Science of the Total Environment 449 (2013) 390–400

Contents lists available at SciVerse ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Assessing the public health impacts of urban air pollution in 25 European cities: Results of the Aphekom project ☆

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HIGHLIGHTS

► Aphekom performed health impact assessments of urban air pollution in Europe.
► Improving air quality would result in significant health and monetary gains.
► $PM_{2.5}$ annual mean to 10 μg/m^3 could add more than 6 months of life expectancy at age 30 in half of the cities.
► The associated costs would reach 30 billion Euros annually.

ARTICLE INFO

Article history:
Received 9 July 2012
Received in revised form 11 November 2012
Accepted 16 January 2013
Available online xxxx

Keywords:
Air pollution
Public health
Particulate matter
Ozone

ABSTRACT

Introduction: The Aphekom project aimed to provide new, clear, and meaningful information on the health effects of air pollution in Europe. Among others, it assessed the health and monetary benefits of reducing short and long-term exposure to particulate matter (PM) and ozone in 25 European cities.
Method: Health impact assessments were performed using routine health and air quality data, and a common methodology. Two scenarios were considered: a decrease of the air pollutant levels by a fixed amount and a decrease to the World Health Organization (WHO) air quality guidelines. Results were economically valued by using a willingness to pay approach for mortality and a cost of illness approach for morbidity.
Results: In the 25 cities, the largest health burden was attributable to the impacts of chronic exposure to $PM_{2.5}$. Complying with the WHO guideline of 10 μg/m^3 in annual mean would add up to 22 months of life expectancy at age 30, depending on the city, corresponding to a total of 19,000 deaths delayed. The associated monetary gain would total some €31 billion annually, including savings on health expenditures, absenteeism and intangible costs such as well-being, life expectancy and quality of life.
Conclusion: European citizens are still exposed to concentrations exceeding the WHO recommendations. Aphekom provided robust estimates confirming that reducing urban air pollution would result in significant health and monetary gains in Europe. This work is particularly relevant now when the current EU legislation is being revised for an update in 2013.

Abbreviations: Aphekom, Improving Knowledge and Communication for Decision Making on Air Pollution and Health in Europe; CI, Confidence Interval; CRF, Concentration response function; HIA, Health Impact Assessment; O_3, Ozone; PM, Particulate Matter; PM_{10}, Particulate Matter with an aerodynamic diameter below 10 μm; $PM_{2.5}$, Particulate Matter with an aerodynamic diameter below 2.5 μm; RR, Relative Risk; TEOM, Tapered Element Oscillating Microbalance; VSL, Value of a Statistical Life; WHO-AQG, Air quality guidelines developed by the World Health Organization.

☆ Funding: The Aphekom project has been co-funded by the European Commission's Programme on community Action in the Field of Public Health (2003–2008) under Grant Agreement No. 2007105, and by the many national and local institutions that have dedicated resources to the fulfilment of this project.

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http://dx.doi.org/10.1016/j.scitotenv.2013.01.077

1. Introduction

Urban air quality represents a major public health burden and is a long-standing concern to European citizens. Despite a major decrease in the pollutant levels in Europe since the 1950s and the implementation of the first European Commission Directive on Ambient Air Quality in 1980, regularly updated since then, important disparities in exposure to air pollution between and within European countries still remain.

Air pollution is associated with a range of diseases, symptoms and infraclinic conditions that impair the health and quality of life in European cities. In the recent years, several epidemiological studies have reported associations between an increase in daily levels of ozone (O_3) and particulate matter (PM), and an increase in the following days, of the mortality and hospital admissions predominantly related to respiratory and cardiovascular diseases. These short-term effects have been extensively documented in multicentre time-series studies (Anderson et al., 2004; Atkinson et al., 2005; Ballester et al., 2006; Bell et al., 2004, 2005; Biggeri et al., 2005; Dominici et al., 2005; Faustini et al., 2011; Garrett and Casimiro, 2011; Gryparis et al., 2004; Ito et al., 2005; Le Tertre et al., 2002; Saez et al., 2002; Stafoggia et al., 2009), producing robust estimates for Europe and North America. Chronic exposure to fine particles ($PM_{2.5}$) has also been associated with an increase in long-term mortality, and with an increased risk of developing lung cancer and cardio-pulmonary diseases (myocardial infarction, chronic obstructive pulmonary disease, asthma) (Brook et al., 2010; Jerrett et al., 2005; Krewski et al., 2009; Pope et al., 2002, 2004). There is less conclusive evidence on the effect of chronic exposure to ozone, although Jerrett et al. linked long-term respiratory mortality with exposure to ozone during summer (Jerrett et al., 2009a). The relation between exposure to ozone, particulate matter and specific health outcomes is supported by the consistency of epidemiological findings across different cities, periods and study designs; the coherence of the observed effects; the indication of an increased risk at higher exposure levels; and the biological plausibility strengthened by clinical and toxicological studies. In particular, several results are in favour of a causal relationship between chronic exposure to $PM_{2.5}$ and cardiovascular morbidity and mortality (Brook et al., 2010; Chen et al., 2008; Pope and Dockery, 2006).So far, threshold levels for no observable health effects have not been identified (World Health Organisation, 2005).

However, current European air quality standards for PM and ozone are still above the World Health Organization Air Quality Guidelines (WHO-AQG) that aim to protect public health. In Europe, annual mean PM_{10} should not exceed 40 μg/m^3 (limit value set in 2005), and Member States are requested to reduce exposure to $PM_{2.5}$ in urban areas below 20 μg/m^3 by 2015 (legally binding value). The WHO-AQG for PM, chosen as the lowest levels at which total, cardiopulmonary and lung cancer mortality have been shown to significantly increase in response to long-term exposure to PM are set as an annual mean of 20 μg/m^3 for PM_{10} and 10 μg/m^3 for $PM_{2.5}$. For ozone, the EU air quality directive still refers to the previous WHO-AQG of 120 μg/m^3 (8-hour mean) (Air Quality Directive, 2008/50/EC). This value should not be exceeded more than 25 days per calendar year. The updated WHO-AQG, chosen as the concentration associated with a 1–2% increase in daily mortality, correspond to 100 μg/m^3 for the maximum daily 8-hour O_3 mean (World Health Organisation, 2005).

Several health impact assessments (HIA) have already reported the major public health burden of PM and ozone in Europe (Ballester et al., 2008; Boldo et al., 2006; Kunzli et al., 2000; Watkiss et al., 2005; World Health Organisation, 2010). In this paper, we present new HIA for 25 European cities, using recent data and new epidemiological knowledge on the impacts of PM and ozone on mortality and hospitalizations.

Since stakeholders drafting policies to reduce air pollution must take into account many considerations, such as economic and social constraints, political orientations and urban planning, the paper also presents an economic valuation of the estimated health gains from reducing air pollution levels in European cities, and an analysis of the overall uncertainties.

These analyses were part of the European project Aphekom, whose objective was to improve knowledge and to develop tools to better assess and communicate the health benefits from an improvement in urban air quality in Europe.

2. Methods

2.1. Study period and study areas

The HIA were performed in the 25 European cities from 12 countries participating in the Aphekom project (Fig. 1). A common study period, 2004–2006, was chosen based on data availability. In each city, a study area was defined according to a common protocol and with the advice of local experts in order to ensure that average pollutant levels measured at fixed monitors could be considered good proxies of the average population exposure.

2.2. Choice of health endpoints of the HIA

Health endpoints were chosen based on available concentration response functions (CRF) in the literature on short and long-term effects of ozone, PM_{10} and $PM_{2.5}$ and on data availability in all cities. We decided to perform HIA for the short-term impacts of ozone on respiratory hospitalizations, the long-term impacts of ozone on respiratory mortality, the short-term impacts of PM_{10} on cardiac and respiratory hospitalizations and the long-term impacts of $PM_{2.5}$ on total and cardiovascular mortality. CRF used in these HIA are summarized in Table 1. Regarding the assessment of the long-term impacts, the American Cancer Society Cancer Prevention Study II estimated a relative risk (RR) of 1.040 [95% CI: 1.010–1.067] (Jerrett et al., 2009a) for a 20 μg/m^3 increase in the average 1-hour maximum ozone levels during summer months. We converted this estimate to obtain an RR value for a 10 μg/m^3 increase in the maximum daily 8-hour O_3 mean during summer months, by using the 8-hour/1-hour ratio of 0.88 observed in the original cohort (Jerrett et al., 2009b).

HIA of the short-term impacts of ozone and PM_{10} on mortality were also performed and results are available from the authors.

2.3. Population and health data

Population and health data were collected from the relevant authorities in each country. Mortality data were selected on the main cause of death of the residents living in the study area regardless of the place of death. Hospitalization data were collected from public and private hospitals within the study area or outside the area, but attracting a large proportion of residents from the study area. Hospitalization data were not available in Athens, Budapest and Dublin.

2.4. Exposure assessment

In each study area, data were provided by the local air quality monitoring networks, who advised on the choice of the relevant monitoring stations. Criteria to select the monitors included its representativity, the number of missing value (<25%), and the consistency with the other selected monitors (overlap of the interquartile range between monitors, and Pearson correlation coefficient above 0.6). Depending on the city, from 1 to 13 monitors were available for ozone, from 1 to 9 for PM_{10} and from 0 to 4 for $PM_{2.5}$ (Table 4).

In all cities, ozone was measured using the standard reference ultraviolet absorption method (World Health Organisation, 2008). Ozone exposure indicator was the daily maximum 8-hour mean(daily maximum of the 8-hour running means) of all selected stations,

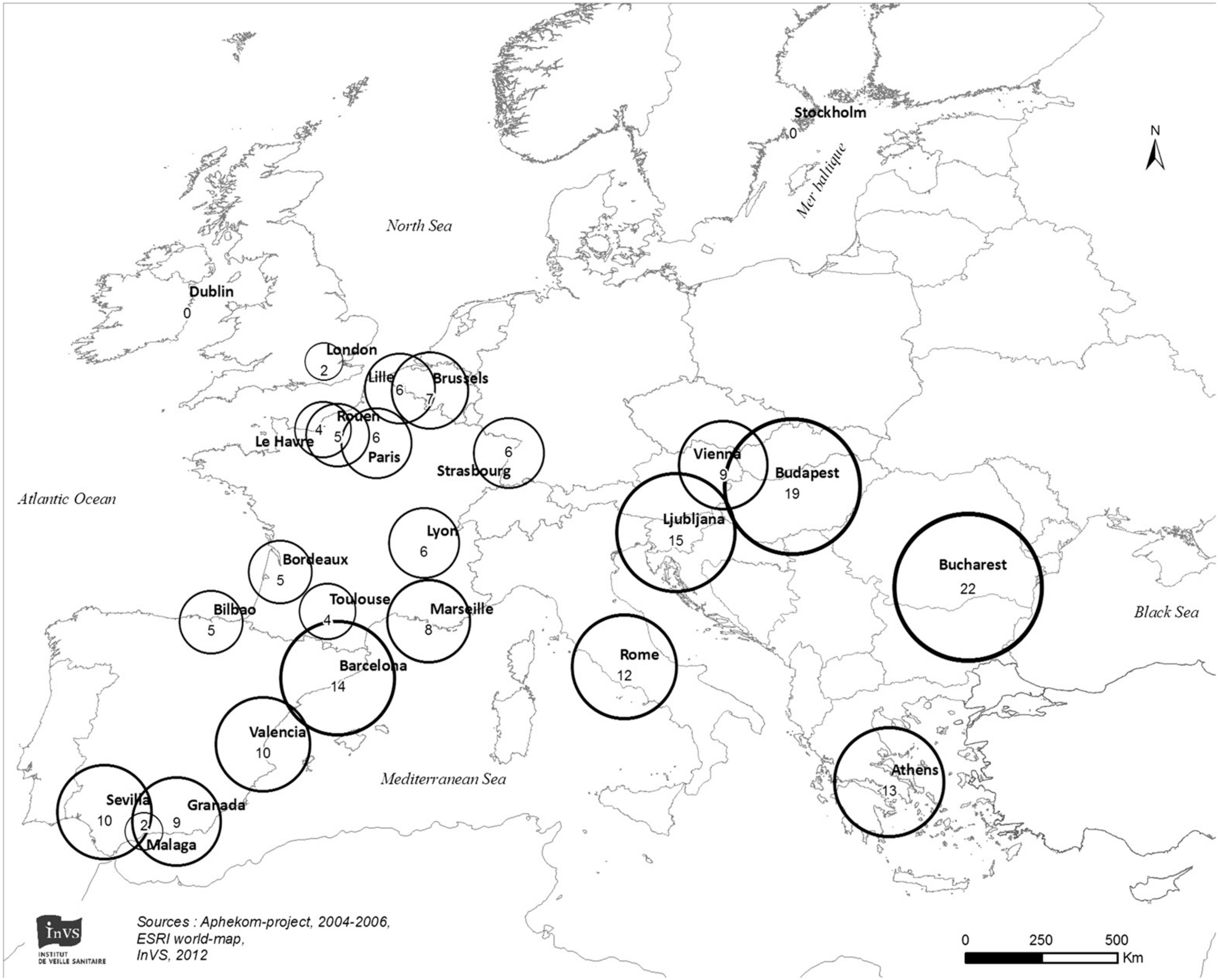

Fig. 1. Gain in life expectancy in months at 30 if $PM_{2.5}$ concentrations did not exceed the WHO-AQG (10 μg/m^3).

Table 1
Health outcomes and associated concentration-response functions, expressed as relative risk (RR) per 10 μg/m^3 increase in each pollutant used in the HIA.

Health outcome	ICD codes	Ages	RR per 10 μg/m^3	Reference
Short-term impacts of ozone				
Respiratory hospitalisations	ICD9 460-519 IDC10 J00-J99	15–64	1.001 [0.991–1.012]	Anderson et al. (2004)
Respiratory hospitalisations	ICD9 460-519 IDC10 J00-J99	≥65	1.005 [0.998–1.012]	Anderson et al. (2004)
Short-term impacts of PM_{10}				
Respiratory hospitalisations	ICD9 460-519 IDC10 J00-J199	All	1.011 [1.006–1.017]	Medina et al. (2005)
Cardiac hospitalisations	ICD9 390-429 ICD10 I00-I52	All	1.006 [1.003–1.009]	Medina et al. (2005)
Long-term impacts of ozone				
Respiratory mortality	ICD9 460-519 IDC10 J00-J99	>30	1.023 [1.006–1.038]	Derived from Jerrett et al. (2009a)
Long-term impacts of $PM_{2.5}$				
Total (including external) mortality	ICD9 000-999 ICD10 A00-Y98	>30	1.06 [1.02–1.11]	Pope et al. (2002)
Cardiovascular mortality	ICD9 390-429 ICD10 I00-I99	>30	1.12 [1.08-1.15]	Pope et al. (2004)

including only those years with less than 25% missing data, as outlined in the EU Directive on the assessment of air pollution (Directive 2008/50/EC of the European Parliament and of the Council). Consistently with the studies providing the CRF, short-term HIA were performed using ozone data for the whole year, while long-term HIA were carried out using ozone data from April to September.

Regarding PM measurements, 13 cities used the Tapered Element Oscillating Microbalance (TEOM) method, 9 cities used the Beta-attenuation method and only 3 cities used the reference gravimetric method. TEOM measurements were corrected to compensate losses of semi-volatile compounds, using either a real-time correction, a fixed local correction factor if available, or the value of 1.3 recommended by the European Commission working group on PM (Williams and Bruckmann, 2002). For Athens, Budapest, Dublin, Granada, Ljubljana, Malaga, Seville and Valencia, $PM_{2.5}$ measures were not available, and exposure to $PM_{2.5}$ was estimated from PM_{10} using a 0.7 conversion factor, consistently with the method previously defined in the Apheis project (Medina et al., 2005). PM exposure indicators were the annual mean concentrations, calculated from the daily concentrations of the stations, again including only years with less than 25% daily missing data.

Detailed protocols on the definition of the study area, the choice of the monitors, and the construction of the exposure indicators are available on request to the authors.

2.5. Calculation of the health impacts

We assessed the health benefits that could be obtained if pollutant concentrations were lowered, all other things being equal. For the short-term exposure to PM_{10} and ozone, the health impact was computed as:

$$\Delta y = y_0\left(1 - e^{-\beta\Delta x}\right)$$

where:

- Δy is the decrease in the health outcome associated with the decrease in pollutant concentrations, in annual number of deaths or hospitalizations.
- y_0 is the baseline health outcome, in annual number of deaths or hospitalizations.
- β is the coefficient of the concentration response function.
- Δx is the decrease in the pollutant concentration in a given scenario, in μg/m^3.

We used two scenarios: a decrease of the annual means by a fixed amount of 5 μg/m^3 and a decrease of the annual means down to the annual WHO-AQG (20 μg/m^3 for PM_{10} and 10 μg/m^3 for $PM_{2.5}$). For the assessment of long-term impacts of ozone, we only considered a decrease by a fixed amount of 5 μg/m^3 in ozone levels during the April–September season. For the short-term impacts of ozone, the WHO-AQG (100 μg/m^3) was applied to daily values.

For the long-term exposure to $PM_{2.5}$, health impacts were assessed using a standard abridged life table methodology. The baseline life tables were compared to impacted life tables, computed from:

$$_nD_m^{impacted} = {_nD_x}.e^{\beta\Delta x}$$

where $_nD_m$ is the total number of deaths in the age group starting at age n and covering m years.

We applied that function to 5-year age groups starting at age 30, using the same β for all age groups, to compute the average potential gain in life expectancy.

Results were expressed as number of postponed deaths and as gains in life expectancy at 30.

The annual burden of survival, expressed as the total life years which could have been gained was computed as the product of the average life expectancy at age 30 by the estimated number of population at age 30.

All computations were performed by each city using Microsoft Excel® spreadsheets developed by the Aphekom project, available at http://si.easp.es/aphekom/. Detailed equations for the computations of the life expectancy are given in these tools.

All results were centralized and checked using R scripts.

2.6. Economic valuation

2.6.1. Mortality

A value for a decrease in mortality needs to be deduced from stated or revealed economic behaviour. In recent years, there has been a growing interest in stated preference techniques in which people are surveyed and express trade-offs between risk of death and money. A Value of a Statistical Life (VSL) is deduced from their answers and used to value postponed deaths. Empirical assessments have so far provided a range of values generally between €0.7 and €10 million (ASCC, 2008; OECD, 2012; US-EPA, 2012). A key finding from this literature is that the VSL depends on the characteristics of the risk of death: age at death, time between exposure and death (i.e. latency), and nature of the underlying risk have largely been found to be relevant factors (Cropper et al., 2011; Dekker et al., 2011).

For the purpose of this study we chose to rely on the mortality valuation study undertaken for the EC DG Research-funded New-Ext. The Central VSL of €1,655,000 was taken as the average of the Low (median, €1,090,000) and High (mean, €2,220,000) values reported by Friedrich et al. (2004), and applied to each postponed deaths to compute annual benefits (Friedrich et al., 2004). This VSL was chosen because it is representative of the European population, and is within the range of the VSL used in other major European studies (Bickel et al., 2006; Holland et al., 2005). In addition, this VSL is comparable to the one obtained in the only survey to date especially designed for air pollution exposure risk in Europe (€1.61 million (Chanel and Luchini, 2012)).

This VSL is clearly lower than the one generally used in the USA. For instance, when computing the human health effects of the Clean Air Act from 1990 to 2020, US-EPA (2011) used $7.4 million (2006) (US-EPA, 2011). Such a discrepancy is mainly due to the fact that this value is based on the mean of a distribution fitted on 26 VSL estimated 20 to 35 years earlier, of which 21 are wage-risk studies, i.e. revealed preference studies.

2.6.2. Hospitalizations

For hospitalizations, the standard cost of illness approach was used. It consists in applying a unique economic value that combines the direct and indirect costs for each hospitalization. It is especially suitable for the assessment of these costs because it relies on actual health expenditures. Note that the cost of illness cannot account for intangible costs like the assessment of pain, grief and suffering as there are no market prices for these cost factors.

The direct medical costs related to cardiac and respiratory hospitalizations were computed from an average cost per inpatient day and an average length of stay in hospital. These cost data were taken from the Commission of European Communities (Commission of the European Communities, 2008) for the twelve countries included in the study. The average lengths of stay in days were obtained from the OECD Health Database (OECD, 2010) for all countries except Romania, which was imputed from the population-weighted average lengths of the 11 other countries (Table 2).

The indirect medical costs were computed as the average gross loss of production per day times twice the average length of stay in hospital, as described in Ready et al. (2004). The daily loss of production was computed as the average gross earnings in industry and services (full employment). It was obtained from Paternoster (2003) for each country and divided by 365 days.

Consistently with the study period, all monetary values were expressed in Euros 2005.

2.7. Uncertainty analysis

All HIA results are reported with a confidence interval (CI) based on the confidence intervals of the CRF, which represents only a part of the total uncertainty. When the lower estimate of the CRF was below 1, as was the case for the effect of ozone on respiratory hospitalizations, we considered that the minimum health benefits of reducing air pollution would be null, and set the lower limit of the HIA-CI to 0.

Regarding the economic valuation, the Low and High estimates of the VSL proposed above provide a range of possible values for mortality effects. For the morbidity, a +/−33% range around the total hospitalization costs was used (Hurley et al., 2005). Two ways of combining the uncertainties from the CRF and from the economic valuation have then been used.

Table 2
Average length of stay, average cost per day and total hospitalisation cost per patient.

Country	Average length of stay in days in 2005[a]		Average cost per day (€ 2005)		Total costs related to hospitalisation (€ 2005)	
	Circulatory system	Respiratory system	Hosp. all causes[b]	Work loss[c]	Circulatory system	Respiratory system
Austria	8.2	6.6	319	83	3977	3201
Belgium	9.2	8.8	351	98	5032	4814
France	7.1	7.1	366	83	3777	3777
Greece	7.0	5.0	389	48	3395	2425
Hungary	7.4	6.5	59	18	703	618
Ireland	10.5	6.9	349	81	5366	3526
Italy	7.7	8.0	379	62	3873	4024
Romania	8.5[d]	7.4[d]	57	6	587	511
Slovenia	8.6	7.3	240	34	2649	2248
Spain	8.5	7.4	321	55	3664	3189
Sweden	6	5.2	427	92	3666	3177
United Kingdom	11.4	8.0	581	116	9268	6504
Mean[d]	8.5	7.4	373	73	4411	3840

[a] OECD (2010).
[b] Commission of the European Communities (2008)), annex 7, cost/bed/day corr.
[c] Eurostat (2003).
[d] Population-weighted average, 2005 population data from OECD (2010).

First, we applied for each city the Low and High estimates of the economic values associated with each health outcome to the HIA results provided by the epidemiological computations. A range of monetary benefits (labelled Low, Central and High) for the HIA and the related upper and lower 95% CI bounds are presented in the Results section.

On a second step, the analysis was restricted to the long-term mortality effects of PM which represented the largest part of the overall effect. The sensitivity analysis was performed on the total estimates of the HIA across the 25 cities rather than at the city level. The number of postponed deaths and the VSL were treated as random variables with specified distributions of probability. A normal distribution was used to characterize the spread of the mortality data, defined in terms of its mean and standard deviation. This choice was based on the assumptions and data obtained in the HIA. A triangular distribution was used for the VSL, between the Central value, a High value and the Low value. The triangular distribution is typically used when knowledge of the variable is more subjective than objective.

Once these probability distributions defined, Monte Carlo simulations were used to propagate the uncertainty in the HIA results and the economic values. It consisted in randomly and independently drawing a number of postponed deaths from the mortality distribution and a VSL from the VSL distribution (the quasi random number generator of the Humboldt-Universitaet zu Berlin, 2012 was used to guarantee randomness) (Humboldt-Universitated zu Berlin, 2012). Then, their product generated one estimate of the annual long-term mortality benefits. This has been repeated 10,000 times, which is sufficient to accurately characterize the distribution of these monetary benefits (Holland et al., 2005; Ostro et al., 2006).

3. Results

3.1. Characteristics of the centres

The population of cities studied varied from 236,982 inhabitants in Granada to 7,484,900 inhabitants in London (median: 955,702), totalling nearly 39 million inhabitants in the 25 cities, of which 21% (5,849,709 inhabitants) were older than 65 years of age. The standardized mortality rate for all-causes mortality in the population 30 years old varied from 634 per 100,000 in Rome to 1572 per 100,000 in Bucharest (median 975 per 100,000), with a notably larger share of cardiovascular mortality in Budapest and Bucharest (Table 3). The annual number of hospitalizations for cardiac causes varied from 418 per 100,000 in Malaga to 2997 per 100,000 in Bucharest. Numbers were similar for respiratory causes. Hospitalization rates for both groups of causes were notably higher in Bucharest and Vienna (Table 3).

The maximum daily 8-hour ozone mean varied from 50.0 $\mu g/m^3$ in Seville to 82.8 $\mu g/m^3$ in Athens (median: 65.0 $\mu g/m^3$). No city complied fully with the WHO-AQG and the proportion of days with the maximum daily 8-hour ozone mean above 100 $\mu g/m^3$ varied from 0.7% in Dublin to 32.3% in Rome (median: 9.4%) (Table 4). Only Dublin, Malaga and Stockholm complied with the WHO-AQG of 20 $\mu g/m^3$ for PM_{10} and only Stockholm complied with the WHO-AQG of 10 $\mu g/m^3$ for $PM_{2.5}$. The PM_{10} annual mean concentrations varied from 15.0 $\mu g/m^3$ in Dublin to 55.3 $\mu g/m^3$ in Bucharest (median: 27.6 $\mu g/m^3$). $PM_{2.5}$ annual mean concentrations varied from 9.4 $\mu g/m^3$ in Stockholm to 38.2 $\mu g/m^3$ in Bucharest (median: 16.6 $\mu g/m^3$) (Table 4).

3.2. Short and long term effects of exposure to ozone on mortality and hospitalizations

A decrease of 5 $\mu g/m^3$ in the maximum daily 8-hour ozone mean would have resulted in an annual decrease of more than 380 respiratory hospitalizations in the population aged 15 and older (Table 6). By comparison, compliance with the WHO-AQG of 100 $\mu g/m^3$ for the maximum daily 8-hour ozone mean would have resulted in the

Table 3
Population, mean annual mortality, hospital admissions observed in the study areas for the period 2004–2006 (/100,000 inhabitants).

City	Total Population	Total non-external mortality (all ages)	Total mortality (including external) (30 and over)	Cardiovascular mortality (30 and over)	Respiratory mortality (30 years and older)	Cardiac hospitalisations (all ages)	Respiratory hospitalisations (all ages)
Athens[a]	3,412,740	834	851	401			
Barcelona	1,593,075	964	957	313	168	894	1152
Bilbao	706,533	827	848	267	239	926	1198
Bordeaux	642,397	688	725	214	92	872	782
Brussels	1,012,776	870	911	311	29	646	955
Bucharest	1,924,959	1078	1099	636	60	2997	3506
Budapest[a]	1,690,109	1294	1353	681	49		
Dublin[a]	506,211	797	819	286	183		
Granada	236,982	876	892	330	264	623	696
Le Havre	245,461	831	864	218	80	1029	1042
Lille	1,107,861	686	719	192	11	1122	1228
Ljubljana	266,935	966	867	336	205	850	1005
London	7,484,900	673	661	238	2	536	817
Lyon	1,012,715	645	660	181	722	849	745
Malaga	558,287	790	804	301	67	418	473
Marseille	955,702	829	861	252	51	1362	1028
Paris	6,507,783	582	597	149	8	815	873
Rome	2,808,960	732	737	304	84	1402	994
Rouen	446,382	787	827	231	273	1159	970
Seville	704,154	815	828	358	33	624	550
Stockholm	1,257,302	814	879	347	38	1113	698
Strasbourg	440,264	659	676	204	174	741	968
Toulouse	744,284	561	583	173	23	924	1019
Valencia	738,441	755	851	286	33	578	735
Vienna	1,657,559	914	944	442	45	2250	1858
Total	38,662,772	773	789	300	64	882	932

[a] Hospitalisation data were not available over the study period.

avoidance of more than 150 respiratory hospitalizations annually in the population aged 15 years and older (Table 6). Reducing the summer ozone mean concentrations by 5 µg/m^3 would have postponed on the long-range about 280 respiratory deaths per annum (Table 5). Results for each individual city are available in the Supplementary material (eTable 1).

3.3. Short-term impacts of exposure to PM_{10} on hospitalizations

In the 25 cities, compliance with the WHO-AQG of 20 µg/m^3 would have avoided more than 8000 hospitalizations for cardiovascular and respiratory causes annually (Table 6). A decrease by 5 µg/m^3 would and avoided more than 3000 hospitalizations (Table 6). Results for

Table 4
Ozone and PM concentrations observed in the study areas for the period 2004–2006.

City	Average of ozone daily 8 h-maximum values (µg/m^3); entire year	% of days ozone daily 8 h-maximum values > 100 µg/m^3	Average of the ozone daily 8 h-maximum values (µg/m^3) April-September	PM_{10} Annual mean (µg/m^3)	$PM_{2.5}$ Annual mean (µg/m^3)	Number of stations ozone	Number of stations PM_{10}	Number of stations $PM_{2.5}$
Athens	82.8	32.2	105.3	42.0	29.4[a]	8	4	0
Barcelona	59.6	6.7	75.4	37.1	27.0	1	2	1
Bilbao	60.9	2.2	73.2	36.1	15.7	9	6	3
Bordeaux	68.1	14.1	88.3	24.9	15.7	4	4	2
Brussels	58.5	9.4	80.2	25.8	19.0	4	2	1
Bucharest	58.9	7.8	73.6	55.3	38.2	7	6	2
Budapest	65.0	12.1	85.2	48.2	33.7[a]	6	2	0
Dublin	55.7	0.7	59.1	15.0	10.5[a]	2	2	0
Granada	65.4	7.0	81.4	30.6	21.4[a]	1	1	0
Le Havre	66.8	7.4	81.2	22.5	14.5	3	2	2
Lille	61.1	9.6	81.5	27.6	16.6	6	5	1
Ljubljana	76.3	23.9	95.0	33.5	29.4[a]	1	1	0
London	50.8	3.2	63.5	25.0	13.1	13	3	1
Lyon	67.8	18.9	94.6	24.7	16.5	5	1	1
Malaga	70.2	4.4	80.4	18.3	12.8[a]	1	1	0
Marseille	78.4	28.0	103.5	29.9	18.5	4	3	1
Paris	58.7	9.8	82.0	25.0	16.4	12	9	4
Rome	75.1	32.3	102.8	38.5	20.9	2	1	1
Rouen	61.6	7.9	79.4	22.6	15.3	5	2	2
Seville	50.0	1.0	68.3	32.7	22.9[a]	3	1	0
Stockholm	66.1	5.1	76.7	16.0	9.4	1	1	1
Strasbourg	61.7	14.8	88.2	24.8	16.6	6	3	2
Toulouse	77.6	22.9	98.0	21.9	14.2	5	3	2
Valencia	59.0	2.6	76.7	32.8	23.0[a]	1	1	0
Vienna	73.0	17.3	92.2	29.1	21.6	5	9	2

[a] $PM_{2.5}$ computed from PM_{10}.

Table 5
Long-term mortality impacts of decreasing ozone and $PM_{2.5}$ in the 25 European cities (total annual number of postponed deaths) [CI 95%][a], and minimal and maximal values across the cities.

	Mortality	Total impact	Range in cities (min, max)
Long-term impacts of decreasing ozone by 5 μg/m^3	Respiratory mortality	289 [76:472]	Le Havre (1 [0:2]) London (83 [22:135])
Long- term impacts of complying with the WHO AQG of 10 μg /m^3 for $PM_{2.5}$	Total mortality > 30 years old	18,801 [6597:32,434]	Stockholm (0 [0:0]) Bucharest (3211 [1151:5402])
	Cardiovascular mortality > 30 years old	15,015 [10,531:18,095]	Stockholm (0 [0:0]) Bucharest (3353 [2391:5402])
	Gain in life expectancy at 30 (months)	198 [67;358]	Stockholm (0[0:0]) Bucharest 22 [7:40]
Long-term impacts of decreasing $PM_{2.5}$ by 5 μg/m^3	Total mortality > 30 years old	8761 [3006:15,513]	Granada (61 [21:108]) London (1420 [487:2515])
	Cardiovascular mortality > 30 years old	6399 [4385:7841]	Le Havre (30 [20:108]) London (982 [673:2515])

[a] 95% CI: confidence interval based only on the uncertainty in the estimation of the CRF in the reference study.

each individual city are available in the Supplementary material eTable 2 and eTable 3 for these two scenarios.

3.4. Long-term impacts of chronic exposure to $PM_{2.5}$ mortality

In the 25 cities, compliance with the WHO-AQG of 10 μg/m^3 resulted in an increase in life expectancy at age 30 ranging from 0 (Stockholm) to 22 months (Bucharest) (median: 5.8 months) of life expectancy at age 30, depending on the city (Fig. 1). This is equivalent to a burden on survival of nearly 421,000 life years lost per annum, and to a burden on mortality of nearly 19,000 postponed deaths per annum, of which more than 15,000 are caused by cardiovascular diseases (Table 5). In that scenario, on median 302 deaths would have been postponed per city (from 0 in Stockholm to more than 3000 in Bucharest). City-specific results are available in the Supplementary material eTable 2.

A decrease by 5 μg/m^3 in the average $PM_{2.5}$ levels could have led to a gain in life expectancy at 30 of 3 to 5 months (median: 4 months), equivalent to an annual burden on survival of more than 224,000 years and to a total burden on mortality of nearly 9000 deaths annually (nearly 6400 of which are from cardiovascular diseases) (Table 5). In that scenario, on median 192 deaths would have been postponed per city (from 61 in Granada to more than 1400 in Bucharest). Results per cities are available in the Supplementary material eTable 2 and eTable 3 for these two scenarios.

Table 6
Short-term morbidity impacts of decreasing ozone and PM_{10} in 22 European cities (total annual number of postponed deaths) [CI 95%][a], and minimal and maximal values across the cities.

	Hospitalisation	Total impact	Range in cities (min, max)
Short term impacts of complying with the WHO AQG of 100 μg/m^3 for ozone	Respiratory hospitalisations 15–64 years old	27 [0:324]	Seville, Granada, Valencia, Malaga (0 [0:0]) Roma (6 [0:67])
	Respiratory hospitalisations > 64 years old	129 [0:309]	Seville, Granada, Valencia, Malaga (0 [0:0]) Roma (36 [0:86])
Short-term impacts of decreasing ozone by 5 μg/m^3	Respiratory hospitalisations 15–64 years old	63 [0:755]	Granada (0 [0:3]) Paris (10 [0:116])
	Respiratory hospitalisations > 64 years old	314 [0:750]	Granada (1 [0:3]) Paris (42 [0:100])
Short-term impacts of complying with the WHO AQG of 20 μg /m^3 for PM_{10}	Respiratory hospitalisations	5325 [2921:7734]	Malaga, Stockholm (0 [0:0]) Bucharest (2649 [1459:3836])
	Cardiovascular hospitalisations	2648 [1330:3952]	Malaga, Stockholm (0 [0:0]) Bucharest (1207 [607:1798])
Short-term impacts of decreasing PM_{10} by 5 μg/m^3	Respiratory hospitalisations	2035 [1111:2970]	Granada (9 [5:13]) Bucharest (381 [208:557])
	Cardiovascular hospitalisations	1018 [510:1524]	Granada (4 [2:7]) Bucharest (172 [86:258])

[a] 95% CI: confidence interval based only on the uncertainty in the estimation of the CRF in the reference study. Negative values set to 0.

3.5. Economic valuation

Results in Tables 7 and 8 represent a range of monetary benefits (Low, Central and High as previously defined) for each of the health impacts as well as for the related upper and lower 95% CI bounds.

For the 25 European cities, the highest monetary gains were associated with the mortality impacts of complying with the $PM_{2.5}$ WHO-AQG. They would have totalled some €31 billion per year [95% CI €11 billion: €54 billion], the smallest share for Dublin (€20 million, [95% CI €7 million: €36 million]) and the greatest share for Athens (€5 billion, [95% CI €2 billion: €9 billion]). The annual economic benefits associated with a decrease of $PM_{2.5}$ by 5 μg/m^3 would have amounted to €14.5 billion [95% CI €5 billion: €25,7 billion]. The greatest share would be for Bucharest (€1.1 billion) and the smallest share for Le Havre (€0.2 million).

For the 25 cities, the annual economic benefits of a decrease of ozone by 5 μg/m^3 would amount to €520 million [95% CI €140 million: €860 million], mainly associated with the long-term postponed mortality (more than 99%).

The annual economic benefits of a decrease of PM_{10} to the WHO-AQG would amount to €19 million [95% CI €10 million: €28 million] associated to the avoided hospitalizations. The annual economic benefits associated with a decrease of PM_{10} by 5 μg/m^3 would have amounted to €11million [95% CI €6 million: €13 million].

Results of the Monte-Carlo uncertainty analysis are only presented here for the impacts of the long-term mortality effects of $PM_{2.5}$, because of their overwhelming importance in the overall economic benefits. Fig. 2 shows the distribution of the annual mortality benefits of complying with the WHO-AQG of 10 μg/m^3 for $PM_{2.5}$, with a mean gain of €31.1 billion, and an empirical 95% CI of €9.4 billion–€56.3 billion. When reducing $PM_{2.5}$ by 5 μg/m^3, the mean gain would be of €14.5 billion, and the empirical 95% CI of [€4: €26.4] billion (see e-Fig. 1). Although both means are very close to the values obtained in Table 7, the ranges associated with these estimates are slightly wider, because they jointly account for epidemiological and economic uncertainties.

Table 7
Total annual monetary valuations of the long-term mortality impacts of reducing ozone and $PM_{2.5}$ in the 25 cities (Low, Central and High monetary estimates of mean, upper 95% CI and lower 95% CI number of cases. Costs rounded to the nearest € million.).

	Mortality	Associated benefits, Low estimates € million [95% CI]	Associated benefits, Central estimates € million [95% CI]	Associated benefits, High estimates € million [95% CI]
Long-term impacts of decreasing ozone by 5 μg/m^3	Respiratory mortality	315 [83:514]	521 [137:851]	1 157 [304:1890]
Long- term impacts of complying with the WHO AQG of 10 μg /m^3 for $PM_{2.5}$	Total mortality > 30 years old	20 493 [7 190:35 353]	31 116 [10 918:53 678]	41 738 [14 645: 72,003]
	Cardiovascular mortality > 30 years old	16 366 [11 479: 19 724]	24 850 [17 429:29 947]	33 333 [23 379:40 171]
Long-term impacts of decreasing $PM_{2.5}$ by 5 μg/m^3	Total mortality > 30 years old	9 549 [3277:16,909]	14 499 [4 975:25 674]	19 449 [6 673:34 439]
	Cardiovascular mortality > 30 years old	6 975 [4 780:8 547]	10 590 [7 257:12 977]	14 206 [9 735:17 407]

4. Discussion

4.1. Summary of main findings

In the 25 cities, population is still exposed to air pollutant levels higher than those recommended by the WHO to protect public health. The largest health burden was attributable to the impacts of chronic exposure to $PM_{2.5}$. Complying with the WHO guideline of 10 μg/m^3 in annual mean would add up to 22 months of life expectancy at age 30, depending on the city, corresponding to 19,000 postponed deaths each year. The associated monetary gain would total some €31 billion annually, including savings on health expenditures, absenteeism and intangible costs such as well-being, life expectancy and quality of life. Each year, all other things staying constant, the proportion of all-causes mortality over 30 years old attributable to a reduction to 10 μg/m^3 in $PM_{2.5}$ levels would be on average 6.2% of the total burden of mortality in the 25 cities. This represents a significant share of the mortality that can be contrasted with other causes. In Paris, for instance, this corresponds to 1423 deaths per year. This number is in the same order of magnitude than the number of deaths over 30 years old attributable to Acute Myocardial Infarction (I21 in ICD-10) which was 1385 in 2005, and almost ten times larger than those attributable to transportation accidents (V01 in ICD-10) representing 155 deaths in 2005 (Centre d'épidémiologie sur les causes médicales de décès, 2012).

4.2. Uncertainties

In this study we applied standardized guidelines and data quality control to allow comparison of results between cities, and to reduce the errors associated with the HIA computations. We assessed part of the uncertainty associated with the CRF using the 95% CI of the RR when reporting our findings. This 95% CI reflects the uncertainty of the estimate for one particular study, but does not capture uncertainties related to other aspects, such as representativeness of the population, shape of the CRF, omitted confounding variables or exposure misclassification.

A first source of uncertainty lies within the exposure assessment, when we estimated the average exposure for the population in each city by averaging data from fixed monitors. Doing so, we probably underestimated the impact of chronic exposure to PM, since there can be significant within-city variations of exposure, notably in relation to road traffic. Even when concentrations measured at the urban background monitoring sites are below the WHO-AQG, parts of the population may actually be exposed to higher levels of PM. Recent epidemiological findings suggest that within-city variations can be even larger than differences in pollution levels between cities (Jerrett et al., 2005). The use of TEOM measurements in most cities is also likely to underestimate the actual exposure, even when using a local correction factor. In 7 cities (Dublin, Athens, Budapest, Ljubljana, and 4 Spanish cities), $PM_{2.5}$ data were not available and had to be approximated

Table 8
Total annual monetary valuations of the morbidity impacts of reducing ozone and PM_{10} in the 22 cities (Low, Central and High monetary estimates of mean, upper 95% CI and lower 95% CI number of cases. Costs rounded to the nearest € thousand.).

	Hospitalization	Associated benefits, Low estimates € thousands [95% CI]	Associated benefits, Central estimates € thousands [95% CI]	Associated benefits, High estimates € thousands [95% CI]
Short term impacts of complying with the WHO AQG of 100 μg/m^3 for ozone	Respiratory hospitalizations 15-64 years old	61 [0:486]	91 [0:729]	121 [0:972]
	Respiratory hospitalizations > 64 years old	327 [0:464]	491 [0:695]	655 [0:927]
Short-term impacts of decreasing ozone by 5 μg/m^3	Respiratory hospitalizations 15–64 years old	137 [0:1 779]	206 [0:2 669]	275 [0:3 559]
	Respiratory hospitalizations > 64 years old	837 [0: 1 768]	1 256 [0: 2 651]	1 675 [0:3 535]
Short-term impacts of complying with the WHO AQG of 20 μg /m^3 for PM_{10}	Respiratory hospitalizations	7 987 [4 338:11 486]	11 982 [6 573:17 403]	15 976 [8 764:23 203]
	Cardiovascular hospitalizations	4 578 [2 299:6 832]	6 867 [3 449:10 249]	9 156 [4 599:13 665]
Short-term impacts of decreasing PM_{10} by 5 μg/m^3	Respiratory hospitalizations	4 795 [2 618:6 999]	7 194 [3 927:10 499]	9 591 [5 237:13 999]
	Cardiovascular hospitalizations	2 659 [1 332:3 982]	3 990 [1 999:5 973]	5 319 [2 665:7 963]

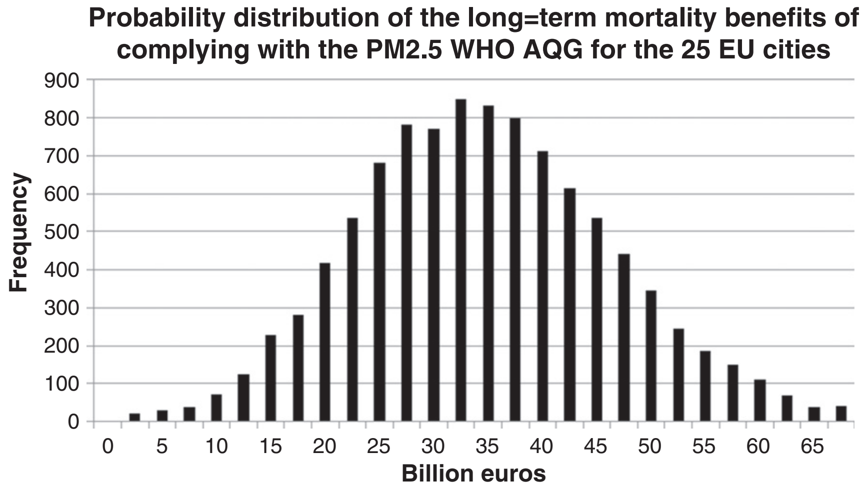

Fig. 2. Probability distribution of the annual long term mortality benefits of complying with the WHO AQG of 10 μg /m^3.

using PM_{10} levels. Earlier work by the Apheis project (Medina et al., 2005) showed that this may cause a small overestimation of the $PM_{2.5}$ levels. An analysis of the $PM_{2.5}$-to-PM_{10} ratio in 34 monitoring sites across Europe found a mean ration of 0.73, and a range from 0.5 to 0.9 (Van Dingenen et al., 2004). In Dublin, concentrations of PM_{10} and $PM_{2.5}$ corresponding to a ratio around 0.6 have been reported (Yin et al., 2005). In Spain, in urban areas, concentrations reported correspond to ratio between 0.5 and 0.8 (Querol et al., 2004). Lower ratio has also been reported in Central Europe (Houthuijs et al., 2001), as in Athens, where a recent work reported $PM_{2.5}$-to-PM_{10} ratio of 0.43 (Lianou et al., 2011). This can result in large differences in the HIA outputs: for Athens, using a ratio of 0.4 instead of 0.7 gives an estimated $PM_{2.5}$ annual mean of 17 μg/m^3 (compared to 29 μg/m^3). It represents 834 deaths postponed, corresponding to 4.8 months of life gained at age 30 (compared to 3100 deaths and 13 months of life). To avoid such uncertainties, routine $PM_{2.5}$ monitoring should be extended to more European cities.

Mortality remains our first choice for health outcomes, as it is robust, easy to obtain from existing records in all the European cities involved in the study, and for all-cause mortality is not subject to misclassification. We also used hospitalization data but they are less robust and more heterogeneous. For example, the high baseline rates reported in Bucharest and Vienna strongly influenced the results of HIA calculations, causing comparability problems between cities. This can be due to differences in the coding practices or due to differences in the use of hospitalisation in the health-care systems.

We used CRF from European multi-country and multi-city studies. For short-term effects of PM_{10}, the chosen CRF were consistent with other recent results from one-country multi-city studies such as the PSAS French study (Larrieu et al., 2007), the EMECAS Spanish study (Ballester et al., 2006) and the EPI-AIR Italian study (Colais et al., 2009; Stafoggia et al., 2009). For long-term effects of $PM_{2.5}$, we used the results from the ACS study (Pope et al., 2002, 2004). Due to its statistical power and its adjustment on major individual risk factors, the ACS CRF remains the best evidence available on the long-term effects of chronic exposure to mortality. It is interesting to note, that a European cohort study (Beelen et al., 2008) obtained CRF consistent with the ACS study. Recently, results of a large Canadian cohort study showed a slightly higher long-term effect of $PM_{2.5}$ on non-accidental mortality (Crouse et al., 2012).

Kinney et al. (2010) showed how eliciting expert judgement on the uncertainty of the CRF can inform HIA interpretation. We used the results of work by the Committee on the medical effects of air pollutants (Ayres, 2009), which elicited the view of seven of its experts on the uncertainty surrounding the CRF for long-term effects of $PM_{2.5}$ on total mortality. The resulting aggregated 95% plausibility interval ranged from 1.00 to 1.15, i. e. approximately from 6% below to 8% above the central estimate (1.06). Translating these figures into the total number of deaths postponed would lead to a range from 0 to 42,229.65, and to a median gain in average life expectancy at age 30 of 0 to 14.1 months in the 25 cities.

We restricted the analysis to those over 30 years old, and therefore ignore the gains for the population under 30 when they would reach that age. This choice leads to an underestimation of the gains that could be associated with an improved air quality. In the computation of the gain in life expectancy, we used a standard abridged life table methodology, using 5-years age groups, without differences by sex. We also did not take into account a lag between the decrease of the exposure and the health outcomes. Indeed, our objectives were to assess the burden of air pollution all in a scenario where all other variables stayed constant over time. To perform a more realistic cost-benefits analysis, one would need to consider measures implemented to achieve the reduction of air pollution, and the time required to observe this reduction. Tools to compute the gain in life expectancy with more details for such purposes have been developed by Miller and Hurley (2006).

Finally, we performed the HIA only for ozone and PM. Health effects have been associated with NO_2 but it is generally agreed that at the levels observed in an urban setting, which are lower than levels used in controlled human studies, the results of the numerous epidemiologic studies showing a link between NO_2 levels and various health effects, notably respiratory effects, could be due in part to other traffic-related pollutants, e.g. ultrafine particles, for which NO_2 is a proxy (HEI panel on the health effects of traffic-related air pollution, 2010; Searl, 2004). Performing HIAs on both NO_2 and PM could have led us to do double counting, so we chose not to perform HIA on NO_2, even if we probably underestimate the health impacts of urban air pollution, particularly in the short term. As NO_2 levels are currently increasing in Europe, due to an increasing trend in traffic-related primary NO_2 emissions (Carslaw et al., 2007), the use of NO_2 in HIAs should be examined further in future work.

HIA of traffic were performed in a subset of European cities, using proximity to traffic rather than pollutant concentrations to characterize exposure. This work focused on the impacts of pollution chronic diseases and exacerbations using the concepts framed by Kunzli et al. (2008), et is described elsewhere (paper in press).

Considering the above uncertainties, we assume that our results are *at least* estimates of the real impact of air pollution.

In the economic valuation, we used the same values for the VSL in all the cities. Indeed, accounting for differences in countries' Gross Domestic Product (GDP) per capita would seem ethically unacceptable: it would, for instance, lead to a sevenfold lower VSL in Romania than in Ireland. We used low and high estimates of the VSL as well as hospitalization costs to provide a range of possible monetary benefits for the HIA results and for the related upper and lower 95% CI bounds. We also performed an uncertainty analysis that simultaneously accounts for uncertainties concerning epidemiology and economic valuation through an integrated approach.

The economic valuation associated to the gains in life expectancy was not presented in that paper, but is available from the authors. Using gains in life expectancy instead of number of deaths, resulted in a estimated 400,000 life years lost and about 30 billion Euros.

4.3. Policy relevance and ways forward

Our results show that air pollution still has a major public health impact in European cities and that life expectancy and monetary benefits increase significantly when levels of fine particles and ozone are reduced further in Europe. Our work probably underestimates the total impact of air pollution on European's health, and confirms the previous statement (Medina et al., 2009) that setting more ambitious objectives for PM_{10} and $PM_{2.5}$ would significantly reduce mortality and morbidity in Europe, thus improving the health status and quality of life of the population. The clean air for Europe (CAFE) project estimated that ozone was responsible for about 21,000 respiratory admissions in Europe (25 countries) in 2000, and PM for 348,000 premature deaths, and 100,000 hospitalizations for respiratory and cardiovascular causes. The associated economic valuation ranges between 276 and 790 billion Euros. This translates to an estimated average benefit of €191 and €397 per person per year (Hurley et al., 2005; Watkiss et al., 2005). In a previous study in Austria, France, and Switzerland, Kunzli et al. (2000) reported more than 40,000 premature deaths per year. Comparison between our results and these studies is difficult due to the differences in study area, population, study period… However, they all show the major burden of air pollution on public health in Europe.

Specifically in our 25 cities, compliance with the WHO-AQG would result in larger overall benefits, concentrated on the cities with the highest pollution levels, while a decrease by 5 $\mu g/m^3$ would result in smaller benefits, but visible in all participating cities. For the 25 cities studied, the potential public health gains associated with postponed mortality largely exceed the gains associated with morbidity. For PM_{10}, postponed mortality represents about 85% of the overall monetary benefits. Chronic effects largely exceed acute effects for mortality, as the long-term monetary benefits associated with a reduction of $PM_{2.5}$ are more than 200 times greater than the monetary benefits associated with a reduction of PM_{10}, whatever the scenario. This result certainly deserves the attention of public decision-makers. Linking the reduction of air pollution with, for instance, a reduction of traffic-related noise or a reduction of greenhouse gases emissions would result in potentially larger benefits, providing additional arguments to promote more ambitious air quality targets in Europe.

Beside the need for more stringent regulations, our results also underline the need to improve the monitoring of PM air pollution in Europe. Similarly, more knowledge on the chronic effects of air pollution in Europe is needed. Foreseen results from cohort studies in Europe should be an important step in that process.

Competing interest

None.

Acknowledgements

The huge amount of work behind the Aphekom project is the fruit of the generous and constructive input from all the members of the Aphekom network. We wish to give our special thanks and appreciation to all of them. We also thank M. Stempfelet for her help in producing Fig. 1.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2013.01.077.

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