimetric concentrations on the $PM_{2.5}$ scale. This source of uncertainty depends essentially on the quality of the network of monitoring stations. The ratio $PM_{2.5}/PM_{10} = 0.7$ is presently the best choice possible, even though it does not account for regional variations in the composition of PM.

As a result of changes in ambient pollution concentrations, some questions and uncertainties exist about the use of epidemiological evidence to predict effects on health. However, as new research findings became available after the publication of the first WHO Regional Office for Europe report (Martuzzi et al., 2002), some of these uncertainties have been removed.

One important question is about the choice and application of the risk coefficients used in health impact assessment studies. These are often derived from epidemiological studies carried out in locations and populations other than those considered for the impact assessment. For example, evidence on long-term effects is largely based on studies in the United States, transferred to European populations. Such extrapolations may involve some approximations, as some characteristics differ between the two settings, such as weather, smoking status, socioeconomic conditions, access to health care, diet, time spent outdoors and housing characteristics. Substantial bias is unlikely, however, because the original evidence is itself based on studies of a mix of relatively heterogeneous observations and because the composition of ambient air pollution is comparable. Extrapolating from one population to another is a consolidated procedure applied in health impact assessment studies, recently adopted to an even more heterogeneous context – the global context (Cohen et al., 2004). For acute mortality, however, the evidence was often based on studies carried out in European or Italian cities, where the uncertainty in the extrapolation becomes negligible.

A second consideration in the assessment of exposure is fixed-site monitoring stations. Data from these are generally used to calculate an average concentration value, which is used as an approximation of exposure. Factors likely to affect individual exposures, such as personal time activity patterns, are not taken into account by these data. This may introduce a non-differential error, leading to dilution or underestimation of the impact of air pollution. Risk estimates used in health impact assessment studies, however, are generated in observational studies that are also based on the same approximation. In epidemiological studies, risks are measured by comparing different groups of people (or the same people in different time periods) who experience different average concentrations. This may involve a random, non-differential error of unknown magnitude, which may in turn produce an underestimation of the risk. Applying these risks to impact assessments that use average concentrations provides consistency between estimated risks and estimated impacts on health from air pollution, involving possibly underestimations of both.

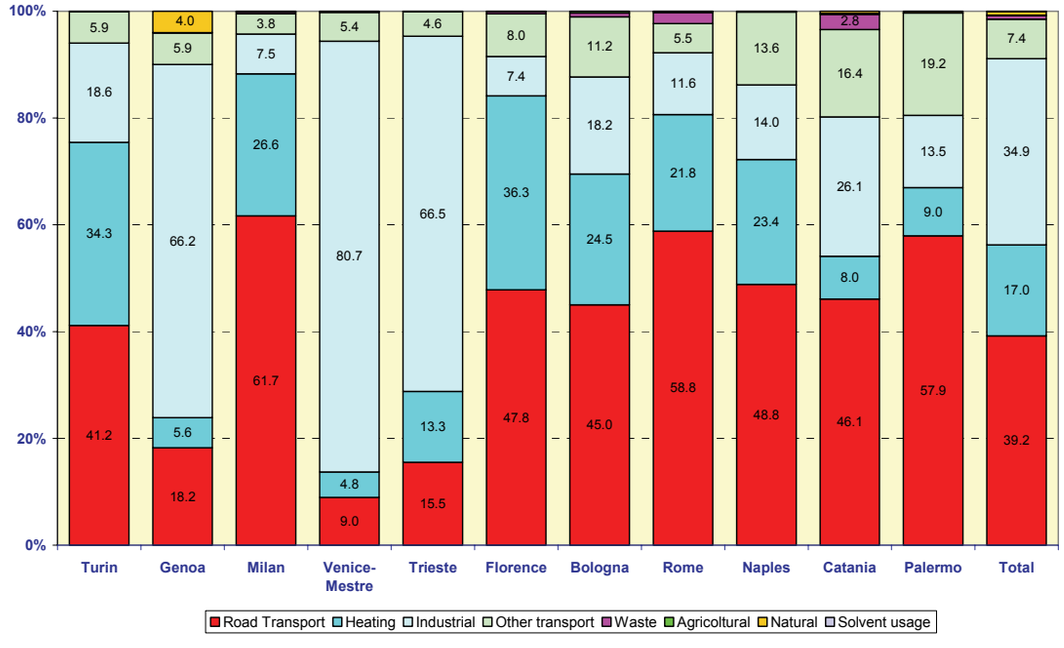
4.6 Policy response

The magnitude of the health impact of air pollution estimated for the 13 Italian cities underscores the need for urgent action to reduce its burden in these cities and, likely, in many others. Compliance with EU legislation results in substantial savings, by avoiding ill health, and it is important that the limits on PM_{10} introduced in Directive 1999/30/EC (EU, 1999) are met and that they should not be relaxed (a position recently taken by a large group of researchers in the field (Brunekreef, 2005)). Italy, however, is one of the EU Member States where this may be a challenge. In 2005, in Italy, many of the major cities had reached the allowed 35 days in excess of 50 $\mu g/m^3$ of PM_{10} by the end of March; only some cities are in compliance with the annual average of 40 $\mu g/m^3$ of PM_{10}; none is in compliance with the average value of 20 $\mu g/m^3$ of PM_{10}, which is the limit to be reached in 2010. Within Europe, in general, the concentrations of PM_{10} decreased substantially between 1997 and 1999, but this decline stopped in more recent years. Instead, there was a steady increase between 2001 and 2003. However, on average, levels in 2004 were lower than in 2003.

Then how should the goals set by European Commission legislation be achieved? For one thing, information on sources can be used to identify the most profitable area of policy response for achieving gains in health. The data in the present report suggest that substantial gains can be achieved through policies aimed mainly at reducing emissions from two sources: urban transport and energy production. A recent report prepared by APAT (Italian Agency for the Protection of the Environment and for Technical Services, 2005) showed that PM_{10} from road transport (excluding resuspended dust) represents the main source of total primary

emissions in Italian metropolitan areas (see Fig. 6; Padua and Verona are excluded). The contribution of road transport is between 40% and 60% (average 51%) in all metropolitan areas, except Venice-Mestre, Trieste and Genoa, where there are large industries or harbours (or both) and where industrial activities account for most (from 66% to 81%) of total emissions. On average, about 48% of PM_{10} primary emissions originate from industrial activities in northern metropolitan areas (Turin, Genoa, Milan, Venice-Mestre and Trieste), as compared with 15% in central and southern areas (Florence, Bologna, Rome, Naples, Catania and Palermo). Heating is responsible for about 17% of PM_{10} primary emissions in northern metropolitan areas, 27.5% in central cities and 13.5% in the southern areas.

Fig. 6. Percentages of PM_{10} primary emissions by source and metropolitan area



Source: Data elaborated from APAT (2005).

Identifying specific policies for reducing concentrations is not an easy task. With reference to emissions of PM, health gains can be obtained by reducing concentrations through different strategies; since the association between air pollution and its adverse effects on health is linear and has no threshold, the effects of air pollution will decrease in proportion to the average concentration, for all health outcomes. So different interventions that produce the same yearly average will provide the same health benefits. In principle, this suggests that a variety of policy options are available. However, empirical data show that measures that reduce peak concentrations also produce reductions in average concentrations (Cirillo, 2003). Thus emissions from the main urban sources, notably from motor vehicles, must be reduced substantially, through policies that aim to contain private motorized transport and promote public transport, cycling and walking. In Italian cities, special attention should also be paid to the contribution from motorcycles, especially those with two-stroke engines; in a pilot study in Rome, these have been estimated to contribute sizeably to emissions (Faberi, Martuzzi & Pirrami, 2004). Within the general policy goal of reducing emissions, attention should be given to local circumstances. In particular, PM_{10} concentrations observed in the present study were high in northern cities (50 $\mu g/m^3$), compared with urban areas located in central (43 $\mu g/m^3$) and southern Italy (35 $\mu g/m^3$). These differences are likely to be due mainly to

differences in transport, industrial activities, and heating-related emissions at the city level and also at the regional level – together with climatic factors. For example, the cities of the Po-Venetian Plain (Verona, Milan and Padua) have high concentrations of PM_{10} (59 $\mu g/m^3$ annual average for the period 2002–2004), due to intense local urban traffic, intense regional traffic and intense industrial activities, combined with climatic conditions that limit the dispersion of pollutants. Under these circumstances, action taken by one municipality to reduce, for example, emissions from motor vehicles is likely to have modest results. Instead policy action at the regional level may be needed to achieve substantial gains in reducing concentrations of air pollutants and in improving health.

Similar considerations apply to ozone. Ozone contributes a considerable additional impact on health, although its impact is smaller than the one for PM. Repeated epidemiological studies have demonstrated that risks to health increase linearly with ozone concentration and are observed not only on days with ozone peaks, but are also observed on non-peak days. For this reason, as with PM, strategies for reducing ozone levels should target not only peak days but should also target average concentrations. Given that precursors of ozone are produced mainly by combustion processes, preventive action, again, should target emissions from transport and, where relevant, industry.

Policy directed at the traffic sector is particularly appropriate for several other reasons. Apart from the importance of traffic emissions of primary PM, other emissions from road transport (such as resuspended road dust and wear of tyres and brake linings) are the main source of the coarse fraction of PM ($PM_{10–2.5}$).

Technological advances and stricter emission standards will decrease emissions per kilometer driven. However, the growing demand for transport, the increasing number of diesel cars, the large number of short trips and traffic congestion may outweigh the benefits derived from these improvements. Motor vehicle traffic will thus contribute increasingly to air pollution. Also, alternative vehicle technologies are unlikely to gain a substantial share of the market and are unlikely to have a significant impact on air quality (Krzyzanowski, Kuna-Dibbert & Schneider , 2005).

Finally, action in the transport sector is an attractive policy option, considering the co-benefits to be achieved by measures that aim to reduce air pollution emissions. Restrictions on private motor vehicle traffic would result in a number of health benefits – for example, through the reduced impact of road accidents and exposure to noise and through the possible increase of walking and cycling and psychosocial effects, such as social severance of non-motorized groups, which among others includes the elderly. In the case of road accidents, the number of fatal injuries recorded among residents of the 13 Italian cities in 2001 is of the same order of magnitude as the short-term impact of PM_{10}: the 844 acute deaths per year attributable to PM_{10} concentrations above 30 $\mu g/m^3$ are comparable to the 781 traffic fatalities (ISTAT & ACI, 2003), to which a large number of non-fatal injuries must be added.

Indeed, methods to quantify the effect on health of broad policies, rather than individual risk factors (such as air pollution), are of growing interest in the fields of environment and health. The methods for an integrated assessment of urban transport, in particular, was addressed by the Health Effects and Risks of Transport Systems (HEARTS) project, funded by the European Commission and carried out by a European research consortium coordinated by WHO (Mudu et al., to be published).

References

Abbey DE et al. (1993). Long-term ambient concentrations of total suspended particulates, ozone, and sulfur dioxide and respiratory symptoms in a nonsmoking population. *Archives of Environmental Health*, 48(1):33–46.

Abbey DE et al. (1995a). Estimated long-term ambient concentrations of PM10 and development of respiratory symptoms in a nonsmoking population. *Archives of Environmental Health*, 50(2):139–152.

Abbey DE et al. (1995b). Chronic respiratory symptoms associated with estimated long-term ambient concentrations of fine particulates less than 2.5 microns in aerodynamic diameter (PM2.5) and other air pollutants. *Journal of Exposure Analysis and Environmental Epidemiology*, 5(2):137–159.

Abbey DE et al. (1999). Long-term inhalable particles and other air pollutants related to mortality in nonsmokers. *American Journal of Respiratory and Critical Care Medicine*, 159(2):373–382.

Abt Associates (2000). *The particulate-related health benefits of reducing power plant emissions*. Bethesda, MD, Abt Associates Inc. (http://www.abtassociates.com/reports/particulate-related.pdf, accessed 16 April 2006).

Adams PF, Hendershot GE, Marano MA (1999). *Current estimates from the National Health Interview Survey*, 1996. Hyattsville, MD, U.S. Department of Health and Human Services, Public Health Service, National Center for Health Statistics.

AIRNET Work Group 2 (2004). Air pollution and the risks to human health: epidemiology. Utrecht, Institute for Risk Assessment Sciences (http://airnet.iras.uu.nl/products/pdf/airnet_wg2_epidemiology_report.pdf, accessed 19 April 2006).

AIRNET Work Group 3 (2004). Air pollution and the risks to human health: a toxicological perspective. Utrecht, Institute for Risk Assessment Sciences (http://airnet.iras.uu.nl/products/pdf/airnet_wg3_toxicology_report.pdf, accessed 19 April 2006.

Amann M et al. (2005). *Baseline Scenarios for the Clean Air for Europe (CAFE) Programme. Final Report.* Laxenburg, Austria, International Institute for Applied Systems Analysis.

Anderson HR et al. (1996). Air pollution and daily mortality in London: 1987–92. *British Medical Journal*, 312(7032):665–669.

Anderson HR et al. (2001). Particulate matter and daily mortality and hospital admissions in the west midlands conurbation of the United Kingdom: associations with fine and coarse particles, black smoke and sulphate. *Occupational and Environmental Medicine*, 58(8):504–510.

Anderson HR et al. (2004). *Meta-analysis of time-series studies and panel studies of particulate matter (PM) and ozone (O_3): report of a WHO task group***.** Copenhagen, WHO Regional Office for Europe (document EUR/04/5042688; http://www.euro.who.int/document/E82792.pdf, accessed 15 April 2006).

Atkinson RW, APHEA 2 Project (2004). Acute effects of air pollution on admissions: reanalysis of APHEA 2. *American Journal of Respiratory and Critical Care Medicine*, 169(11):1257–1258 (http://ajrccm.atsjournals.org/cgi/content/full/169/11/1257, accessed 16 April 2006).

Atkinson RW et al. (2001). Acute effects of particulate air pollution on respiratory admissions: results from APHEA 2 project. Air Pollution and Health: a European approach. *American Journal of Respiratory and Critical Care Medicine*, 164(10 Pt 1):1860–1866 (http://ajrccm.atsjournals.org/cgi/content/full/164/10/1860, accessed 16 April 2006).

Bayer-Oglesby L et al. (2005). Decline of ambient air pollution levels and improved respiratory health in Swiss children. *Environmental Health Perspectives*, 113(11):1632–1637.

Bell ML, Dominici F, Samet JM (2005). A meta-analysis of time-series studies of ozone and mortality with comparison to the national morbidity, mortality, and air pollution study. *Epidemiology*, 16(4):436–445.

Bell ML et al. (2004). Ozone and short-term mortality in 95 US urban communities, 1987–2000. *The Journal of American Medical Association*, 292(19):2372–2378.

Bergendorff S (2003). Sickness absence in Europe – a comparative study. In: *Fourth international research conference on social security, Antwerp, 5-7 May 2003 – "Social security in a long life society"*. Stockholm, The National Social Insurance Board (http://www.issa.int/pdf/anvers03/topic5/2bergendorff.pdf, accessed 17 October 2005).

Biggeri A, Bellini P, Terracini B (2001). Metanalisi Italiana degli studi sugli effetti a breve termine dell'inquinamento atmosferico [Meta-analysis of the Italian studies on short-term effects of air pollution]. *Epidemiologia e Prevenzione*, 25(S2):1–71 (in Italian).

Biggeri A, Bellini P, Terracini B (2004). Metanalisi Italiana degli studi sugli effetti a breve termine dell'inquinamento atmosferico 1996-2002 [Meta-analysis of the Italian studies on short-term effects of air pollution – MISA 1996–2002]. *Epidemiologia e Prevenzione*, 28(S4–5):4–100 (in Italian).

Bliksvaer T, Helliesen A (1997). *Sickness absence: a study of 11 LES countries.* Oslo, NOVA – Norwegian Social Research (http://www.lisproject.org/publications/leswps/leswp3.pdf, accessed January 2006).

Boezen HM et al. (1999). Effects of ambient air pollution on upper and lower respiratory symptoms and peak expiratory flow in children. *Lancet*, 353(9156):874–878.

Boezen M et al. (1998). Peak expiratory flow variability, bronchial responsiveness, and susceptibility to ambient air pollution in adults. *American Journal of Respiratory and Critical Care Medicine*, 158(6):1848–1854.

BRACE [online database] (2004). Rome, APAT (http://www.brace.sinanet.apat.it/web/struttura.html, accessed 15 January 2005) (in Italian).

Braun-Fahrlander C et al. (1997). Respiratory health and long-term exposure to air pollutants in Swiss schoolchildren. SCARPOL Team. Swiss Study on Childhood Allergy and Respiratory Symptoms with Respect to Air Pollution, Climate and Pollen. *American Journal of Respiratory and Critical Care Medicine*, 155(3):1042–1049.

Bremner SA et al. (1999). Short-term associations between outdoor air pollution and mortality in London 1992–4. *Occupational and Environmental Medicine*, 56(4):237–244.

Brook RD et al. (2004). Air pollution and cardiovascular disease: a statement for healthcare professionals from the Expert Panel on Population and Prevention Science of the American Heart Association. *Circulation*, 109(21):2655–2671.

Brunekreef B (2005). *Letter to Committee for Environment, Public Health and Food Safety* (http://www.iras.uu.nl/EPletter/EUletterFINAL.pdf, accessed 15 November 2005).

Burnett RT et al. (1995). Associations between ambient particulate sulfate and admissions to Ontario hospitals for cardiac and respiratory diseases. *American Journal of Epidemiology*, 142(1):15–22.

Burnett RT et al. (1997a). The role of particulate size and chemistry in the association between summertime ambient air pollution and hospitalization for cardiorespiratory diseases. *Environmental Health Perspectives*, 105(6):614–620.

Burnett RT et al. (1997b). Association between ambient carbon monoxide levels and hospitalizations for congestive heart failure in the elderly in 10 Canadian cities. *Epidemiology*, 8(2):162–167.

Burnett RT et al. (1999). Effects of particulate and gaseous air pollution on cardiorespiratory hospitalizations. *Archives of Environmental Health*, 54(2):130–139.

Burnett RT et al. (2000). Association between particulate- and gas-phase components of urban air pollution and daily mortality in eight Canadian cities. *Inhalation Toxicology*, 12(S4):15–39.

Cadum E (2006). Personal communication.

Cadum E et al. (1999). Inquinamento atmosferico e mortalità giornaliera a Torino, 1991-1996 [Air pollution and daily mortality in Turin, 1991–1996]. *Epidemiologia e Prevenzione*, 23 (4):268–276 (in Italian).

CAFE Working Group on Particulate Matter (2004). *Second position paper on particulate matter – final draft.* Brussels, Directorate General for Environment, European Commission. (http://europa.eu.int/comm/environment/air/cafe/pdf/working_groups/2nd_position_paper_pm.pdf, accessed 1 April 2005).

Cassadou S, Chardon B, D'Helf M (2004). Vague de chaleur de l'été 2003: relations entre température, pollution atmosphérique et mortalité dans neuf villes françaises [Heat waves in summer 2003: relationships between temperature, air pollution and mortality in nine French cities]. Rapport d'ètude. Saint Maurice, Institute de Veille Sanitaire [Institute of Health Surveillance] (Rapport d'étude 2004) (in French).

Chellini E, ed. (2005). *Inquinamento atmosferico urbano e salute in Toscana [Urban air pollution and health in Tuscany region]*. Florence, Centre for Oncological Study and Prevention, Tuscany Health Service, Health Agency of Tuscany Region (in Italian).

Cifuentes L et al. (2001). Assessing the health benefits of urban air pollution reductions associated with climate change mitigation (2000–2020): Santiago, Sao Paulo, Mexico City, and New York City. *Environmental Health Perspectives*, 109(Suppl 3)419–425.

Cirillo M (2003). Piani e programmi: obiettivi, scenari, interventi e risorse finanziarie [Plans and programmes: targets, scenarios, interventions and financial resources]. In: *La valutazione e gestione della qualità dell'aria alla luce del nuovo quadro normativo [Air quality assessment and management after the new regulations]*. Bologna, 27 March 2003 (http://www.liberiamolaria.it/download/convegno_cirillo.pdf, accessed 30 April 2006) (in Italian).

Cislaghi C (2005). GIS8 – Atlante italiano di mortalità 1981–2001 [GIS8 – Italian Mortality Atlas 1998–2001]. Milan, CILEA (in Italian).

Cohen AJ et al. (2004). Urban air pollution. In: Ezzati M et al., eds. Comparative quantification of health risks: global and regional burden of disease attributable to selected major risk factors, Vol. 2. Geneva, World Health Organization:1353–1433.

Cohen AJ et al. (2005). The global burden of disease due to outdoor air pollution. *Journal of Toxicology and Environmental Health Part A*. 68(13–14):1301–1307.

Committee on the Medical Effects of Air Pollutants (1998). *Quantification of the effects of air pollution on health in the United Kingdom*. London, The Stationery Office (http://www.advisorybodies.doh.gov.uk/comeap/statementsreports/airpol7.htm, accessed 16 April 2006).

Conceicao GM et al. (2001). Air pollution and child mortality: a time-series study in Sao Paulo, Brazil. *Environmental Health Perspectives*, 109(S3):347–350.

Crosignani P et al. (2003). Inquinamento atmosferico: quante vittime? [Air pollution: how many victims]. *Epidemiologia e Prevenzione*, 27(4):242–243 (in Italian).

Daniels MJ et al. (2000). Estimating particulate matter-mortality dose-response curves and threshold levels: an analysis of daily time-series for the 20 largest US cities. *American Journal of Epidemiology*, 152(5):397–406.

Declercq C, Macquet V (2000). Effets à court terme de l'ozone sur la santé respiratoire d'enfants d'Armentières, Nord de la France [Short-term effects of ozone on respiratory health of children in Armentieres, North of France]. *Revue d'Epidémiologie et de Santé Publique*, 48(S2):37–43 (in French).

Dejmek J et al. (1999). Fetal growth and maternal exposure to particulate matter during pregnancy. *Environmental Health Perspectives*, 107(6):475–480.

Dejmek J et al. (2000). The impact of polycyclic aromatic hydrocarbons and fine particles on pregnancy outcome. *Environmental Health Perspectives*, 108(12):1159–1164.

de'Munari E et al. (2004). Linee guida per la predisposizione delle reti di monitoraggio della qualità dell'aria in Italia [Guidelines for the predisposition of air quality monitoring network in Italy]. Rome, APAT, (http://www.sinanet.apat.it/site/_contentfiles/00037000/37045_ACE4.pdf, accessed 15 June 2005) (in Italian).

Department of Environmental Medicine and Public Health, University of Padua – Office of Hygiene, ARPAV – Padua Department, and Local Health Authority No. 16 of Padua (2005). *Analisi degli effetti a breve termine sulla salute correlati all'inquinamento atmosferico a Padova nel triennio 1999–2001 [Analysis of air pollution short-term health effects in Padua in the triennium 1999–2001]*. Padua, Department of Environmental Medicine and Public Health, University of Padua – Office of Hygiene, ARPAV – Padua Department, and Local Health Authority No. 16 of Padua, 2005 (http://www.biznet.it/bia/biaindex.htm www.arpa.veneto.it/salute/docs/amb_salute_rapporto_MISA2_PD.pdf, accessed 8 November 2005) (in Italian).

Dockery DW, Pope CA, 3rd (1994). Acute respiratory effects of particulate air pollution. *Annual Review of Public Health*, 15:107–132.

Dockery DW et al. (1989). Effects of inhalable particles on respiratory health of children. *The American Review of Respiratory Disease*, 139(3):587–594.

Dockery DW et al. (1993). An association between air pollution and mortality in six U.S. cities. *The New England Journal of Medicine*, 329(24):1753–1759.

Dockery DW et al. (1996). Health effects of acid aerosols on North American children: respiratory symptoms. *Environmental Health Perspectives*, 104(5):500–505.

Dominici F (2004). *Time-series analysis of air pollution and mortality: a statistical review*. Boston, MA, Health Effects Institute, (HEI Research Report 123; http://www.healtheffects.org/Pubs/Dominici.pdf, accessed 16 April 2006).

Dominici F, Samet JM, Zeger SL (2000). Combining evidence on air pollution and daily mortality from the largest 20 US cities: a hierarchical modeling strategy (with discussion). *Journal of the Royal Statistical Society*, A163(3):263–302.

Dominici F et al. (2002). On the use of generalized additive models in time-series studies of air pollution and health. *American Journal of Epidemiology*, 156(3):193–203.

Dominici F et al. (2003a). Shape of the exposure-response relation and mortality displacement in the NMMAPS database. In: Revised analyses of time-series studies of air pollution and health – revised analyses of the National Morbidity, Mortality, and Air Pollution Study. Part II. Revised analyses of selected time-series studies. Boston, Health Effects Institute: 91–96 (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006).

Dominici F et al. (2003b). Airborne particulate matter and mortality: timescale effects in four US cities. *American Journal of Epidemiology*, 157(12):1055–1065.

Dusseldorp A et al. (1995). Associations of PM10 and airborne iron with respiratory health of adults living near a steel factory. *American Journal of Respiratory and Critical Care Medicine*, 152(6 Pt 1):1932–1939.

EC Working Group on Particulate Matter (2004). *Guidance to the Member States on PM10 monitoring and intercomparisons with the reference method.* Bruxelles, European Commission. (http://europa.eu.int/comm/environment/air/pdf/finalwgreporten.pdf, accessed 10 June 2005).

Editors (2005). On health effects of ozone exposure and exposing the epidemiological process. *Epidemiology*, 16(4):425–426 (http://www.epidem.com/pt/re/epidemiology/fulltext.00001648-200507000-00001.htm;jsessionid=ECXUgcOt4Q3fUwYSZ316SizLtU12ru4v2bau0NPp9m17pm0E1byE!329698451!-949856144!9001!-1, accessed 16 April 2006).

Eftim S, Dominici, F (2005). Multisite time-series studies versus cohort studies: methods, findings, and policy implications. *Journal of Toxicology and Environmental Health, Part A*, 68(13):1191–1205.

EPA (1996). *Air quality criteria for particulate matter*. Research Triangle Park, NC, United States Environmental Protection Agency, Office of Air Quality Planning and Standards.

EPA (2004). *Air quality criteria for particulate matter*. Research Triangle Park, NC, United States Environmental Protection, Office of Air Quality Planning and Standards.

EPA (2005a). Review of the national ambient air quality standards for particulate matter: policy assessment of scientific and technical information. OAQPS Staff Paper. Research Triangle Park, NC, United States Environmental Protection Agency, Office of Air Quality Planning and Standards.

EPA (2005b). *Air quality criteria for ozone and related photochemical oxidants (second external review draft)*. Washington, DC, United States Environmental Protection Agency (EPA/600/R-05/004aB-cB).

EU (1999). Council Directive 1999/30/EC of 22 April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air. *Official Journal of the European Union*, L163(29/06/1999):41–60. (http://europa.eu.int/eur-lex/lex/LexUriServ/LexUriServ.do?uri=CELEX:31999L0030:EN:HTML, accessed 15 April 2006).

EU (2001). 2001/752/EC (Commission Decision of 17 October 2001 "amending the Annexes to Council Decision 97/101/EC establishing a reciprocal exchange of information and data from networks and individual stations measuring ambient air pollution within the Member States". *Official Journal of the European Union*, L282(26/10/2001)69–76 (http://europa.eu.int/eur-lex/lex/LexUriServ/LexUriServ.do?uri=CELEX:32001D0752:EN:HTML, accessed 15 April 2006).

EU (2002). Directive 2002/3/EC of the European Parliament and of the Council of 12 February 2002 relating to ozone in ambient air. *Official Journal of the European Union*, L67(9-/3-/2002):14–30. (http://europa.eu.int/eur-lex/pri/en/oj/dat/2002/l_067/l_06720020309en00140030.pdf, accessed 15 April 2006).

Ezzati M et al., eds (2004). Comparative quantification of health risks: global and regional burden of disease attributable to selected major risk factors, Vol. 2. Geneva, World Health Organization.

Faberi M, Martuzzi M, Pirrami M (2004). *Assessing the health impact and social costs of mopeds: feasibility study in Rome*. Copenhagen, WHO Regional Office for Europe (document number: EUR/02/5040650; http://www.euro.who.int/document/e83257.pdf, accessed 16 April 2006).

Filleul L et al. (2005). Twenty five year mortality and air pollution: results from the French PAARC survey. *Occupational and Environmental Medicine*, 62(7):453–460.

Fischer PH, Brunekreef B, Lebret E (2004). Air pollution related deaths during the 2003 heat wave in the Netherlands. *Atmospheric Environment*, 38:1083–1085.

Forastiere F et al. (2005). La frazione dei disturbi respiratori dell'infanzia attribuibile a fattori di rischio modificabili e non modificabili [The proportion of respiratory disorders in childhood attributable to preventable and not preventable risk factors]. *Epidemiologia e Prevenzione*, 29 (S2):67–69 (in Italian).

Fung K et al. (2003). *Testing the harvesting hypothesis by time domain regression analysis. Technical report.* Ottawa, Ontario, Institute of Population Health, University of Ottawa.

Galassi C, De Sario M, Forastiere F (2005). Studi Italiani sui Disturbi Respiratori dell' Infanzia e l'Ambiente [Italian Studies on respiratory disorders in children and environment. Second Phase]. *Epidemiologia e Prevenzione*, 29 (S2):1–98 (in Italian).

Gauderman WJ et al. (2002). Association between air pollution and lung function growth in Southern California children: results from a second cohort. *American Journal of Respiratory and Critical Care Medicine*, 166(1):76–84.

Gelfand AE, Hills SE, Racine-Poon A (1990). Illustration of Bayesian inference in normal data models using Gibbs sampling. *Journal of the American Statistical Association*, 85:972–985.

Gielen MH et al. (1997). Acute effects of summer air pollution on respiratory health of asthmatic children. *American Journal of Respiratory and Critical Care Medicine*, 155(6):2105–2108.

Gilboa SM et al. (2005). Relation between ambient air quality and selected birth defects, seven county study, Texas, 1997–2000. *American Journal of Epidemiology*, 162(3):238–252.

Gilliland FD et al. (2001). The effects of ambient air pollution on school absenteeism due to respiratory illnesses. *Epidemiology*, 12(1):43–54.

Glinianaia SV et al. (2004a). Particulate air pollution and fetal health: a systematic review of the epidemiologic evidence. *Epidemiology*, 15(1):36–45.

Glinianaia SV et al. (2004b). Does particulate air pollution contribute to infant death? A systematic review. *Environmental Health Perspectives*, 112(14):1365–1371.

Goldberg MS et al. (2001). Associations between daily cause-specific mortality and concentrations of ground-level ozone in Montreal, Quebec. *American Journal of Epidemiology*, 154(9):817–826.

Gouveia N, Fletcher T (2000). Time series analysis of air pollution and mortality: effects by cause, age and socioeconomic status. *Journal of Epidemiology and Community Health*, 54(10):750–755.

Gryparis A et al. (2004). Acute effects of ozone on mortality from the "air pollution and health: a European approach" project. *American Journal of Respiratory and Critical Care Medicine*, 170(10):1080–1087.

Gunier RB et al. (2003). Traffic density in California: socioeconomic and ethnic differences among potentially exposed children. *Journal of Exposure Analysis and Environmental Epidemiology*, 13(3):240–246.

Health for All Italia [online database] (2005). ISTAT (http://www.istat.it/sanita/Health/, accessed 15 June 2005) (in Italian).

HEI (2000a). The National Morbidity, Mortality, and Air Pollution Study. Part I: Methods and methodologic issues. Boston, Health Effects Institute, (HEI Research Report No. 94, part I).

HEI (2000b). The National Morbidity, Mortality, and Air Pollution Study. Part II: Morbidity and mortality from air pollution in the United States. Boston, Health Effects Institute, (HEI Research Report No. 94, part II).

HEI (2003). Revised analyses of time-series studies of air pollution and health. Special report. Revised analyses of the National Morbidity, Mortality, and Air Pollution Study, Part II. Boston, MA, Health Effects Institute (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006).

HEI (2004). *Health effects of outdoor air pollution in developing countries of Asia: a literature review*. Boston, MA, Health Effects Institute, (HEI Special Report 15; http://www.healtheffects.org/International/PAPA-progress4.pdf, accessed 16 April 2006).

Hertz-Picciotto I (1995). Epidemiology and quantitative risk assessment: a bridge from science to policy. *American Journal of Public Health*, 85(4):484–493.

Higgins BG et al. (1995). Effects of air pollution on symptoms and peak expiratory flow measurements in subjects with obstructive airways disease. *Thorax*, 50(2):149–155.

Hiltermann TJ et al. (1998). Asthma severity and susceptibility to air pollution. *European Respiratory Journal*, 11(3):686–693.

Hoek G, Brunekreef B (1995). Effect of photochemical air pollution on acute respiratory symptoms in children. *American Journal of Respiratory and Critical Care Medicine*, 151(1):27–32.

Hoek G et al. (2000). Daily mortality and air pollution in The Netherlands. *Journal of the Air & Waste Management Association*, 50(8):1380–1389.

Hoek G et al. (2002). Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. *Lancet*, 360(9341):1203–1209.

Holland MR et al. (2005). *Methodology for the cost-benefit analysis for CAFE. Vol. 1. Overview of methodology*. Didcot, Oxon, United Kingdom, AEA Technology Environment (AEAT/ED51014/Methodology Paper Issue 4; http://europa.eu.int/comm/environment/air/cafe/pdf/cba_methodology_vol1.pdf, accessed 16 April 2006).

Hong YC et al. (2002). Air pollution: a new risk factor in ischemic stroke mortality. *Stroke*, 33(9):2165–2169.

Hubbell BJ et al. (2005). Health-related benefits of attaining the 8-hr ozone standard. *Environmental Health Perspectives*, 113(1):73–82.

Hurley F et al. (2005). *Methodology for the cost-benefit analysis for CAFE. Vol. 2. Health impact assessment*. Didcot, Oxon, United Kingdom, AEA Technology Environment (AEAT/ED51014/Methodology Volume 2 Issue 1; http://europa.eu.int/comm/environment/air/cafe/pdf/cba_methodology_vol2.pdf, accessed 16 April 2006).

ISTAT & ACI (Automobile Club of Italy) (2003). Statistica degli incidenti stradali – Anno 2001. [Statistics on road traffic accidents – Year 2001]. Rome, ISTAT (Informazioni n.18 – 2003) (in Italian).

ISTAT (2001). 14° Censimento generale della popolazione e delle abitazioni [14th general population and housing census] [online database]. Rome, ISTAT (http://dawinci.istat.it/daWinci/jsp/MD/dawinciMD.jsp?a1=m0GG0c0I0&a2=mJ2Y8048f8&n=1UH30T09OG008, accessed 1 June 2005) (in Italian).

ISTAT (2005). Sistema di indicatori territoriali [System of territorial indicators] [online database]. Rome, ISTAT (http://sitis.istat.it/sitis/html/, accessed 1 October 2005) (in Italian).

Italian Agency for the Protection of the Environment and for Technical Services (APAT) (2005). *Qualità dell'ambiente urbano. II rapporto APAT. Edizione 2005 [Urban environment quality. 2nd APAT report. 2005 ed]*. Rome, Italian Agency for the Protection of the Environment and for Technical Services (APAT) (in Italian).

Italian Ministry of the Environment (1991). DM 20.5.1991 (Criteri per la raccolta dei dati inerenti la qualità dell'aria [Criteria for collection of air quality data]). Rome, Italian Ministry of the Environment (in Italian).

Italian Ministry of the Environment (2002). DM n° 60 2.4.2002 ("Recepimento della direttiva 1999/30/CE del Consiglio del 22 aprile 1999 concernente i valori limite di qualità dell'aria

ambiente per il biossido di zolfo, il biossido di azoto, gli ossidi di azoto, le particelle e il piombo e della direttiva 2000/69/CE relativa ai valori limite di qualità aria ambiente per il benzene ed il monossido di carbonio" [Endorsement of Council Directive 1999/30/EC of 22 April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air and of Directive 2000/69/EC of the European Parliament and of the Council of 16 November 2000 relating to limit values for benzene and carbon monoxide in ambient air]). Rome, Italian Ministry of the Environment (in Italian).

Ito K, De Leon SF, Lippmann M (2005). Associations between ozone and daily mortality: analysis and meta-analysis. *Epidemiology*, 16(4):446–457.

Jalaludin BB et al. (2000). Acute effects of low levels of ambient ozone on peak expiratory flow rate in a cohort of Australian children. *International Journal of Epidemiology*, 29(3):549–557.

Janssen NA et al. (2002). Air conditioning and source-specific particles as modifiers of the effect of PM(10) on hospital admissions for heart and lung disease. *Environmental Health Perspectives*, 110(1):43–49.

Jerrett M et al. (2004). Do socioeconomic characteristics modify the short term association between air pollution and mortality? Evidence from a zonal time series in Hamilton, Canada. *Journal of Epidemiology and Community Health*, 58(1):31–40.

Just J et al. (2002). Short-term health effects of particulate and photochemical air pollution in asthmatic children. *European Respiratory Journal*, 20(4):899–906.

Kaiser J (1997). Showdown over clean air science. *Science*, 277(5325):466–469.

Katsouyanni K et al. (2001). Confounding and effect modification in the short-term effects of ambient particles on total mortality: results from 29 European cities within the APHEA2 project. *Epidemiology*, 12(5):521–531.

Katsouyanni K et al. (2002). Different convergence parameters applied to the S-PLUS GAM function. *Epidemiology*, 13(6):742–743.

Katsouyanni K et al. (2003). Sensitivity analysis of various models of short-term effects of ambient particles on total mortality in 29 cities in APHEA2. In: *Revised analyses of time-series studies of air pollution and health.* Boston: Health Effects Institute: 157–164 (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006).

Kim SY et al. (2004). Determining the threshold effect of ozone on daily mortality: an analysis of ozone and mortality in Seoul, Korea, 1995–1999. *Environmental Research,* 94(2):113–119.

Klemm RJ et al. (2000). Is daily mortality associated specifically with fine particles? Data reconstruction and replication of analyses. *Journal of the Air and Waste Management Association*, 50(7):1215–1222.

Klemm RJ, Mason ES (2003). Replication of reanalysis of Harvard Six City mortality study. In: *Revised analyses of time-series studies of air pollution and health*. Boston, MA, Health

Effects Institute: 165–172 (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006).

Krewski D et al. (2000a). Reanalysis of the Harvard Six Cities Study and the American Cancer Society study of particulate air pollution and mortality. Part 1: replication and validation. Cambridge, MA, Health Effects Institute (http://www.healtheffects.org/Pubs/Rean-ExecSumm.pdf, accessed 16 April 2006).

Krewski D et al. (2000b). Reanalysis of the Harvard Six Cities Study and the American Cancer Society study of particulate air pollution and mortality. Part 2: sensitivity analyses. Cambridge, MA, Health Effects Institute (http://www.healtheffects.org/Pubs/Rean-ExecSumm.pdf, accessed 16 April 2006).

Krewski D et al. (2003). Overview of the reanalysis of the Harvard Six Cities Study and American Cancer Society Study of Particulate Air Pollution and Mortality. *Journal of Toxicology and Environmental Health, Part A*, 66(16–19):1507–1551.

Krzyzanowski M, Kuna-Dibbert B, Schneider J, eds (2005). *Health effects of transport-related air pollution*. Copenhagen, WHO Regional Office for Europe (http://www.euro.who.int/document/e86650.pdf, accessed 16 April 2006).

Künzli N et al. (1999). Health costs due to road traffic-related air pollution. An impact assessment project of Austria, France and Switzerland: air pollution attributable cases. (Prepared for the third Ministerial Conference for Environment and Health, London, 1999). Copenhagen, WHO Regional Office for Europe (http://ns.who.dk/document/trt/cases.pdf, accessed 7 May 2006).

Künzli N et al. (2000). Public-health impact of outdoor and traffic-related air pollution: a European assessment. *Lancet*, 356(9232):795–801.

Künzli N et al. (2001). Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies? *American Journal of Epidemiology*, 153(11):1050–1055.

Laden F et al. (2006). Reduction in fine particulate air pollution and mortality: extended follow-up of the Harvard Six Cities study. *American Journal of Respiratory and Critical Care Medicine*, 173(6):667–672.

Le Tertre A et al. (2002a). Short-term effects of particulate air pollution on cardiovascular diseases in eight European cities. *Journal of Epidemiology and Community Health*, 56(10):773–779.

Le Tertre A et al. (2002b). Short-term effects of air pollution on mortality in nine French cities: a quantitative summary. *Archives of Environmental Health*, 57(4):311–319.

Le Tertre A et al. (2003). Short-term effects of particulate air pollution on cardiovascular diseases in eight European cities. In: *Revised analyses of time-series studies of air pollution and health*. Boston, MA, Health Effects Institute: 173–176 (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006).

Levy D et al. (2001). A case-crossover analysis of particulate matter air pollution and out-of-hospital primary cardiac arrest. *Epidemiology*, 12(2):193–199.

Levy JI, Chemerynski SM, Sarnat JA (2005). Ozone exposure and mortality: an empiric bayes metaregression analysis. *Epidemiology*, 16(4):458–468.

Levy JI et al. (2001). Assessing the public health benefits of reduced ozone concentrations. *Environmental Health Perspectives*, 109(12):1215–1226.

Liguria Region, Department of the Environment, Policy Division for Sustainable Development (Regione Liguria – Dipartimento Ambiente – Settore Politiche dello Sviluppo Sostenibile) (2005). *Valutazione Annuale della qualità dell'aria – Anno 2004 [Annual air quality assessment – year 2004]*. Genoa, Liguria Region, Department of the Environment, Policy Division for Sustainable Development (http://www.regione.liguria.it/MenuSezione.asp?Parametri=4_10_2_368_$4_10_2_368_1767_$Inquinamento_acustico$4_10_2_368_-1$valutazione.htm$, accessed 1 November 2005) (in Italian).

Lipfert FW et al. (2000). The Washington University-EPRI Veterans' Cohort Mortality Study: preliminary results. *Inhalation Toxicology*, 12(S4):41–73.

Loomis D et al. (1999). Air pollution and infant mortality in Mexico City. *Epidemiology*, 10(2):118–123.

Maisonet M et al. (2001). Relation between ambient air pollution and low birth weight in the Northeastern United States. *Environmental Health Perspectives*, 109(Suppl. 3)351–356.

Maisonet M et al. (2004). A review of the literature on the effects of ambient air pollution on fetal growth. *Environmental Research*, 95(1):106–115.

Marconi A et al. (2004). Campagne di monitoraggio del numero di particelle a Roma [Monitoring campaign on number of particles in Rome]. In: *Primo convegno nazionale sul particolato atmosferico. Problematiche e prospettive [First national congress on atmospheric particulate matter:problems and perspectives]*, Milan, 12–14 May 2004. (http://www.disat.unimib.it/chimamb/PM2004/ORALI_pdf/marconi.pdf, accessed 7 May 2006) (in Italian).

Martuzzi M (2001). Re: "Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies?" *American Journal of Epidemiology*, 154(10):974–975.

Martuzzi M et al. (2002). *Health impact assessment of air pollution in the eight major Italian cities*. Copenhagen, WHO Regional Office for Europe, (EURO/02/5040650; http://www.euro.who.int/document/e75492.pdf, accessed 16 April 2006).

Mathers CD et al., eds (2001). *National burden of disease studies: a practical guide. Edition 2.0*. Geneva, World Health Organization, Global Programme on Evidence for Health Policy (www.ihpp.thaigov.net/BOD/NBD%20Manual%20V2.0.doc, accessed 16 April 2006).

McConnell R et al. (1999). Air pollution and bronchitic symptoms in Southern California children with asthma. *Environmental Health Perspectives*, 107(9):757–760.

McConnell R et al. (2002). Asthma in exercising children exposed to ozone: a cohort study. *Lancet*, 359(9304):386–391.

Medina S et al. (2005). *APHEIS health impact assessment of air pollution and communication strategy. Third-year report, 2002-2003)*. Saint-Maurice, Institute de Veille Sanitaire (http://www.apheis.net/, accessed 16 April 2006).

Michelozzi P et al. (1998). Air pollution and daily mortality in Rome, Italy. *Occupational and Environmental Medicine*, 55(9):605–610.

Moorcroft S et al. (1999). Assistance with the review and assessment of PM10 in relation to the proposed stage 1 of EU limit values: a report prepared for the Department of Environment, Transports and the Regions, the Welsh Office and the Scottish Office. Croydon, United Kingdom, Stanger Science & Environment (http://www.aeat.co.uk/netcen/airqual/reports/stanpm/spmain.html, accessed 16 April 2006).

Morris RD (2001). Airborne particulates and hospital admissions for cardiovascular disease: a quantitative review of the evidence. *Environmental Health Perspectives*, 109(Suppl. 4)495–500.

Mortimer KM et al. (2000). The effect of ozone on inner-city children with asthma: identification of susceptible subgroups. *American Journal of Respiratory and Critical Care Medicine*, 162(5):1838–1845.

Mudu P et al. (to be published). *Health effects and risk of transport systems*. Copenhagen, WHO Regional Office for Europe.

Municipality of Catania (Comune di Catania) (2003). *Rapporto Annuale 2002 sulla qualità dell'aria [Annual report on air quality, 2002]*. Catania, Municipality of Catania (http://www.comune.catania.it/portale/comct/direzioni/ambiente_online/arch/RapportoQualitaAria2002/rapporto_qualità_dell%27aria_2002.htm, accessed 21 December 2005) (in Italian).

Municipality of Catania (Comune di Catania) (2004). *Rapporto Annuale 2003 sulla qualità dell'aria [Annual report on air quality, 2003]*. Catania, Municipality of Catania (http://www.comune.catania.it/portale/comct/direzioni/ambiente_online/rapporto_qualit_aria_2003.asp, accessed 21 December 2005) (in Italian).

Municipality of Catania (Comune di Catania) (2005). *Rapporto annuale 2004 sulla qualità dell'aria [Annual report on air quality, 2004]*. Catania, Municipality of Catania (http://www.comune.ct.it/portale/comct/direzioni/ambiente_online/rapporto_2004/rapp_qual_aria.pdf, accessed 21 December 2005) (in Italian).

Municipality of Venice (Comune di Venezia) (2003). *Qualità dell'aria nel comune di Venezia: rapporto annuale 2002 [Air quality in the city of Venice: annual report, 2002]*. Venice, Municipality of Venice – Local Environmental Aurthority & Regional Agency for Environmental Prevention and Protection in Veneto – Air Observatory (http://www.ambiente.venezia.it/aria/doc/2002/RAQA%202002.pdf, accessed 10 September 2005) (in Italian).

Municipality of Venice (Comune di Venezia) (2004). *Qualità dell'aria nel comune di Venezia. Rapporto Annuale 2003 [Air quality in the city of Venice: annual report, 2003]*.

Venice, Municipality of Venice – Local Environmental Aurthority & Regional Agency for Environmental Prevention and Protection in Veneto – Air Observatory (http://www.ambiente.venezia.it/aria/doc/2003/RAQA%202003.pdf, accessed 10 September 2005) (in Italian).

Municipality of Venice (Comune di Venezia) (2005). *Qualità dell'aria nel comune di Venezia. Rapporto Annuale 2004 [Air quality in the city of Venice: annual report, 2004]*. Venice, Municipality of Venice – Local Environmental Aurthority & Regional Agency for Environmental Prevention and Protection in Veneto – Air Observatory (http://www.ambiente.venezia.it/aria/doc/2004/RAQA%202004.pdf, accessed 10 November 2005) (in Italian).

Municipality of Verona (Comune di Verona) (2006). *Bollettino inquinamento atmosferico & acustico [Bulletin of atmosphercia and acoustic pollution]* Verona, Municipality of verona (http://www.biznet.it/bia/biaindex.htm, accessed 20 January 2006) (in Italian).

Murray C, Lopez AD, eds (1996). The global burden of disease*: a comprehensive assessment of mortality and disability from diseases, injuries, and risk factors in 1990 and projected to 2020*. Geneva, World Health Organization, Harvard School of Public Health, World Bank.

Nafstad P et al. (2003). Lung cancer and air pollution: a 27 year follow up of 16209 Norwegian men. *Thorax*, 58(12):1071–1076.

Nafstad P et al. (2004). Urban air pollution and mortality in a cohort of Norwegian men. *Environmental Health Perspectives*, 112(5):610–615.

National Research Council, Committee on Research Priorities for Airborne Particulate Matter (2004). *Research priorities for airborne particulate matter. IV. Continuing research progress.* Washington, DC, The National Academy Press (http://www.nap.edu/books/0309091993/html/, accessed September 2005).

Neas LM, Schwartz J, Dockery D (1999). A case-crossover analysis of air pollution and mortality in Philadelphia. *Environmental Health Perspectives*, 107(8):629–631.

Neukirch F et al. (1998). Short-term effects of low-level winter pollution on respiratory health of asthmatic adults. *Archives of Environmental Health*, 53(5):320–328.

Nyberg F et al. (2000). Urban air pollution and lung cancer in Stockholm. *Epidemiology*, 11(5):487–495.

O'Neill MS et al. (2003). Health, wealth, and air pollution: advancing theory and methods. *Environmental Health Perspectives*, 111(16):1861–1870.

Ostro B (1987). Air pollution and morbidity revisited: a specification test. *Journal of Environmental Economics and Management*, 14(1):87–98.

Ostro B (1993). The association of air pollution and mortality: examining the case for inference. *Archives of Environmental Health*, 48(5):336–342.

Ostro B (2004). *Outdoor air pollution: assessing the environmental burden of disease at national and local levels*. Geneva, World Health Organization (Environmental Burden of

Disease Series, No 5; http://whqlibdoc.who.int/publications/2004/9241591463.pdf, accessed 15 April 2006).

Ostro B, Chestnut L (1998). Assessing the health benefits of reducing particulate matter air pollution in the United States. *Environmental Research*, 76(2):94–106.

Ostro B, Rothschild S (1989). Air pollution and acute respiratory morbidity: an observational study of multiple pollutants. *Environmental Research*, 50(2):238–247.

Ostro B et al. (1999). The impact of particulate matter on daily mortality in Bangkok, Thailand. *Journal of the Air and Waste Management Association*, 49(9 Spec. No.):100–107.

Pan American Health Organization (2005). *An assessment of health effects of ambient air pollution in Latin America and the Caribbean*. Washington, DC, Pan American Health Organization.

Park H et al. (2002). Association of air pollution with school absenteeism due to illness. *Archives of Pediatrics & Adolescent Medicine*, 156(12):1235–1239.

Pereira LA et al. (1998). Association between air pollution and intrauterine mortality in Sao Paulo, Brazil. *Environmental Health Perspectives*, 106(6):325–329.

Perera FP et al. (1992). Molecular and genetic damage in humans from environmental pollution in Poland. *Nature*, 360(6401):256–258.

Perera FP et al. (1999). Molecular epidemiologic research on the effects of environmental pollutants on the fetus. *Environmental Health Perspectives*, 107(Suppl. 3):451–460.

Peters A et al. (2001). Increased particulate air pollution and the triggering of myocardial infarction. *Circulation*, 103(23):2810–2815.

Peters A et al. (2005). Particulate air pollution and nonfatal cardiac events. Part I. Air pollution, personal activities, and onset of myocardial infarction in a case-crossover study. Boston, Health Effect Institute, (HEI Research Report 124, part I).

Peters JM et al. (1999). A study of twelve Southern California communities with differing levels and types of air pollution. I. Prevalence of respiratory morbidity. *American Journal of Respiratory and Critical Care Medicine*, 159(3):760–767.

Ponce de Leon A et al. (1996). Effects of air pollution on daily hospital admissions for respiratory disease in London between 1987–88 and 1991–92. *Journal of Epidemiology and Community Health*, 50(Suppl. 1):s63–s70.

Pope CA III (2000). Particulate matter-mortality exposure-response relations and threshold. American *Journal of Epidemiology*, 152(5):407–412.

Pope CA III, Dockery DW (1992). Acute health effects of PM10 pollution on symptomatic and asymptomatic children. *The American Review of Respiratory Disease*, 145(5):1123–1128.

Pope CA III, Dockery DW (1999). Epidemiology of particle effects. In: Holgate ST et al., eds. *Air pollution and health*. London, Academic Press.

Pope CA III et al. (1995). Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *American Journal of Respiratory and Critical Care Medicine*, 151(3 Pt 1):669–674.

Pope CA III et al. (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *The Journal of the American Medical Association*, 287(9):1132–1141.

Pope CA III et al. (2004). Cardiovascular mortality and long-term exposure to particulate air pollution: epidemiological evidence of general pathophysiological pathways of disease. *Circulation*, 109(1):71–77.

Portney PR, Mullahy J (1986). Urban quality and acute respiratory illness. *Journal of Urban Economics*, (20):31–38.

Prati P et al. (2004). Studio della composizione del particolato atmosferico nell'area urbana genovese: tecniche di analisi e risultati nelle frazioni PM10, PM2.5 e PM1 [Study on the composition of atmospheric particulate matter in the urban area of Genoa: techniques of analysis and results for PM10, PM2.5 and PM1]. In: *Primo convegno nazionale sul particolato atmosferico. Problematiche e prospettive [First national congress on atmospheric particulate matter:problems and perspectives],* Milan, 12–14 May 2004 (http://www.disat.unimib.it/chimamb/PM2004/ORALI_pdf/prati.pdf, accessed 7 May 2006) (in Italian).

Regional Agency for Environmental Prevention and Protection in Friuli Venezia-Giulia (ARPA FVG) (2003). *Relazione qualità dell'aria anno 2002 [Air Quality Report, year 2002]*. Trieste, Regional Agency for Environmental Prevention and Protection in Friuli Venezia-Giulia (http://www.arpa.fvg.it/Aria-Radia/Tutela-Qua/RETE-DI-RI/CENTRALINE/Relazione-/, accessed 21 December 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection in Friuli Venezia-Giulia (ARPA FVG) (2004). *Relazione qualità dell'aria anno 2003 [Air quality report, year 2003]*. Trieste, Regional Agency for Environmental Prevention and Protection in Friuli Venezia-Giulia (http://www.arpa.fvg.it/Aria-Radia/Tutela-Qua/RETE-DI-RI/CENTRALINE/Relazione-2/body.htm, accessed 21 December 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection in Lombardy (ARPA Lombardia) (2003). *Rapporto sulla qualità dell'aria di Milano e Provincia: anno 2002 [Air Quality Report in Milan and its District: year 2002]*. Milan, Regional Agency for Environmental Prevention and Protection in Lombardy (http://www.arpalombardia.it/qaria/pdf/RQA-2002/RQA_MI_02.pdf, accessed 1 November 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection in Lombardy (ARPA Lombardia) (2004). *Rapporto sulla qualità dell'aria di Milano e Provincia: anno 2003 [Air quality report in Milan and its district: year 2003]*. Milan, Regional Agency for Environmental Prevention and Protection in Lombardy

(http://www.arpalombardia.it/qaria/pdf/RQA-2003/RQA-MI%202003.pdf, accessed 1 November 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection in Lombardy (ARPA Lombardia) (2005a). *Rapporto sulla qualità dell'aria di Milano e Provincia: anno 2004 [Air quality report in Milan and its district: year 2004]*. Milan, Regional Agency for Environmental Prevention and Protection in Lombardy (http://www.arpalombardia.it/qaria/pdf/RQA-2004/RQA-MI%202004.pdf, accessed 1 November 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection in Lombardy (ARPA Lombardia) (2005b), Il sistema di misura classico (SMC) e sistema di misura 2005 (SM2005) [The classical measurement system (SMC) and 2005 measurement system (SM2005)] Milan, Regional Agency for Environmental Prevention and Protection in Lombardy (http://www.arpalombardia.it/qaria/docPM10.asp, accessed 20 December 2005) (in Italian)[1].

Regional Agency for Environmental Prevention and Protection of in Tuscany (ARPA Toscana) (2003). Rapporto annuale sulla qualità dell'aria (dati dell'anno 2002, area omogenea di Firenze) [Annual report on air quality, Florence area, year 2002]. Florence, Regional Agency for Environmental Prevention and Protection of in Tuscany (http://www.arpat.toscana.it/aria/ar_relazioni2.html#2002, accessed 1 October 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection of in Tuscany (ARPA Toscana) (2004). Rapporto annuale sulla qualità dell'aria (dati dell'anno 2003, area omogenea di Firenze) [Annual report on air quality, Florence area, year 2003]. Florence, Regional Agency for Environmental Prevention and Protection of in Tuscany (http://www.arpat.toscana.it/aria/ar_relazioni2.html#2003, accessed 1 October 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection in Tuscany (ARPA Toscana) (2005). Rapporto annuale sulla qualità dell'aria (dati dell'anno 2004, area omogenea di Firenze) [Annual report on air quality, Florence area, year 2004]. Florence, Regional Agency for Environmental Prevention and Protection of in Tuscany (http://www.arpat.toscana.it/aria/ar_relazioni2.html#2004, accessed 1 October 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection in Emilia-Romagna (ARPA Emilia Romagna) (2005). *Lo stato di qualità dell'aria nella provincia di Bologna - Anno 2004 [Air quality status in Bologna District – year 2004]*. Bologna, Regional Agency for Environmental Prevention and Protection in Emilia-Romagna (http://www.arpa.emr.it/bologna/download/report_aria04.pdf, accessed 15 October 2005) (in Italian).

Regional Agency for Environmental Prevention and Protection of in Veneto (ARPAV) (2005). *Report Qualità Aria - Polveri PM10 nel Veneto [Air quality report – PM10 in Veneto region]*. Padua, Regional Agency for Environmental Prevention and Protection of in Veneto - Meteorological Center of Teolo Regional Air Observatory (http://www.arpa.veneto.it/indice.asp?l=aria.htm, accessed 15 December 2005) (in Italian).

[1] Since then page content has changed.

Ritz B, Yu F (1999). The effect of ambient carbon monoxide on low birth weight among children born in southern California between 1989 and 1993. *Environmental Health Perspectives*, 107(1):17–25.

Ritz B et al. (2000). Effect of air pollution on preterm birth among children born in Southern California between 1989 and 1993. *Epidemiology*, 11(5):502–511.

Roemer WH, van Wijnen JH (2001). Daily mortality and air pollution along busy streets in Amsterdam, 1987–1998. *Epidemiology*, 12(6):649–653.

Saez M et al. (2002). A combined analysis of the short-term effects of photochemical air pollutants on mortality within the EMECAM project. *Environmental Health Perspectives*, 110(3):221–228.

Saldiva PH et al. (1994). Association between air pollution and mortality due to respiratory diseases in children in Sao Paulo, Brazil: a preliminary report. *Environmental Research*, 65(2):218–225.

Samet JM, DeMarini DM, Malling HV (2004). Biomedicine. Do airborne particles induce heritable mutations? *Science*, 304(5673):971–972.

Samet JM et al. (2000). Fine particulate air pollution and mortality in 20 U.S. cities, 1987-1994. *The New England Journal of Medicine*, 343(24):1742–1749.

Samet JM et al. (2003). New problems for an old design: time series analyses of air pollution and health. *Epidemiology*, 14(1):11–12.

Samoli E et al. (2005). Estimating the exposure-response relationships between particulate matter and mortality within the APHEA multicity project. *Environmental Health Perspectives*, 113(1):88–95.

Sartor F (2004). *La surmortalité en Belgique au cours de l'été 2003*. Brussels, Institut Scientifique de Santé Publique, section de l'épidémiologie [Scientific Institute of Public Health, Epidemiology Unit] (Report number: D/2004/2505/17; http://www.iph.fgov.be/epidemio/epifr/ozonefr/surmor03.pdf, accessed 16 April 2006) (in French).

Schwartz J (1997). Air pollution and hospital admissions for cardiovascular disease in Tucson. *Epidemiology*, 8(4):371–377.

Schwartz J (1999). Air pollution and hospital admissions for heart disease in eight U.S. counties. *Epidemiology*, 10(1):17–22.

Schwartz J (2000). Harvesting and long term exposure effects in the relation between air pollution and mortality. *American Journal of Epidemiology*, 151(5):440–448.

Schwartz J (2001). Is there harvesting in the association of airborne particles with daily deaths and hospital admissions? *Epidemiology*, 12(1):55–61.

Schwartz J (2003). Airborne particles and daily deaths in 10 US cities. In: *Revised analyses of time-series studies of air pollution and health*. Boston, Health Effects Institute. 211–218 (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006).

Schwartz J (2004). Air pollution and children's health. *Pediatrics*, 113(Suppl. 4):1037–1043.

Schwartz J (2005). How sensitive is the association between ozone and daily deaths to control for temperature? *American Journal of Respiratory and Critical Care Medicine*, 171(6):627–631.

Schwartz J, Dockery DW, Neas LM (1996). Is daily mortality associated specifically with fine particles? *Journal of the Air and Waste Management Association*, 46(10):927–939.

Schwartz J, Neas LM (2000). Fine particles are more strongly associated than coarse particles with acute respiratory health effects in schoolchildren. *Epidemiology*, 11(1):6–10.

Schwartz J, Zanobetti A (2000). Using meta-smoothing to estimate dose-response trends across multiple studies, with application to air pollution and daily death. *Epidemiology*, 11(6):666–672.

Schwartz J et al. (2001). The concentration-response relation between air pollution and daily deaths. *Environmental Health Perspectives*, 109(10):1001–1006.

SIDRIA (1997). Asthma and respiratory symptoms in 6–7 yr old Italian children: sex, latitude, urbanization and socioeconomic factors. SIDRIA (Italian Studies on Respiratory Disorders in Childhood and the Environment). *European Respiratory Journal*, 10(8):1780–1786 (http://erj.ersjournals.com/cgi/reprint/10/8/1780, accessed 16 April 2006).

SIDRIA (unpublished).

Somers CM et al. (2002). Air pollution induces heritable DNA mutations. *Proceedings of the National Academy of Sciences of the United States of America,* 99(25):15904–15907.

Somers CM et al. (2004). Reduction of particulate air pollution lowers the risk of heritable mutations in mice. *Science*, 304(5673):1008–1010.

Spiegelhalter DJ, Thomas A, Best NG (1999). *WinBUGS Version 1.2 User Manual*. Cambridge, England, MRC Biostatistics Unit.

Spix C et al. (1998). Short-term effects of air pollution on hospital admissions of respiratory diseases in Europe: a quantitative summary of APHEA study results. Air Pollution and Health: a European Approach. *Archives of Environmental Health*, 53(1):54–64.

Stedman JR (2004). The predicted number of air pollution related deaths in the UK during the August 2003 heat wave. *Atmospheric Environment*, 38:1087–1090.

Stieb DM, Judek S, Burnett RT (2002). Meta-analysis of time-series studies of air pollution and mortality: effects of gases and particles and the influence of cause of death, age, and season. *Journal of the Air & Waste Management Association*, 52(4):470–484.

Stieb DM et al. (2002). Air pollution and disability days in Toronto: results from the national population health survey. *Environmental Research*, 89 (3):210–219.

Sullivan J et al. (2003). Exposure to ambient fine particulate matter and primary cardiac arrest among persons with and without clinically recognized heart disease. *American Journal of Epidemiology*, 157(6):501–509.

Sullivan J et al. (2005). Relation between short-term fine-particulate matter exposure and onset of myocardial infarction. *Epidemiology*, 16(1):41–48.

Thurston GD, Ito K (2001). Epidemiological studies of acute ozone exposures and mortality. *Journal of Exposure Analysis and Environmental Epidemiology*, 11(4):286–294.

Tiittanen P et al. (1999). Fine particulate air pollution, resuspended road dust and respiratory health among symptomatic children. *European Respiratory Journal*, 13(2):266–273.

Tominz R, Mazzoleni B, Daris F (2005). Stima dei potenziali benefici sanitari della riduzione dell'inquinamento atmosferico da PM10 nella citta di Trieste [Estimate of potential health benefit of reduction of air pollution due to PM10 in Trieste (Italy)]. *Epidemiologia e Prevenzione*, 29 (3–4):149–155 (in Italian).

UNECE (2004). *Modelling and assessment of the health impact of particulate matter and ozone*. Geneva, United Nations Economic Commission for Europe (http://www.unece.org/env/documents/2004/eb/wg1/eb.air.wg1.2004.11.e.pdf, accessed 31 May 2005).

van der Heijden KA, Krzyzanowski M (1994). Priorities in environmental health risks related to air pollution throughout Europe. *Zentralblatt für Bakteriologie: International Journal of Medical Microbiology*, 281(3):296–302.

van der Zee S et al. (1999). Acute effects of urban air pollution on respiratory health of children with and without chronic respiratory symptoms. *Occupational and Environmental Medicine*, 56(12):802–812.

van der Zee SC et al. (2000). Acute effects of air pollution on respiratory health of 50-70 yr old adults. *European Respiratory Journal*, 15(4):700–709.

Van Dingenen R et al. (2004). A European aerosol phenomenology – 1: physical characteristics of particulate matter at kerbside, urban, rural and background sites in Europe. *Atmospheric Environment*, 38:2561–2577.

Vineis P et al. (2004). Outdoor air pollution and lung cancer: recent epidemiologic evidence. *International Journal of Cancer*, 111(5):647–652.

von Klot S et al. (2005). Ambient air pollution is associated with increased risk of hospital cardiac readmissions of myocardial infarction survivors in five European cities. *Circulation*, 112(20):3073–3079.

Wang X et al. (1997). Association between air pollution and low birth weight: a community-based study. *Environmental Health Perspectives*, 105(5):514–520.

Ward DJ, Ayres JG (2004). Particulate air pollution and panel studies in children: a systematic review. Occupational and *Environmental Medicine*, 61(4):e13 (http://oem.bmjjournals.com/cgi/content/full/61/4/e13, accessed 17 April 2006).

Weiland S, Forastiere F (2005). The impact of air pollution on asthma and allergies in children. In: *Effects of air pollution on children's health and development. A review of the evidence*. Copenhagen, WHO Regional Office for Europe: 70–107 (document number: EUR/05/5046027; http://www.euro.who.int/document/e86575.pdf, accessed 17 April 2006).

Weisel CP et al. (2002). Concepts in developing health-based indicators for ozone. *International Archives of Occupational and Environmental Health*, 75(6):415–422.

WHO (1978). International classification of diseases*: [9th] ninth revision, basic tabulation list with alphabetic index*. Geneva, World Health Organization .

WHO Regional Office for Europe (2000). *Air quality guidelines for Europe, second edition*. Copenhagen, WHO Regional Office for Europe (WHO Regional Publications, European Series, No. 91; http://www.who.dk/document/e71922.pdf, accessed 17 April 2006).

WHO Regional Office for Europe (2003). Health aspects of air pollution with particulate matter, ozone and nitrogen dioxide*: report on a WHO working group, Bonn, Germany 13-15 January 2003*. Copenhagen, WHO Regional Office for Europe, (document number: EUR/03/5042688; http://www.euro.who.int/document/e79097.pdf, accessed 17 April 2006).

WHO Regional Office for Europe (2005a. *Effects of air pollution on children's health and development: a review of the evidence*. Copenhagen, WHO Regional Office for Europe (document number: EUR/05/5046027; http://www.euro.who.int/document/e86575.pdf, accessed 17 April 2006).

WHO Regional Office for Europe (2005b). *Particulate matter air pollution: how it harms health*. Berlin, Copenhagen, Rome, WHO Regional Office for Europe (Fact sheet EURO/04/05; http://www.euro.who.int/document/mediacentre/fs0405e.pdf, accessed 17 April 2006).

Wilhelm M, Ritz B (2003). Residential proximity to traffic and adverse birth outcomes in Los Angeles county, California, 1994–1996. *Environmental Health Perspectives*, 111(2):207–216.

Willis A et al. (2003). Selection of ecologic covariates in the American Cancer Society study. *Journal of Toxicology and Environmental Health, Part A*, 66(16–19):1563–1589.

Wong CM et al. (2002). A tale of two cities: effects of air pollution on hospital admissions in Hong Kong and London compared. *Environmental Health Perspectives*, 110(1):67–77.

Woodruff TJ, Grillo J, Schoendorf KC (1997). The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States. *Environmental Health Perspectives*, 105(6):608–612.

Xu X, Ding H, Wang X (1995). Acute effects of total suspended particles and sulfur dioxides on preterm delivery: a community-based cohort study. *Archives of Environmental Health*, 50(6):407–415.

Yang Q et al. (2003). Association between ozone and respiratory admissions among children and the elderly in Vancouver, Canada. *Inhalation Toxicology*, 15(13):1297–1308.

Ye F et al. (2001). Effects of temperature and air pollutants on cardiovascular and respiratory diseases for males and females older than 65 years of age in Tokyo, July and August 1980–1995. *Environmental Health Perspectives*, 109(4):355–359.

Zanini G (2004). Caratteri del particolato e dei precursori, efficacia delle azioni di contrasto [Characteristics of particulate matter and precursors, efficacy of contrast policies]. *ARPA Rivista*, (5):18–19 (in Italian).

Zanobetti A, Schwartz J (2003a). Airborne particles and hospital admissions for heart and lung disease. In: *Revised analyses of time-series studies of air pollution and health*. Boston, Health Effects Institute: 241–248 (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006.

Zanobetti A, Schwartz J (2003b). Multicity assessment of mortality displacement within the APHEA2 project. In: *Revised analyses of time-series studies of air pollution and health*. Boston, Health Effects Institute: 249–254 (http://www.healtheffects.org/Pubs/TimeSeries.pdf, accessed 16 April 2006).

Zanobetti A, Schwartz J (2005). The effect of particulate air pollution on emergency admissions for myocardial infarction: a multicity case-crossover analysis. *Environmental Health Perspectives*, 113(8):978–982.

Zanobetti A et al. (2000). Generalized additive distributed lag models: quantifying mortality displacement. *Biostatistics*, 1(3):279–292.

Zanobetti A et al. (2003). The temporal pattern of respiratory and heart disease mortality in response to air pollution. *Environmental Health Perspectives*, 111(9):1188–1193.

Zeger SL, Dominici F, Samet J (1999). Harvesting-resistant estimates of air pollution effects on mortality. *Epidemiology*, 10(2):171–175.

Zeger SL et al. (2000). Exposure measurement error in time-series studies of air pollution: concepts and consequences. *Environment Health Perspectives*, 108(5):419–426.

Annex

The Annex has been structured as follows: respiratory- and cardiac-related hospital admissions are reported in Table 1; mortality from specific causes (by city and sex) are reported in Tables 2-6; the monitoring stations selected for PM_{10} and ozone are reported in Tables 7 and 8; the coefficient applied in Milan to correct PM_{10} TEOM data to gravimetric data is reported in Table 9; the city specific life expectancies applied in this study are reported in Tables 10 and 11; the SOMO0 annual average and three-year average are reported in Table 12; some detailed results by age group for PM_{10} are reported in Tables 13–16 (in terms of attributable deaths) and in Tables 17–20 (in terms of YLL). Detailed results by age group for ozone are reported in Tables 21 and 22 (in terms of attributable deaths) and in Tables 23 and 24 (in terms of YLL).

Table 1. Average annual number of respiratory- and cardiac-related hospital admissions for major Italian cities (1996–2002)

City	Respiratory-related hospital admissions	Cardiac-related hospital admissions
Turin	4583	6097
Genoa	4088	8091
Milan	8930	12648
Padua	1181	1986
Verona	1505	2120
Venice-Mestre	853	1904
Trieste	1783	3354
Bologna	3220	5720
Florence	3243	6054
Rome	21937	40266
Naples	11071	16751
Catania	1776	2937
Palermo	5461	5843
Total	**69630**	**113772**

Source: Data elaborated from MISA-2 study (2004).

Table 2. Mortality from lung cancer for major Italian cities by sex (2001)

City	Men	Women	Total
Turin	507	140	647
Genoa	384	114	498
Milan	730	285	1 015
Padua	102	56	158
Verona	129	47	176
Venice-Mestre	192	70	262
Trieste	135	44	179
Bologna	206	96	302
Florence	196	64	260
Rome	1334	454	1788
Naples	504	155	659
Catania	136	36	172
Palermo	231	68	299
Total	**4786**	**1629**	**6415**

Source: Cislaghi (2005).

Table 3. Mortality from infarction for major Italian cities by sex (2001)

City	Men	Women	Total
Turin	553	504	1 057
Genoa	419	423	842
Milan	929	885	1 814
Padua	140	137	277
Verona	171	214	385
Venice-Mestre	236	260	496
Trieste	218	359	577
Bologna	370	358	728
Florence	266	267	533
Rome	1912	1883	3795
Naples	693	688	1 381
Catania	225	180	405
Palermo	358	270	628
Total	**6490**	**6428**	**12 918**

Source: Cislaghi (2005).

Table 4. Mortality from stroke for major Italian cities by sex (2001)

City	Men	Women	Total
Turin	473	736	1209
Genoa	289	529	818
Milan	545	929	1474
Padua	73	144	217
Verona	80	117	197
Venice-Mestre	94	168	262
Trieste	119	217	336
Bologna	170	255	425
Florence	169	289	458
Rome	921	1352	2273
Naples	436	691	1127
Catania	178	265	443
Palermo	259	384	643
Total	**3806**	**6076**	**9882**

Source: Cislaghi (2005).

Table 5. Mortality from cardiovascular causes for major Italian cities by sex (2001)

City	Males	Females	Total
Turin	1 677	2 125	3 802
Genoa	1 294	1 922	3 216
Milan	2 246	3 067	5 313
Padua	318	475	793
Verona	435	615	1 050
Venice-Mestre	550	751	1 301
Trieste	505	774	1 279
Bologna	864	1 063	1 927
Florence	756	1 046	1 802
Rome	4 469	5 686	10 155
Naples	1 755	2 285	4 040
Catania	644	799	1 443
Palermo	1 045	1 291	2 336
Total	**16 558**	**21 899**	**38 457**

Source: Cislaghi (2005).

Table 6. Mortality from respiratory causes for major Italian cities by sex (2001)

City	Males	Females	Total
Turin	328	277	605
Genoa	246	195	441
Milan	471	487	958
Padua	89	88	177
Verona	70	69	139
Venice-Mestre	82	75	157
Trieste	110	119	229
Bologna	163	161	324
Florence	181	131	312
Rome	745	610	1355
Naples	369	223	592
Catania	103	72	175
Palermo	207	172	379
Total	**3164**	**2679**	**5843**

Source: Cislaghi (2005).

Table 7. Monitoring stations selected for the health impact assessment: PM_{10}

PM_{10}						
City	**Station**	**Station (type)**	**Zone (type)**	**Zone**	**Method**	**Data source**
Turin	Consolata	Traffic	Urban	R/C	Gravimetric	BRACE
	Gaidano	Background	Urban	R	Gravimetric	Regional Agency for Environmental Prevention and Protection in Piedmont
Genoa	Brignole	Traffic	Urban	R/C	Beta	BRACE
	Quarto	Background	Urban	R	Gravimetric	BRACE and Liguria Region
Milan	Juvara	Background	Urban	R	TEOM	BRACE
	Messina	Background	Urban	R	Gravimetric	BRACE and Regional Agency for Environmental Prevention and Protection in Lombardy
	Verziere	Traffic	Urban	R/C	TEOM	BRACE
Verona	Corso Milano	Traffic	Urban	R/C	Gravimetric	BRACE and Municipality of Verona
	San Giacomo	Traffic	Urban	R/C	Gravimetric	Municipality of Verona
	Piazza Bernardi[a]	Background	Urban	R	Gravimetric	Municipality of Verona
Venice-Mestre	Mestre Circonvallazione	Traffic	Urban	R/C	Gravimetric	Regional Agency for Environmental Prevention and Protection in Veneto
	Parco Bissuola	Background	Urban	R	Gravimetric	Regional Agency for Environmental Prevention and Protection in Veneto
Padua	Arcella	Traffic	Urban	R/C	Gravimetric	Regional Agency for Environmental Prevention and Protection in Veneto
	Mandria	Background	Urban	R	Gravimetric	Regional Agency for Environmental Prevention and Protection in Veneto
Trieste	Goldoni	Traffic	Urban	R/C	Beta	BRACE
	Libertà	Traffic	Urban	R	Beta	Giulia
	Tor Bandena	Traffic	Urban	R/C	Beta	BRACE
Bologna	Fiera	Traffic	Urban	C	Beta	BRACE and Regional Agency for Environmental Prevention and Protection in Emilia-Romagna
	San Felice	Traffic	Urban	R/C	Beta	BRACE
	Giardini Margherita[a]	Background	Urban	R	Beta	Regional Agency for Environmental Prevention and Protection in Emilia-Romagna
Florence	Bassi	Background	Urban	R/C	Beta	BRACE
	Boboli	Background	Urban	R/C	Beta	BRACE and Regional Agency for Environmental Prevention and Protection in Tuscany
	Mosse	Traffic	Urban	R/C	Beta	Regional Agency for Environmental Prevention and Protection in Tuscany
	Rosselli	Traffic	Urban	R/C	Beta	Regional Agency for Environmental Prevention and Protection in Tuscany
	Gramsci	Traffic	Urban	R/C	Beta	Regional Agency for Environmental Prevention and Protection in Tuscany

Table 7 (contd)

PM_{10}						
City	Station	Station (type)	Zone (type)	Zone	Method	Data source
Rome	Arenula	Traffic	Urban	R	Beta	BRACE
	Fermi	Traffic	Urban	R	Beta	BRACE
	Magna Grecia	Traffic	Urban	R	Beta	BRACE
	Villa Ada	Background	Urban	N	Beta	BRACE
Naples	Ente Ferrovie	Traffic	Urban	R/C	Beta	BRACE and Italian Agency for the Protection of the Environment and for Technical Services, Regional Agency for Environmental Prevention and Protection in Campania-Regional Centre for Air Pollution
	Scuola Vanvitelli	Traffic	Urban	R/C	Beta	Italian Agency for the Protection of the Environment and for Technical Services, Regional Agency for Environmental Prevention and Protection in Campania-Regional Centre for Air Pollution
	Policlinico	Traffic	Urban	R	Beta	Italian Agency for the Protection of the Environment and for Technical Services, Regional Agency for Environmental Prevention and Protection in Campania-Regional Centre for Air Pollution
	Ospedale Santo Bono	Traffic	Urban	R	Beta	BRACE and Italian Agency for the Protection of the Environment and for Technical Services, Regional Agency for Environmental Prevention and Protection in Campania-Regional Centre for Air Pollution
	Scuola Silio Italico	Traffic	Urban	R/C	Beta	BRACE and Italian Agency for the Protection of the Environment and for Technical Services, Regional Agency for Environmental Prevention and Protection in Campania-Regional Centre for Air Pollution
Catania	Europa	Traffic	Urban	R/C	Beta	Municipality of Catania
	Garibaldi	Traffic	Urban	R/C	Beta	Municipality of Catania
	Risorgimento	Traffic	Urban	R/C	Beta	Municipality of Catania
	Veneto	Traffic	Urban	R/C	Beta	Municipality of Catania
	Moro	Background	Urban	R/C	Beta	Municipality of Catania
Palermo	Belgio	Traffic	Urban	R/C	Beta	BRACE and Municipal Agency for Environmental Hygiene
	Castelnuovo	Traffic	Urban	R/C	Beta	BRACE and Municipal Agency for Environmental Hygiene
	Di Blasi	Traffic	Urban	R/C	Beta	BRACE and Municipal Agency for Environmental Hygiene
	Giulio Cesare	Traffic	Urban	R/C	Beta	BRACE and Municipal Agency for Environmental Hygiene
	Indipendenza	Traffic	Urban	R/C	Beta	BRACE and Municipal Agency for Environmental Hygiene
	Unità d'Italia	Traffic	Urban	R/C	Beta	BRACE and Municipal Agency for Environmental Hygiene

[a]TSP.
Note: R/C=residential/commercial; R=residential; N=natural; A=agricultural; C=commercial.

Table 8. Monitoring stations selected for the health impact assessment: ozone

Ozone					
City	Station	Station (type)	Zone (type)	Zone	Data source
Turin	Lingotto	Background	Urban	I/R	BRACE
Genoa	Acquasola	Background	Urban	R	BRACE
	Quarto	Background	Urban	R	BRACE
Milan	Juvara	Background	Urban	R	BRACE and Regional Agency for Environmental Prevention and Protection in Lombardy
Verona	Cason	Background	Rural	A	BRACE
Venice-Mestre	Maerne	Background	Urban	R/C	BRACE
	Parco Bissuola	Background	Urban	R	BRACE
	Sacca Fisola	Background	Urban	R	BRACE
Padua	Mandria	Background	Urban	R	BRACE and Regional Agency for Environmental Prevention and Protection in Veneto
Trieste	Monte San Pantaleone	Background	Suburban	I	BRACE and Regional Agency for Environmental Prevention and Protection in Friuli Venezia Giulia
Bologna	Giardini Margherita	Background	Urban	R	BRACE
Florence	Boboli	Background	Urban	R/C	BRACE
Rome	Villa Ada	Background	Urban	N	BRACE and Regional Agency for Environmental Prevention and Protection in Lazio
Naples	Osservatorio Astronomico	Background	Suburban	N	BRACE and Regional Agency for Environmental Prevention and Protection in Campania-Regional Centre for Air Pollution
Catania	Moro	Background	Urban	R/C	Municipality of Catania
Palermo	Boccadifalco	Background	Suburban	N	BRACE

Note: I/R=industrial/residential; R=residential; R/C=residential/commercial; A=agricultural; N=natural.

Table 9. A monthly conversion coefficient from TEOM to gravimetric for Milan

Month	Coefficient	Month	Coefficient	Month	Coefficient
January	1.35	May	1.09	September	1.09
February	1.33	June	1.02	October	1.17
March	1.26	July	1.00	November	1.26
April	1.18	August	1.02	December	1.33

Source: Regional Agency for Environmental Prevention and Protection in Lombardy (2005b).

Table 10. Life expectancy in major Italian cities by age group: males

Age group (years)	Life expectancy (years) by city												
	Turin	Genoa	Milan	Padua	Verona	Venice-Mestre	Trieste	Bologna	Florence	Rome	Naples	Catania	Palermo
< 1	77.41	76.71	77.13	78.62	78.38	76.63	75.97	78.27	77.94	76.92	74.56	75.13	76.24
1–4	76.78	75.94	76.57	77.81	77.86	75.98	75.06	77.51	77.37	76.30	74.18	74.92	75.70
5–9	72.87	71.97	72.63	73.81	73.86	71.98	71.16	73.62	73.43	72.33	70.20	70.97	71.87
10–14	67.91	67.01	67.67	68.81	68.86	66.98	66.16	68.68	68.49	67.39	65.26	66.08	66.90
15–19	62.99	62.09	62.72	63.81	63.86	61.98	61.16	63.74	63.60	62.45	60.34	61.15	61.97
20–24	58.15	57.23	57.88	58.97	58.92	57.35	56.33	58.85	58.89	57.63	55.51	56.38	57.07
25–29	53.31	52.46	53.06	54.20	54.05	52.58	51.52	53.89	54.03	52.84	50.66	51.57	52.24
30–34	48.52	47.69	48.26	49.37	49.11	47.83	46.86	49.05	49.13	48.02	45.80	46.88	47.43
35–39	43.74	42.86	43.43	44.59	44.28	43.00	42.15	44.14	44.40	43.24	41.12	42.12	42.68
40–44	39.05	38.10	38.71	39.81	39.53	38.37	37.41	39.44	39.77	38.48	36.39	37.30	37.97
45–49	34.34	33.48	34.08	34.99	35.02	33.72	32.85	34.85	35.02	33.85	31.80	32.71	33.33
50–54	29.76	28.91	29.51	30.23	30.47	29.30	28.31	30.29	30.46	29.32	27.23	28.25	28.72
55–59	25.39	24.53	25.15	25.85	25.97	25.06	23.96	25.85	26.09	24.92	22.99	23.89	24.31
60–64	21.14	20.42	20.94	21.67	21.65	20.91	19.93	21.63	21.88	20.75	19.07	19.74	20.09
65–69	17.14	16.55	17.05	17.90	17.65	17.07	16.30	17.59	17.81	16.86	15.32	15.97	16.13
70–74	13.42	13.06	13.57	14.26	13.98	13.53	12.99	13.91	14.37	13.27	12.19	12.49	12.76
75–79	10.17	9.97	10.42	11.02	10.78	10.83	10.08	10.59	11.11	10.22	9.37	9.78	9.86
80–84	7.66	7.48	7.93	8.28	8.37	8.35	7.52	8.12	8.41	7.74	7.17	7.19	7.32
85–89	5.26	5.30	5.77	5.98	6.02	5.80	5.35	5.87	6.19	5.45	5.47	4.94	5.38
90–94	3.72	3.86	4.32	4.18	4.31	4.29	3.88	4.34	4.19	3.90	4.45	3.73	3.63
≥95	2.51	2.79	3.14	3.56	2.71	2.62	2.45	3.30	2.65	3.03	3.53	3.60	2.41

Source: Data elaborated from ISTAT (2001) and Cislaghi (2005).

Table 11. Life expectancy in major Italian cities by age group: females

Age group (years)	Life expectancy (years) by city												
	Turin	Genoa	Milan	Padua	Verona	Venice-Mestre	Trieste	Bologna	Florence	Rome	Naples	Catania	Palermo
< 1	83.00	82.23	82.99	82.88	83.81	83.03	82.71	83.42	84.27	82.61	80.52	80.79	80.99
1–4	82.31	81.62	82.26	82.50	83.04	82.11	81.83	82.48	83.46	81.97	80.08	80.43	80.71
5–9	78.38	77.62	78.33	78.61	79.12	78.20	77.83	78.55	79.46	78.05	76.11	76.54	76.80
10–14	73.41	72.66	73.36	73.71	74.12	73.20	72.83	73.62	74.46	73.07	71.13	71.67	71.84
15–19	68.46	67.73	68.38	68.71	69.26	68.28	68.04	68.68	69.52	68.12	66.16	66.67	66.86
20–24	63.52	62.82	63.46	63.81	64.26	63.42	63.04	63.87	64.58	63.16	61.19	61.77	61.88
25–29	58.62	57.89	58.51	58.81	59.26	58.47	58.11	58.95	59.62	58.23	56.25	56.85	56.94
30–34	53.72	53.02	53.56	53.85	54.42	53.51	53.11	54.15	54.69	53.33	51.31	51.94	52.04
35–39	48.78	48.18	48.68	48.94	49.47	48.63	48.11	49.24	49.77	48.43	46.38	47.07	47.13
40–44	43.93	43.33	43.85	44.14	44.49	43.76	43.28	44.32	44.87	43.58	41.54	42.22	42.29
45–49	39.15	38.63	39.08	39.39	39.61	38.96	38.45	39.56	40.18	38.81	36.76	37.49	37.47
50–54	34.53	33.99	34.41	34.54	34.90	34.39	33.94	34.87	35.47	34.11	32.08	32.79	32.74
55–59	29.94	29.38	29.89	30.08	30.35	29.75	29.30	30.26	30.83	29.49	27.56	28.32	28.09
60–64	25.49	24.95	25.42	25.63	25.81	25.36	24.86	25.82	26.20	24.89	23.19	23.91	23.53
65–69	21.21	20.62	21.05	21.25	21.45	21.01	20.60	21.46	21.80	20.51	18.97	19.55	19.28
70–74	17.06	16.51	16.95	17.17	17.31	16.88	16.52	17.27	17.63	16.35	15.08	15.51	15.28
75–79	13.28	12.67	13.10	13.36	13.27	13.27	12.83	13.48	13.68	12.54	11.69	11.90	11.60
80–84	9.93	9.31	9.63	10.01	9.73	10.13	9.60	9.86	10.04	9.19	8.78	8.78	8.28
85–89	6.87	6.57	6.61	6.97	6.75	7.19	6.48	7.07	6.92	6.35	6.19	6.04	5.63
90–94	4.77	4.50	4.26	4.74	4.72	4.93	4.59	5.09	4.75	4.34	4.41	4.11	3.95
≥95	3.29	3.09	1.90	3.28	2.74	3.21	2.99	3.41	3.60	2.74	3.15	3.46	2.61

Source: Data elaborated from ISTAT (2001) and Cislaghi (2005).

Table 12. Period and global averages for ozone: SOMO0

City	Annual concentration by year			Average
	2002	2003	2004	
Turin	74.2	87.9	77.3	79.8
Genoa	83.0	85.3	75.9	81.4
Milan	53.4	61.8	33.8	49.6
Padua	69.4	92.9	80.4	80.9
Verona	36.0	86.3	73.5	65.3
Venice-Mestre	47.0	82.0	64.2	64.4
Trieste	60.1	60.1	85.9	68.7
Bologna	53.8	67.9	67.9	63.2
Florence	75.5	77.8	63.5	72.3
Rome	61.4	81.4	76.6	73.1
Naples	83.2	65.6	96.0	81.6
Catania	62.8	47.7	46.8	52.4
Palermo	107.1	112.1	88.9	102.7

Source: Annex Table 8.

Table 13. Deaths attributable to levels of PM_{10} exceeding 20 μg/m^3

Causes of death	Age group (years) deaths								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	96	339	686	1338	1450	**3909**
Lung cancer[a]	0	0	0	5	64	154	232	95	**551**
Infarction[a]	0	0	0	16	102	214	443	518	**1293**
Stroke[a]	0	0	0	1	6	13	40	66	**126**
Acute mortality (excluding accidents)	4	1	3	16	56	114	221	240	**654**
Cardiovascular diseases	0	0	1	5	25	48	117	166	**362**
Respiratory diseases	0	0	0	1	3	9	32	54	**99**
Females									
Mortality (excluding accidents)[a]	0	0	0	57	217	416	1044	2576	**4311**
Lung cancer[a]	0	0	0	4	26	41	69	52	**191**
Infarction[a]	0	0	0	3	19	73	294	880	**1269**
Stroke[a]	0	0	0	1	4	10	40	149	**203**
Acute mortality (excluding accidents)	3	0	1	9	36	69	173	427	**718**
Cardiovascular diseases	0	0	0	2	8	25	96	349	**481**
Respiratory diseases	0	0	0	0	1	4	19	61	**86**
Total									
Mortality (excluding accidents)[a]	0	0	0	153	556	1103	2382	4026	**8220**
Lung cancer[a]	0	0	0	9	90	195	301	147	**742**
Infarction[a]	0	0	0	19	122	287	737	1398	**2562**
Stroke[a]	0	0	0	2	10	23	80	214	**329**
Acute mortality (excluding accidents)	6	1	5	25	92	182	394	666	**1372**
Cardiovascular diseases	0	0	1	8	33	74	212	514	**843**
Respiratory diseases	0	0	0	2	5	13	51	115	**186**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 14. Deaths attributable to levels of PM_{10} exceeding 30 µg/m^3

Causes of death	Age group (years) deaths								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	59	211	432	843	919	**2465**
Lung cancer[a]	0	0	0	3	40	99	149	62	**354**
Infarction[a]	0	0	0	10	67	139	293	342	**851**
Stroke[a]	0	0	0	1	4	8	24	41	**78**
Acute mortality (excluding accidents)	2	0	2	10	34	70	136	148	**401**
Cardiovascular diseases	0	0	1	3	15	29	71	102	**221**
Respiratory diseases	0	0	0	1	2	5	19	33	**61**
Females									
Mortality (excluding accidents)[a]	0	0	0	36	137	262	653	1644	**2731**
Lung cancer[a]	0	0	0	2	16	27	45	34	**124**
Infarction[a]	0	0	0	2	12	47	192	581	**833**
Stroke[a]	0	0	0	1	2	6	24	92	**125**
Acute mortality (excluding accidents)	2	0	1	6	22	42	105	265	**443**
Cardiovascular diseases	0	0	0	1	5	15	58	215	**295**
Respiratory diseases	0	0	0	0	1	3	12	39	**54**
Total									
Mortality (excluding accidents)[a]	0	0	0	95	347	695	1497	2563	**5196**
Lung cancer[a]	0	0	0	6	56	125	194	96	**478**
Infarction[a]	0	0	0	12	79	186	484	923	**1684**
Stroke[a]	0	0	0	2	6	14	48	133	**203**
Acute mortality (excluding accidents)	4	1	3	15	56	112	241	413	**844**
Cardiovascular diseases	0	0	1	5	20	45	129	317	**516**
Respiratory diseases	0	0	0	1	3	8	31	72	**115**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 15. Deaths attributable to levels of PM_{10} exceeding 40 μg/m^3

Causes of death	Age group (years) deaths								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	24	88	187	365	404	**1068**
Lung cancer[a]	0	0	0	1	17	44	67	29	**157**
Infarction[a]	0	0	0	4	28	60	132	153	**376**
Stroke[a]	0	0	0	0	2	3	10	18	**33**
Acute mortality (excluding accidents)	1	0	1	4	14	30	57	64	**170**
Cardiovascular diseases	0	0	0	1	6	12	30	43	**93**
Respiratory diseases	0	0	0	0	1	2	8	15	**26**
Females									
Mortality (excluding accidents)[a]	0	0	0	15	60	114	281	732	**1202**
Lung cancer[a]	0	0	0	1	7	12	20	16	**56**
Infarction[a]	0	0	0	1	4	19	84	264	**373**
Stroke[a]	0	0	0	0	1	3	10	41	**55**
Acute mortality (excluding accidents)	1	0	0	2	9	18	44	116	**191**
Cardiovascular diseases	0	0	0	1	2	6	24	92	**125**
Respiratory diseases	0	0	0	0	0	1	5	18	**25**
Total									
Mortality (excluding accidents)[a]	0	0	0	39	148	301	646	1136	**2270**
Lung cancer[a]	0	0	0	2	24	56	87	45	**214**
Infarction[a]	0	0	0	5	32	79	216	417	**749**
Stroke[a]	0	0	0	1	3	6	20	59	**88**
Acute mortality (excluding accidents)	1	0	1	6	23	48	102	179	**361**
Cardiovascular diseases	0	0	0	2	8	18	54	135	**218**
Respiratory diseases	0	0	0	0	1	3	13	32	**51**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 16. Deaths attributable to levels of PM_{10} being reduced by 10%

Causes of death	Age group (years) deaths								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	19	67	135	262	283	**766**
Lung cancer[a]	0	0	0	1	13	31	47	19	**111**
Infarction[a]	0	0	0	4	23	49	101	118	**295**
Stroke[a]	0	0	0	0	1	2	7	12	**23**
Acute mortality (excluding accidents)	1	0	1	3	11	21	41	45	**123**
Cardiovascular diseases	0	0	0	1	5	9	21	30	**66**
Respiratory diseases	0	0	0	0	1	2	6	10	**18**
Females									
Mortality (excluding accidents)[a]	0	0	0	11	42	82	206	502	**843**
Lung cancer[a]	0	0	0	1	5	8	14	10	**38**
Infarction[a]	0	0	0	1	5	17	68	201	**291**
Stroke[a]	0	0	0	0	1	2	7	27	**38**
Acute mortality (excluding accidents)	1	0	0	2	7	13	32	80	**135**
Cardiovascular diseases	0	0	0	0	2	5	18	64	**88**
Respiratory diseases	0	0	0	0	0	1	3	11	**16**
Total									
Mortality (excluding accidents)[a]	0	0	0	30	109	216	468	786	**1610**
Lung cancer[a]	0	0	0	2	18	39	60	29	**149**
Infarction[a]	0	0	0	4	28	66	168	320	**586**
Stroke[a]	0	0	0	0	2	4	15	39	**61**
Acute mortality (excluding accidents)	1	0	1	5	17	34	74	125	**258**
Cardiovascular diseases	0	0	0	1	6	14	39	94	**154**
Respiratory diseases	0	0	0	0	1	2	9	21	**34**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 17. YLL by sex, age group and scenario (PM_{10} reduced to 20 µg/m^3)

Causes of death	Age group (years)								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	3778	8 696	11 318	13 725	7 795	**45 311**
Lung cancer[a]	0	0	0	204	1 612	2 544	2 422	556	**7 339**
Infarction[a]	0	0	0	605	2 619	3 526	4 519	2 794	**14 062**
Stroke[a]	0	0	0	56	157	213	404	346	**1 176**
Acute mortality (excluding accidents)	269	47	185	624	1 436	1 873	2 271	1 290	**7 995**
Cardiovascular diseases	15	7	57	208	633	796	1 187	879	**3 782**
Respiratory diseases	14	5	10	48	81	142	323	281	**904**
Females									
Mortality (excluding accidents)[a]	0	0	0	2526	6 620	8 408	13 006	14 279	**44 840**
Lung cancer[a]	0	0	0	155	783	831	876	321	**2 966**
Infarction[a]	0	0	0	154	580	1 428	3 626	4 867	**10 656**
Stroke[a]	0	0	0	41	122	198	485	809	**1 655**
Acute mortality (excluding accidents)	233	32	85	417	1 096	1 391	2 151	2 364	**7 770**
Cardiovascular diseases	22	10	12	97	259	503	1 174	1 890	**3 967**
Respiratory diseases	11	8	9	17	44	84	233	330	**737**
Total									
Mortality (excluding accidents)[a]	0	0	0	6304	15 316	19 725	26 731	22 074	**90 151**
Lung cancer[a]	0	0	0	359	2 395	3 375	3 299	877	**10 305**
Infarction[a]	0	0	0	760	3 199	4 954	8 144	7 661	**24 718**
Stroke[a]	0	0	0	97	279	411	889	1 155	**2 832**
Acute mortality (excluding accidents)	502	80	270	1041	2 532	3 264	4 422	3 654	**15 764**
Cardiovascular diseases	37	17	69	306	892	1 299	2 361	2 769	**7 749**
Respiratory diseases	26	13	19	65	125	226	556	611	**1 641**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 18. YLL by sex, age group and scenario (PM_{10} reduced to 30 μg/m^3)

Causes of death	Age group (years)								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	2346	5429	7 166	8 690	4 956	**28 587**
Lung cancer[a]	0	0	0	128	1010	1 636	1 563	365	**4 703**
Infarction[a]	0	0	0	388	1711	2 309	3 000	1 851	**9 258**
Stroke[a]	0	0	0	36	96	130	246	215	**723**
Acute mortality (excluding accidents)	158	26	108	377	873	1 154	1 399	798	**4 894**
Cardiovascular diseases	9	3	35	127	383	486	728	541	**2 313**
Respiratory diseases	10	3	5	31	48	87	198	175	**557**
Females									
Mortality (excluding accidents)[a]	0	0	0	1580	4186	5 330	8 192	9 106	**28 393**
Lung cancer[a]	0	0	0	95	504	543	570	211	**1 924**
Infarction[a]	0	0	0	102	360	923	2 375	3 213	**6 973**
Stroke[a]	0	0	0	26	73	122	296	502	**1 020**
Acute mortality (excluding accidents)	135	19	49	254	674	859	1 319	1 467	**4 776**
Cardiovascular diseases	12	5	7	59	155	306	714	1 166	**2 424**
Respiratory diseases	5	6	6	9	27	53	146	209	**461**
Total									
Mortality (excluding accidents)[a]	0	0	0	3926	9615	12 496	16 882	14 062	**56 980**
Lung cancer[a]	0	0	0	223	1514	2 179	2 134	577	**6 627**
Infarction[a]	0	0	0	490	2071	3 232	5 375	5 064	**16 231**
Stroke[a]	0	0	0	63	169	253	542	717	**1 743**
Acute mortality (excluding accidents)	293	45	158	631	1547	2 013	2 718	2 265	**9 670**
Cardiovascular diseases	22	9	42	185	538	791	1 442	1 707	**4 737**
Respiratory diseases	15	9	11	40	76	140	344	384	**1 018**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 19. YLL by sex, age group and scenario (PM_{10} reduced to 40 µg/m^3)

Causes of death	Age group (years)								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	970	2290	3139	3791	2186	**12 376**
Lung cancer[a]	0	0	0	52	421	730	704	171	**2 079**
Infarction[a]	0	0	0	151	716	1002	1363	835	**4 068**
Stroke[a]	0	0	0	17	41	55	102	95	**310**
Acute mortality (excluding accidents)	62	8	43	153	361	495	598	345	**2 064**
Cardiovascular diseases	4	0	15	52	157	203	307	230	**968**
Respiratory diseases	5	2	1	14	19	37	83	77	**239**
Females									
Mortality (excluding accidents)[a]	0	0	0	659	1849	2344	3578	4050	**12 480**
Lung cancer[a]	0	0	0	35	224	248	263	100	**870**
Infarction[a]	0	0	0	44	138	391	1057	1453	**3 083**
Stroke[a]	0	0	0	12	31	52	126	222	**443**
Acute mortality (excluding accidents)	49	8	18	104	292	370	564	638	**2 043**
Cardiovascular diseases	5	2	2	23	63	128	300	499	**1 022**
Respiratory diseases	1	3	2	3	12	23	65	95	**205**
Total									
Mortality (excluding accidents)[a]	0	0	0	1629	4139	5484	7369	6235	**24 856**
Lung cancer[a]	0	0	0	87	645	978	968	271	**2 949**
Infarction[a]	0	0	0	195	854	1393	2420	2288	**7 150**
Stroke[a]	0	0	0	29	71	107	228	316	**752**
Acute mortality (excluding accidents)	112	16	60	257	653	865	1162	983	**4 107**
Cardiovascular diseases	9	2	17	75	220	330	608	729	**1 991**
Respiratory diseases	6	5	3	18	31	61	148	172	**444**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 20. YLL by sex, age group and scenario (PM_{10} reduced by 10%)

Causes of death	Age group (years)								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Mortality (excluding accidents)[a]	0	0	0	746	1713	2214	2683	1521	**8 876**
Lung cancer[a]	0	0	0	41	327	510	484	110	**1 473**
Infarction[a]	0	0	0	139	597	806	1027	637	**3 205**
Stroke[a]	0	0	0	10	29	40	75	64	**218**
Acute mortality (excluding accidents)	52	10	35	118	271	349	425	243	**1 503**
Cardiovascular diseases	3	1	10	38	117	146	217	160	**693**
Respiratory diseases	2	1	2	9	15	26	60	51	**167**
Females									
Mortality (excluding accidents)[a]	0	0	0	496	1295	1643	2552	2784	**8 769**
Lung cancer[a]	0	0	0	32	156	165	175	64	**591**
Infarction[a]	0	0	0	35	136	327	830	1114	**2 442**
Stroke[a]	0	0	0	7	23	37	90	149	**306**
Acute mortality (excluding accidents)	44	6	16	79	204	259	403	446	**1 458**
Cardiovascular diseases	4	2	2	18	48	93	216	345	**728**
Respiratory diseases	2	1	2	4	8	15	43	60	**135**
Total									
Mortality (excluding accidents)[a]	0	0	0	1242	3008	3856	5235	4305	**17 646**
Lung cancer[a]	0	0	0	73	484	675	659	174	**2 064**
Infarction[a]	0	0	0	174	732	1133	1857	1751	**5 647**
Stroke[a]	0	0	0	17	52	76	165	213	**524**
Acute mortality (excluding accidents)	96	16	51	197	474	609	829	689	**2 961**
Cardiovascular diseases	7	3	13	56	165	239	433	505	**1 421**
Respiratory diseases	5	2	4	12	23	42	102	111	**302**

[a]Risk based on $PM_{2.5}$ estimates (chronic effects).

Table 21. Deaths attributable to ozone: SOMO35

Causes of death	Age group (years) deaths								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Acute mortality (excluding accidents)	1	0	1	6	22	43	83	89	**247**
Acute mortality, cardiovascular causes	0	0	0	2	9	18	43	59	**132**
Females									
Acute mortality (excluding accidents)	1	0	1	4	13	26	66	158	**269**
Acute mortality, cardiovascular causes	0	0	0	1	3	10	35	122	**172**
Total									
Acute mortality (excluding accidents)	3	0	2	10	35	69	149	247	**516**
Acute mortality, cardiovascular causes	0	0	0	3	13	27	78	181	**303**

Table 22. Deaths attributable to ozone: SOMO0

Causes of death	Age group (years) deaths								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Acute mortality (excluding accidents)	5	1	5	22	80	158	306	328	**908**
Acute mortality, cardiovascular causes	0	0	1	8	34	65	156	217	**482**
Females									
Acute mortality (excluding accidents)	5	1	2	13	49	96	243	581	**989**
Acute mortality, cardiovascular causes	0	0	0	3	13	35	131	447	**630**
Total									
Acute mortality (excluding accidents)	10	2	8	36	130	254	549	909	**1897**
Acute mortality, cardiovascular causes	1	0	2	11	47	101	287	664	**1112**

Table 23. Years of life lost due to ozone: SOMO35

Causes of death	Age group (years)								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Acute mortality (excluding accidents)	110	20	74	240	556	702	846	474	**3023**
Acute mortality, cardiovascular causes	8	4	19	84	237	292	430	311	**1386**
Females									
Acute mortality (excluding accidents)	103	13	34	157	409	522	809	873	**2921**
Acute mortality, cardiovascular causes	8	4	6	40	105	191	432	664	**1450**
Total									
Acute mortality (excluding accidents)	213	33	108	398	965	1225	1655	1348	**5944**
Acute mortality, cardiovascular causes	16	7	26	123	342	483	863	975	**2835**

Table 24. Years of life lost due to ozone: SOMO0

Causes of death	Age group (years)								Total
	0–4	5–14	15–29	30–44	45–59	60–69	70–79	≥80	
Males									
Acute mortality (excluding accidents)	403	77	281	882	2041	2579	3109	1750	**11 123**
Acute mortality, cardiovascular causes	29	13	72	300	865	1069	1576	1144	**5 067**
Females									
Acute mortality (excluding accidents)	375	49	128	585	1499	1904	2976	3215	**10 731**
Acute mortality, cardiovascular causes	31	14	23	144	385	701	1588	2439	**5 324**
Total									
Acute mortality (excluding accidents)	779	126	410	1467	3539	4484	6086	4964	**21 854**
Acute mortality, cardiovascular causes	61	27	95	443	1250	1769	3164	3583	**10 391**