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Economic cost of the health impact of air pollution in Europe

Clean air, health and wealth

Abstract

This paper extends the analyses of the most recent WHO, European Union and Organisation for Economic Co-operation and Development research on the cost of ambient and household air pollution to cover all 53 Member States of the WHO European Region. It describes and discusses the topic of air pollution from a Health in All Policies perspective, reflecting the best available evidence from a health, economics and policy angle and identifies future research areas and policy options.

Keywords
AIR POLLUTION
COST OF ILLNESS
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List of abbreviations

AAP	ambient air pollution
AOP	ambient ozone pollution
APMP	ambient particulate matter pollution
AQG	air quality guidelines
B/C	benefit–cost (ratio)
BOD	burden of disease
CAAA	Clean Air Act Amendments
CAFE	Clean Air for Europe
CAPP	Clean Air Policy Package
CBA	cost–benefit analysis
CLE	current legislation (scenario)
DALY	disability-adjusted life-years
DRG	diagnosis-related group
EEA	European Environment Agency
EU	European Union
GBD	global burden of disease
GDP	gross domestic product
HAP	household air pollution
IARC	International Agency for Research on Cancer
IHME	Institute for Health Metrics and Evaluation
LYL	life-years lost
MIT	Massachusetts Institute of Technology
MTFR	maximum technically feasible reduction
NO_x	nitrogen oxides
OECD	Organisation for Economic Co-operation and Development
PM	particulate matter
PPP	purchasing power parity
QALY	quality-adjusted life-years
US$	United States dollar
EPA	(United States) Environmental Protection Agency
VOLY	value of a life-year
VSL	value of a statistical life
VSLY	value of a statistical life-year
WHO	World Health Organization
WTP	willingness to pay
YLD	years of life lost to disability
YLL	years of life lost

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Executive summary

This study reports on the economic cost of the public health impacts of ambient and household air pollution, with particular reference to the countries of the WHO European Region.

Current estimates of the joint effects of ambient and household air pollution include an estimated 7 million premature deaths globally each year, representing one in eight of the total deaths worldwide.

In the WHO European Region as a whole, the estimated mortality in 2010 was approximately 600 000 premature deaths, which represents a marked decrease from 2005 for the Region overall. Only half-a-dozen countries out of the 53 WHO European Region Member States registered a slight increase in deaths. The evidence from epidemiology underpinning these estimates is well established, while the evidence from economics shows that ambient and household air pollution also imposes an economic cost to society of several trillion dollars per year, globally.

Present-day economics uses a standard method for assessing the cost of mortality at the level of society: the "value of statistical life" (VSL), as derived from aggregating individuals' willingness to pay to secure a marginal reduction in the risk of premature death. This permits researchers and policy-makers to assess the comparative magnitude of the value that societies attach to a given health impact, and of proposals to mitigate it, using money as a common metric. The economic cost of a mortality impact is given by the VSL value, multiplied by the number of premature deaths. The economic benefit of a mitigating action becomes the same VSL value, multiplied by the number of prevented premature deaths.

Owing to a multi-year research effort spearheaded by the Organisation for Economic Co-operation and Development (OECD), a set of values for average adult VSL in 2005 is now available, along with a method to compute country-specific VSL values for countries both within and outside the OECD and for years beyond 2005.

In contrast, a standard and commonly agreed method by which to measure the cost of morbidity is not yet available. Research is currently being progressed toward establishing an agreed method and agreed values but at present this research can only proceed with indicative estimates. Recent practice and available evidence provide a rationale for using an additional 10% of the overall cost of mortality as a best estimate for the additional cost of morbidity.

On the basis of this method, and these approaches and assumptions, it is possible to estimate the economic cost of air pollution health impact in the countries of the WHO European Region. As of 2010:

- the annual economic cost of premature deaths from air pollution across the countries of the WHO European Region stood at US$ 1.431 trillion; and
- the overall annual economic cost of health impacts and mortality from air pollution, including estimates for morbidity costs, stood at US$ 1.575 trillion.

These results are relatively robust, in that the most common variations on this method and these assumptions do not alter the overarching conclusion: the health impact of air pollution is substantial, and given that good health and a long life are obviously highly valued by society, economic analyses show that the economic cost of air pollution – and hence the benefits of cleaner air – are very large.

In comparing these huge estimated societal costs to country-specific gross domestic product figures, the significance and magnitude of these costs become even more evident: in 44 WHO European Member States the societal costs are equivalent to more than 1% of the respective gross domestic product and in only four of the 48 Member States considered in the analysis do these societal costs amount to less than 1%.

Available evidence on air pollution emission sources suggests that, across the WHO European Region as a whole, several sectors should be targeted for abatement policies. Motorized road transport, household fuel combustion together with agriculture and industrial coal burning sources are of special concern, in terms of their contribution to the health impact of ambient and household air pollution, and the consequent societal costs.

A relatively successful, if imperfect, regulatory regime on air quality in Europe has resulted in substantial progress, especially in European Union Member States, in terms of health impacts and costs, even in the absence of a pricing system capable of taking full account of externalities.

However, in view of the persistence of the problem of air pollution in Europe, filling existing knowledge gaps and correcting distortions in taxes and subsidies – for example, the preference of diesel over petrol – remains highly desirable.

To pursue this goal, operating in the anticipated period of time until a full correction of prices can be achieved, there is a case for conceiving the chronological framework of correction following the approach: regulation + investment + pricing, based on:

- strengthening existing regulation and compliance;
- using available evidence on external costs in relevant investment decisions; and
- closing the information gaps required to prepare a model of fully corrected prices.

The framework presented above – and discussed in further detail in the present report – can be used to provide practical guidance on where and how to strengthen the policy response to the problem of air pollution's health impacts.

Introduction

This document addresses the economic cost of the public health impacts of air pollution, with particular reference to the countries of the WHO European Region.

It presents a summary of the relevant epidemiological evidence on air pollution's health impacts, including in particular recent relevant work released by WHO (WHO, 2014a; WHO, 2014b) and the preceding Global Burden of Disease (GBD) Study, GBD-2010 (Lim et al., 2012; IHME, 2013b; IHME, 2014). It describes a methodology for calculating the economic cost of these health impacts, developed and applied in recent work by the Organisation for Economic Co-operation and Development (OECD) (OECD, 2012, 2014), and presents a new estimate of the economic costs for each of the countries of the WHO European Region. Finally, taking into account available information and information gaps relating to the various sectoral sources of air pollution, the report discusses some of the key implications for policy.

The deleterious impact of air pollution on public health has long been assessed; mortality and morbidity outcomes have been extensively described. While the issue includes the complete set of health impacts, this study deliberately focuses exclusively on the economic cost of the health impacts of air pollution, considering health outcomes that allow economic assessment.

Health impacts of air pollution carry many significant financial and economic implications, not only in terms of the societal cost of mortality and morbidity, which is the key issue of interest for this report, but also household, hospital and public budgets and, therefore, decision-making within and outside of the health sector. These impacts also carry implications for social equity both within and between countries.

Moreover, the deleterious impact of air pollution is not confined to human health. There are many other impacts that are worthy of consideration: those on the built environment, on animal and plant health (with further consequential impacts on the productivity of agricultural and forestry resources), and on larger ecological systems. In this perspective, addressing air pollution can have significant co-benefits for other policy objectives and air quality may simultaneously benefit from interventions that pursue other priorities, such as climate change. Nonetheless, the subject of this study – the economic cost (only) of the public health impacts (only) of air pollution (only) – merits attention in its own right.

The past few years have witnessed substantial accumulation of new evidence on the health effects of air pollution, on the economic cost of these impacts, and thus on the costs and benefits of policy initiatives designed to combat air pollution. As a result, it is now possible to state – and important to communicate – that, relative to many other known environmental health risk factors, the health impacts of air pollution are larger than previously assumed. Moreover, this physical toll imposes a greater economic cost than previously assumed and, consequently, the net economic benefit to be gained by reducing this cost is far greater than previously assumed.

Reducing air pollution and the toll it imposes is not primarily a matter for health policy or for the health sector alone. Rather, it is a policy matter for all the many sectors in which air pollution is generated, and, thereby, a matter requiring a whole-of-government policy approach, as underlined by WHO's Health 2020 policy (WHO Regional Office for Europe, 2015). It is therefore desirable to address this problem in terms that can engage decision-makers across the whole of government, and the use of economic evidence provides a well-established common ground, to this end.

1. The evidence from epidemiology

1.1 GBD owing to air pollution

Air pollution is the largest contributor to the burden of disease (BOD) from the environment. WHO estimated that air pollution in 2012 was responsible for 7 million premature deaths, including almost 600 000 in the WHO European Region. This is equivalent to one in eight of the total number of deaths worldwide. This finding more than doubles previous estimates (WHO, 2014a).

Air pollution is a risk factor for several causes of death, but cardiovascular and cerebrovascular causes of death account for the greater share of attributable mortality: 80% in the case of ambient air pollution (AAP) and 60% in the case of household air pollution (HAP) (WHO, 2014b).

AAP is a broader term used to describe air pollution in outdoor environments. The pollutants that are most harmful to health – closely associated with excessive premature mortality – are fine particulate matter (PM) $PM_{2.5}$ particles that penetrate deep into lung passageways. PM is a mixture, with both physical and chemical characteristics, varying by location and time of year, with seasonal trends. The relative contribution of local, national and transboundary air pollution emissions to the air pollution mixture where people live also varies according to the geography of the area and the presence of other sources of pollution. $PM_{2.5}$ is often used as a general indicator of the air pollution mixture. Other pollutants, such as ground-level ozone, also contribute to the BOD from air pollution. However, by far the biggest quantifiable share of the BOD from air pollution comes from exposure to PM, and its long-term health effects in particular. For this reason, this report focuses on PM.

Premature deaths translate into substantial years of life lost (YLL). In addition, air pollution is responsible for a range of diseases, contributing to the BOD, but the years of life lost to disability (YLD) are difficult to quantify and as such can only represent a relatively small fraction of the estimated total BOD, expressed in disability-adjusted life-years (DALYs).

A publication by WHO[1] in 2014 gives a global estimate for 2012 of 3.7 million premature deaths from AAP and 4.3 million premature deaths from HAP, cumulating in 7 million premature deaths from the joint effects of AAP and HAP (WHO, 2014b). The whole is less than the sum of its parts because the effects of AAP and HAP are not fully independent of each other. This subtraction procedure is not applied in many cases, however – for example, in high-income countries where HAP effects are assumed to be minimal – but it can be difficult to estimate where it is relevant. Hence, WHO advises that the estimate for joint effects should be interpreted with caution (WHO, 2014b).

The GBD-2010 Study, which formed the evidence base for the recent OECD study entitled *The cost of air pollution: health impacts of road transport* (OECD, 2014), estimates the 2010 death toll from each of the three types of air pollution – (1) ambient particulate matter pollution (APMP), (2) ambient ozone pollution (AOP) and (3) HAP from solid fuels – to be

1 WHO estimates for 2012 are only available at regional and global levels, not at country level. Therefore, although WHO results are used here (WHO, 2014b), the detailed quantification in this report is based on the GBD-2010 Study.

3.22 million, 0.15 million and 3.48 million, respectively (Lim et al., 2012).

The GBD-2010 Study does not attempt to estimate a composite figure for the joint effects of these. However, whether one adds together all three types of air pollution at 6.85 million premature deaths or only the two main types at 6.7 million premature deaths (and whether one subtracts to arrive at a composite figure for joint effects), the result exceeds by a clear margin each of the other identified risk factors in the global death toll, with the exception only of high blood pressure at 9.4 million, and tobacco smoking at 5.7 million.

The reported BOD from air pollution has increased over time. This is mainly the result of recent advances in knowledge – mainly from critical breakthroughs in exposure assessment and epidemiological method – and is not indicative of large increases in the actual death toll from year to year. On the contrary, the implementation of clean air regulations and other mitigation measures has succeeded in limiting the actual change in the global premature death toll from air pollution to a relatively modest increase. Moreover, as is detailed below, there has been a modest decrease rather than an increase in mortality from air pollution within the WHO European Region. Therefore a two-fold message needs to be communicated: improved knowledge has led to larger estimates of the BOD from air pollution; however, improved practices have helped in reducing emissions of air pollutants and reduce overall population exposure, especially in Europe.

1.2 The evolving evidence

As is reported in WHO (2014b), the estimate of the global mortality from HAP (for the year 2012) is more than twice as much as its previous reported estimate (for 2004): a leap from 2 million to 4.3 million. In the case of AAP, the latest estimate is almost three times as much previously reported (for 2008): a leap from 1.3 million to 3.7 million (in 2010). The increase in the latter (AAP) is the most dramatic and it is illuminating to track this increase over successive studies. Table 1.1 reports the estimates in four successive studies.

Table 1.1. The reported change in estimates of premature deaths from AAP in successive studies, 2000–2012 (selected years)

Study	Year 2000	Year 2008	Year 2010	Year 2012
	Estimated number of premature deaths			
WHO-GBD 2000[a b]	≈ 0.8 million			
WHO-BOD (2008)[c]		≈ 1.3 million		
GBD-2010 Study[d e]			≈ 3.4 million	
WHO-BOD 2012[f]				≈ 3.7 million

Sources: data reported in or extracted from:
[a] Cohen et al., 2004 (p.1414);
[b] Cohen et al. 2005 (p.1302);
[c] WHO, 2011;
[d] Lim et al., 2012 (p.2238);
[e] IHME, 2013a;
[f] WHO, 2014b (p.1).

The GBD-2010 Study was based on evidence incorporating the results of several critical breakthroughs in the technology and methods of epidemiology, as well as continuing advances in toxicology and the clinical knowledge of diseases. Of these, the following developments deserve particular mention.

- Advanced monitoring methods have been employed, including remote-sensing satellite technology, to estimate emissions and ambient concentrations of pollutants (see, inter alia, Brauer et al., 2012; Evans et al., 2013, and Amann, Klimont & Wagner, 2013).
- There is a much-improved understanding of the relation between emissions/ concentrations of pollutants and the exposure of populations to such chemicals, and of the relation between population exposure and the health impacts of it – resulting in the use of new integrated exposure-response functions (currently undergoing continuing refinement) (WHO Regional Office for Europe, 2013a; 2013b).
- A new understanding has arisen of the link between air pollution and lung cancer (see Beelen et al., 2008; Silverman et al., 2012; Fajersztajn et al., 2013; Raaschau-Nielsen et al., 2013), paving the way for the classification by the International Agency for Research on Cancer (IARC) of outdoor air pollution as a human carcinogen (IARC, 2012, 2013; Benbrahim-Tallaa et al., 2012).
- A fuller understanding has emerged of the cardiovascular, cerebrovascular and respiratory health impacts of air pollution (see, inter alia, Shah et al., 2013; Wellenius et al., 2012; and Laumbach & Kipen, 2012).
- A more comprehensive and more consistent methodology is being used to assemble and analyse the epidemiological evidence base, in order to establish the relative risk of each relevant risk factor in terms of mortality and morbidity, for each relevant disease (see Lim et al., 2012), resulting in each risk factor being more accurately assigned to its relative share in the given number of premature deaths and DALYs in any given year (see Lim et al., 2012 and IHME, 2013b).

There are at least three areas in which future studies are likely to generate new results. The first is through the use of better and more complete data for existing risk–outcome pairings, especially in low- and middle-income countries (see WHO Regional Office for Europe, 2014). This need not necessarily entail any change to established exposure-response functions and may involve a more complete gathering of hospital records. The second is through the selection of the air pollutants to be used for estimating health impacts. Whereas the effects of AAP are now measured through the effect of $PM_{2.5}$, it is increasingly accepted that other pollutants are of relevance (see EEA, 2013a; EC, 2013). In particular, there is now an increasing focus on the independent impact of exposure to nitrogen dioxide (NO_2) (see WHO Regional Office for Europe, 2013a; Holland, 2014). The third area in which new results are likely to be generated is through the expansion of the list of diseases against which the relevant risk is paired. For now, including in GBD-2010 Study, the calculation of the BOD of air pollution has been restricted to four main disease groupings: cancers, and cardiovascular, cerebrovascular and respiratory diseases. However, there is evidence to suggest that air pollution may also play a part in a range of other diseases, including neonatal and neuropsychological impairments (see, for example, Guxens & Sunyer, 2012).

Therefore, it cannot be ruled out that continuing improvements in knowledge will result in more evidence on the deleterious health impacts of air pollution being uncovered and presented, further increasing the magnitude of the estimated BOD.

1.3 The improved practice

In contrast to the dramatic revision of the numbers *reported* before and after the GBD-2010 Study – from 1.3 million premature deaths reported for 2008, to 3.7 million reported for 2012 – the *actual change* in estimated premature deaths over time is relatively modest.

As shown in Table 1.2, in the period from 2005 to 2010 the estimated global mortality from AAP – defined here as the sum of APMP and AOP – increased by an absolute figure of approximately 135 000; that is, by about 4%.

Inclusion of AOP values, which is necessary given the available data, limits comparability with other estimates based on PM alone. However, since AOP makes up less than 5% of the 3.376 million reported premature deaths, it can be concluded that there has been a modest increase only with reference to global mortality from AAP since 2005.

As reported by the OECD (OECD, 2014) and as shown in Table 1.2, mortality decreased in the 34 countries of the OECD by about 20 000 premature deaths (≈ 4%), although this was offset by an increase in premature deaths in China, India and the rest of the world. In the same period, the 53 Member States of the WHO European Region, taken together, also recorded a reduction in premature deaths of about 68 000 (≈ 12%) – that is to say, a greater reduction and rate of reduction than that recorded for the 34 OECD countries, taken together.

Table 1.2. Change in estimated premature deaths from AAP, 2005–2010

Deaths from ambient PM + AOP	2005	2010	Change from 2005 to 2010 (%)
Global total	3 240 129	3 375 977	+4.2
OECD-34	497 958	478 104	–4.0
WHO European Region	577 221	509 100	–11.8

Source: IHME (2014).

These data are consistent with the evidence presented by the OECD (OECD, 2014), which shows that most of the OECD's European member countries achieved a reduction in premature deaths over this period – to a greater or lesser extent – while most of its non-European member countries did not: the United States of America achieved a reduction, but the remaining non-European countries – Canada, Mexico and Chile in the Americas, Japan and the Republic of Korea in Asia, as well as Australia and New Zealand – suffered an increase in premature deaths from 2005 to 2010 (see OECD, 2014). The evidence presented here shows that the results from the non-OECD part of the WHO European Region do not alter the basic pattern found for OECD member countries in Europe.

It is important to note that the improvement observed in the WHO European Region follows in the wake of regulatory intervention across the relevant sectors within the European Union (EU), which also influences the rest of the Region. For the EU, the European Environment

Agency (EEA) recorded an overall improvement in the trend of pollutant emissions for the period from 2002 to 2011 (EEA, 2013a), with reductions in emissions of primary PM (14% for $PM_{10,}$ and 16% for $PM_{2.5}$) and in emissions of its main precursors, including by 27% for nitrogen oxides (NO_x). NO_x emissions are also a precursor for ozone and the 27% reduction in NO_x emissions was matched by similar reductions in other ozone precursor gas emissions. Moreover, and mainly owing to progressively tighter emission limits for Euro 4 vehicles in 2005 and Euro 5 vehicles in 2009,[2] the reduction in emissions achieved in the critical transport sector – by 24% for PM_{10}, by 27% for $PM_{2.5}$ and by 31% for NO_x – exceeded the reduction in emissions overall for the period in question.[3]

Notwithstanding the improvements, the problem that the above-mentioned EU regulatory intervention was designed to address remains very serious. In particular, owing to the improvements in monitoring and modelling technology, it is now clear that the relatively rapid decline in pollutant emissions at source has been followed by a slower decline in ambient concentrations of pollutants and human exposure (see, for example, EEA, 2013a and OECD, 2014). The EEA (2013b) reported that in 2011, 33% of the urban population was exposed to PM_{10} levels above the EU limit, and 88% to PM_{10} levels above the tighter WHO air quality guidelines (AQG) limit. The OECD also argued (OECD, 2014) that the problem has been compounded by increasing market penetration of diesel (see also Carslaw et al., 2011; EEA, 2012; Moore & Newey, 2012; Carslaw & Rhys-Tyler, 2013). In contrast to petrol vehicles, diesel vehicles are reported to have not shown significant reduction in NO_x emissions since the mid-1990s (Carslaw & Rhys-Tyler, 2013). However, partly as a consequence of policy and tax regulations designed to combat climate change, the recent past has witnessed a continuing shift from petrol to diesel vehicles (EEA, 2012).

At any rate, the yearly premature death toll from AAP of more than half a million people in the WHO European Region is a remarkably high number, setting the context in which any improving trend should be evaluated.

1.4 APMP and HAP in the WHO European Region

The evidence presented in Table 1.2 is relevant to the global and regional levels for ambient PM. The evidence presented in Table 1.3 is by country and reports mortality due to ambient PM and HAP from solid fuels, comparing the years 2005 and 2010.

Data for PM by country are available for all Member States of the European Region (with the exception only of Monaco and San Marino). Data by country for HAP from solid fuels are not available for 24 high-income countries. Where such data are available, predominantly in the low- and middle-incomes countries of the Region, they reveal a sufficiently serious problem to merit discussion in this report.

Table 1.3 shows the sum of mortality data for ambient PM and HAP from solid fuels, for the years 2005 and 2010. This is a simple sum, rather than an estimate of joint effects; it should therefore be interpreted with caution.

2 These European emission standards define the acceptable limits for exhaust emissions of new vehicles sold in EU Member States. They are defined in a series of EU directives staging the progressive introduction of increasingly stringent standards (Euro 4 was introduced in January 2005, followed in September 2009 by Euro 5) and concentrated on cleaning up emissions from petrol and diesel cars, especially reducing PM and oxides of nitrogen (NO_x).

3 As noted above, a roughly similar pattern is observable in the United States: see, inter alia, the evidence presented in Amann, Klimont & Wagner (2013), American Lung Association (2013), and United States EPA (EPA, 2013).

Table 1.3. Premature deaths from air pollution (APMP, HAP, and APMP + HAP) per country in the WHO European Region, 2005 and 2010

	APMP		HAP		APMP + HAP	
	2005	2010	2005	2010	2005	2010
Albania	1 643	1 512	2 740	2 620	4 382	4 132
Andorra	29	31	–	–	29	31
Armenia	2 590	2 607	2 914	1 847	5 504	4 454
Austria	3 642	3 122	–	–	3 642	3 122
Azerbaijan	5 146	5 131	3 834	1 819	8 980	6 950
Belarus	8 400	8 236	3 257	1 659	11 657	9 895
Belgium	6 169	5 663	–	–	6 169	5 663
Bosnia and Herzegovina	2 171	2 016	4 824	4 775	6 995	6 791
Bulgaria	11 269	9 492	10 106	8 652	21 375	18 145
Croatia	3 692	3 057	1 930	1 316	5 622	4 373
Cyprus	323	299	–	–	323	299
Czech Republic	8 731	7 028	1 306	575	10 037	7 603
Denmark	1 833	1 818	–	–	1 833	1 818
Estonia	189	351	815	537	1 004	888
Finland	386	450	–	–	386	450
France	17 916	16 892	–	–	17 916	16 892
Georgia	2 971	3 282	7 130	7 547	10 101	10 829
Germany	50 051	41 582	–	–	50 051	41 582
Greece	8 797	8 068	–	–	8 797	8 068
Hungary	11 497	9 189	10 114	8 453	21 612	17 642
Iceland	18	22	–	–	18	22
Ireland	482	671	–	–	482	671
Israel	2 552	2 452	–	–	2 552	2 452
Italy	34 511	32 447	–	–	34 511	32 447
Kazakhstan	11 461	10 064	9 361	5 763	20 822	15 827
Kyrgyzstan	3 380	2 858	5 152	4 491	8 532	7 349
Latvia	398	1 145	1 163	655	1 561	1 801
Lithuania	1 405	1 771	1 657	1 036	3 063	2 806
Luxembourg	179	145	–	–	179	145
Malta	231	228	–	–	231	228
Montenegro	480	391	512	439	992	830
Netherlands	7 828	6 553	–	–	7 828	6 553
Norway	353	186	–	–	353	186
Poland	29 301	24 729	27 004	23 816	56 304	48 544
Portugal	3 453	3 683	–	–	3 453	3 683
Republic of Moldova	4 306	3 225	2 641	1 877	6 947	5 103
Romania	26 214	21 674	19 266	15 558	45 480	37 233
Russian Federation	98 035	94 558	37 796	24 894	135 831	119 452
Serbia	9 310	7 081	11 731	9 368	21 041	16 449

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Table 1.3. (continued)

	APMP		HAP		APMP + HAP	
	2005	2010	2005	2010	2005	2010
Slovakia	4 512	3 777	810	389	5 322	4 166
Slovenia	1 011	876	387	237	1 398	1 113
Spain	15 123	14 042	–	–	15 123	14 042
Sweden	1 003	1 040	–	–	1 003	1 040
Switzerland	2 978	2 656	–	–	2 978	2 656
Tajikistan	2 763	2 760	5 160	4 441	7 923	7 200
The former Yugoslav Republic of Macedonia	1 822	1 662	2 212	2 112	4 033	3 774
Turkey	27 175	28 126	9 498	6 647	36 674	34 772
Turkmenistan	4 918	4 930	363	166	5 282	5 096
Ukraine	76 443	52 868	25 522	13 592	101 965	66 460
United Kingdom	27 546	23 373	–	–	27 546	23 373
Uzbekistan	18 637	18 722	11 368	8 951	30 005	27 672
Total (of available data)	**565 271**	**498 538**	**220 575**	**164 231**	**785 846**	**662 769**

Note. Monaco and San Marino are excluded owing to lack of data.
Source: data extracted from IHME (2014).

As already noted, the WHO European Region, taken as a whole, achieved about a 12% reduction in premature deaths from ambient PM from 2005 to 2010. The 29 countries for which data on HAP from solid fuels are available achieved an overall reduction of about 25%. Nonetheless, the premature death toll (sum) as recorded in 2010 remains remarkably high (at about 663 000 in a single year).

In relative terms, HAP from solid fuels constitutes both a lesser and a declining share of the overall mortality from air pollution in the WHO European Region. In 2010, this share was about 25%, while globally, that share exceeded 50% (WHO, 2014b).

In addition to the number of premature deaths, the impact of air pollution on health can be captured by several other indicators, of which the most commonly used are:

- years of life lost (YLLs), sometimes called life-years lost (LYLs) – namely, the number of years by which a life is shortened by a premature death;
- YLDs – a measure of the relative impact of different diseases on the population;
- DALYs lost – the sum of YLLs and YLDs, often referred to as BOD.

Table 1.4 presents country-specific estimates of DALYs lost as a result of air pollution in 2005 and 2010.

Table 1.4. DALYs lost as a result of air pollution (APMP, HAP, and APMP + HAP) per country in the WHO European Region, 2005 and 2010

	APMP		HAP		APMP + HAP	
	2005	2010	2005	2010	2005	2010
Albania	34 136	29 858	63 024	56 095	97 161	85 953
Andorra	453	469	–	–	453	469
Armenia	51 671	49 141	62 803	37 897	114 474	87 039
Austria	55 032	45 883	–	–	55 032	45 883
Azerbaijan	142 085	130 019	139 650	60 198	281 735	190 217
Belarus	169 257	157 970	69 237	33 923	238 494	191 893
Belgium	101 014	89 698	–	–	101 014	89 698
Bosnia and Herzegovina	41 438	36 245	92 684	86 147	134 122	122 392
Bulgaria	205 548	164 432	193 302	156 958	398 850	321 390
Croatia	62 703	49 122	34 670	22 403	97 373	71 525
Cyprus	5 962	5 513	–	–	5 962	5 513
Czech Republic	145 150	112 463	23 631	10 045	168 781	122 508
Denmark	28 483	27 876	–	–	28 483	27 876
Estonia	3 263	5 492	14 634	8 909	17 897	14 401
Finland	6 600	7 326	–	–	6 600	7 326
France	290 973	266 018	–	–	290 973	266 018
Georgia	62 113	66 084	156 091	156 878	218 204	222 962
Germany	774 268	632 545	–	–	774 268	632 545
Greece	130 321	117 569	–	–	130 321	117 569
Hungary	209 322	159 555	194 088	154 739	403 410	314 294
Iceland	286	325	–	–	286	325
Ireland	8 347	11 451	–	–	8 347	11 451
Israel	42 109	39 563	–	–	42 109	39 563
Italy	482 927	436 848	–	–	482 927	436 848
Kazakhstan	281 429	244 457	256 429	159 122	537 858	403 579
Kyrgyzstan	87 449	74 414	162 712	146 609	250 161	221 023
Latvia	7 398	19 339	21 910	11 579	29 308	30 918
Lithuania	25 394	29 974	31 403	18 716	56 796	48 689
Luxembourg	3 058	2 389	–	–	3 058	2 389
Malta	3 817	3 606	–	–	3 817	3 606

Table 1.4. (continued)

	APMP		HAP		APMP + HAP	
	2005	2010	2005	2010	2005	2010
Montenegro	9 433	7 632	10 529	8 909	19 962	16 540
Netherlands	133 936	108 603	–	–	133 936	108 603
Norway	5 336	2 769	–	–	5 336	2 769
Poland	544 312	439 664	525 297	443 009	1 069 609	882 673
Portugal	54 532	54 689	–	–	54 532	54 689
Republic of Moldova	83 501	62 037	55 540	39 001	139 040	101 038
Romania	489 817	386 302	395 584	301 295	885 401	687 597
Russian Federation	2 180 080	1 935 290	864 243	530 063	3 044 323	2 465 353
Serbia	167 021	120 811	217 957	166 119	384 978	286 930
Slovakia	80 022	65 329	15 812	7 409	95 834	72 738
Slovenia	17 136	13 824	7 049	4 036	24 186	17 861
Spain	236 869	211 686	–	–	236 869	211 686
Sweden	13 982	14 048	–	–	13 982	14 048
Switzerland	42 587	36 242	–	–	42 587	36 242
Tajikistan	84 273	79 435	210 202	171 393	294 475	250 828
The former Yugoslav Republic of Macedonia	34 960	30 493	43 761	39 826	78 721	70 319
Turkey	730 800	722 346	285 405	187 738	1 016 205	910 084
Turkmenistan	138 504	133 870	12 084	5 255	150 588	139 125
Ukraine	1 471 450	947 069	520 987	260 023	1 992 437	1 207 092
United Kingdom	434 483	360 700	–	–	434 483	360 700
Uzbekistan	529 726	507 522	439 454	324 726	969 180	832 248
Total (of available data)	**10 944 766**	**9 256 004**	**5 120 172**	**3 609 020**	**16 064 938**	**12 865 024**

Note. Monaco and San Marino are excluded owing to lack of data.
Source: data extracted from IHME (2014).

A comparison of Table 1.3 and Table 1.4 shows that the trend in DALYs from air pollution in the WHO European Region over time closely mirrors the trend in number of premature deaths. This is not surprising, because mortality is by far the larger contributor to the BOD; YLDs are but a small fraction of DALYs, as shown in Table 1.5.

However, YLDs also matter. Table 1.5 presents results by country for YLDs as a percentage of DALYs, for ambient PM only (since data are available for 51 rather than 29 countries) and for 2010 only (since the relative share, rather than the change over time, is the focus here). YLDs expressed as a percentage of DALYs reflect not only the prevalence of illness in a given country, but also that country's ability to respond to illness by treating individuals and prolonging their lives. It is therefore unsurprising that high-income countries with the highest standards of health care provision show the highest values in Table 1.5. See, for example, the results for countries such as Israel, as well as various European countries (Austria, Belgium, France, Iceland, Italy, the Netherlands and Switzerland). It follows that, other things being equal, this value could be expected to increase with the general progress of society.

Table 1.5. YLDs from APMP in relation to DALYs from APMP per country in the WHO European Region, 2010

	YLDs from APMP	DALYs from APMP	YLDs from APMP as a % of DALYs
Albania	1 182	29 858	3.96
Andorra	39	469	8.42
Armenia	2 067	49 141	4.21
Austria	4 720	45 883	10.29
Azerbaijan	4 897	130 019	3.77
Belarus	4 161	157 970	2.63
Belgium	10 157	89 698	11.32
Bosnia and Herzegovina	1 498	36 245	4.13
Bulgaria	5 311	164 432	3.23
Croatia	2 333	49 122	4.75
Cyprus	422	5 513	7.66
Czech Republic	5 695	112 463	5.06
Denmark	2 430	27 876	8.72
Estonia	262	5 492	4.77
Finland	539	7 326	7.36
France	31 416	266 018	11.81
Georgia	2 280	66 084	3.45
Germany	55 743	632 545	8.81
Greece	6 955	117 569	5.92
Hungary	6 788	159 555	4.25
Iceland	33	325	10.05
Ireland	1 033	11 451	9.02
Israel	5 500	39 563	13.90
Italy	47 481	436 848	10.87
Kazakhstan	5 577	244 457	2.28
Kyrgyzstan	2 029	74 414	2.73
Latvia	781	19 339	4.04
Lithuania	1 382	29 974	4.61
Luxembourg	232	2 389	9.70
Malta	308	3 606	8.55
Montenegro	390	7 632	5.11
Netherlands	11 901	108 603	10.96
Norway	251	2 769	9.06
Poland	19 878	439 664	4.52
Portugal	3 823	54 689	6.99
Republic of Moldova	1 861	62 037	3.00
Romania	13 056	386 302	3.38
Russian Federation	51 309	1 935 290	2.65
Serbia	5 209	120 811	4.31
Slovakia	2 690	65 329	4.12
Slovenia	1 087	13 824	7.87

Table 1.5. (continued)

	YLDs from APMP	DALYs from APMP	YLDs from APMP as a % of DALYs
Spain	19 895	211 686	9.40
Sweden	1 123	14 048	7.99
Switzerland	4 681	36 242	12.92
Tajikistan	2 071	79 435	2.61
The former Yugoslav Republic of Macedonia	1 031	30 493	3.38
Turkey	24 298	722 346	3.36
Turkmenistan	3 955	133 870	2.95
Ukraine	25 154	947 069	2.66
United Kingdom	29 412	360 700	8.15
Uzbekistan	12 904	507 522	2.54
Minimum value			**2.28**
Maximum value			**13.90**
Value across the WHO European Region			**4.85**

Note. Monaco and San Marino are excluded owing to lack of data.
Source: data extracted from IHME (2014).

An additional reason for the low share of YLDs is that the data record for premature deaths is more complete than for morbidity, and critical data gaps still remain (WHO Regional Office for Europe, 2014). Since the problem of incomplete morbidity data is more pronounced in low-income countries, the share of YLDs is expected to increase over time, as this information gap is filled. These points do not undermine the finding that YLDs make up only a low share of DALYs, but they do suggest that this share will increase over time and bring with it an increasing focus on the issue of morbidity from air pollution.

2. The evidence from economics

2.1 The valuation of life and health

The epidemiological evidence shows that air pollution is responsible for several million premature deaths per year – a global total of 7 million premature deaths in 2012, as reported in WHO (2014b). The evidence from economics shows that such pollution also imposes, by virtue of being responsible for those deaths, a so-called economic cost to society[4] of several trillion dollars per year. As reported by the OECD (OECD, 2014), in the case of AAP for the 34 OECD countries plus China and India, this cost can be estimated at a combined total of 3.5 trillion United States dollars (US$) for the year 2010.

Economists face a difficulty here, as they often need to address conflicting estimates, produced by those who are not specialists in the field and do not understand its first principles. Also, they are often obliged to address those who might challenge the field from debatable philosophical starting points, misleading decision-makers with proposals based on estimates of costs (including health costs), which, explicitly or implicitly, place a default value of zero on the loss of life itself. It therefore appears necessary to restate, briefly, the first principles of economics as they apply to the problem at hand.

From Aristotle to William Petty, and including the many contributions from thinkers in China, India and elsewhere, there has long been a tradition of studying the management of income and wealth – that of households, of sovereigns and even of nations. But the modern science of economics that emerged from, and was a defining part of, the Franco-Scottish Enlightenment of the 18th century – pre-eminently, in the works of Francois Quesnay and Adam Smith – represents a definite break with this preceding tradition.

The new tradition begins with an explicit rejection of the view that wealth consists in gold or some other form of money – what Smith called the chrysohedonistic illusion – and takes as its object of study the so-called real economic phenomena that lie behind the monetary veil. Money becomes not the thing being measured but, at best, the instrument with which to measure it; an imperfect instrument with which to measure non-monetary phenomena.

In the language of present-day economics, which has developed far beyond but nonetheless remains descended from the tradition of Quesnay and Smith, *value* is a measure of the things that individuals in their millions value in the ordinary sense of the word, and *cost* is a measure of their loss, whether absolutely or as a means of securing other valuable things. For the purpose of the present discussion, these things include those listed below.

- Consumption. Along with consumption comes the sacrifice of some items of consumption in order to secure others, including the sacrifice of current consumption in the act of investment in order to secure greater future consumption.
- Leisure. This also entails the sacrifice of some leisure in the act of labour in order to secure consumption.
- Health. This also involves the sacrifice of some part of consumption in order to secure health.

4 In the relevant literature, this economic cost to society is also referred to as social cost, welfare cost, welfare loss or loss in social welfare. These terms indicate the same thing, which in this report is referred to as economic cost, for simplicity.

- Life. This includes the sacrifice of some part of consumption in order to preserve life.

Individuals are obliged to conduct trade-offs, or substitutions, between different valued things on a daily basis – and therefore to value them relative to each other. The task of the economist then becomes one of aggregating at a social level these millions of individual valuations at their marginal rates of substitution. What is being aggregated here, however, is precisely the valuations by individuals of the value to individuals.

As shown in the sections that follow, economics today possesses a standard method by which to execute this task, at least in the case of the cost of mortality, which is by far the largest component of the cost of air pollution. Before proceeding to the analysis, it is important to describe the contributions from other disciplines and why they need to be considered separately from the contribution of economics, and not conflated with it.

It is obvious that in actual cases of illness, and up to the point of death, the various responsible agents – individuals, households, hospitals, governments – must manage their respective budgets. In recent years, there have been important innovations in the allocation of these financial costs (in particular hospital costs) to particular diseases and risk factors; for example, by means of the method known as diagnosis-related groups (DRGs), and – principally in the United States, but also in Europe – through the work of WHO and other organizations (see, for example, Busse et al., 2011).

This work of accounting is obviously important, but is not to be confused with economics. The financial cost to a household, for example, of the premature death of a family member might be no more than the previously saved-up funeral expense; it might be near-zero for the hospital concerned; it might be a negative cost to government and/or a net saving in pension payments. In terms of economics, the cost being estimated is the loss of the thing to which individuals normally attach great value: life itself. By whatever means it is aggregated, the value to the individual – based on the individual's own valuation – is unlikely to be near-zero, let alone negative, and more likely to be positive and significant.

Even before the point of death, even in cases of illness, where it appears as though accountants and economists are considering the same costs, they are not counting the same thing, but rather addressing two features of the same reality. Consider, for example, a night's hospital stay on the part of a given patient. The financial cost may be found in, inter alia, the attributable part of the wages paid to the relevant medical staff, the attributable part of the bills paid to the relevant suppliers of equipment, energy, materials, and so on. In contrast, the economic cost is the sacrifice of value by the individual patient and, if relevant, the patient's household. This entails the sacrifice of consumption as a result of the wages foregone, the sacrifice of leisure as a result of the free time foregone, and so on. There are different calculations at work: they are not interchangeable, nor can their results be added up.

Similar considerations apply to the impact on the national accounts of air pollution or any other health risk factor. The premature deaths of working-age people will have an impact on the national accounts through the loss of labour inputs to production and the outputs of it. Those responsible for measuring, analysing and forecasting changes in gross domestic product (GDP) will have an interest in measuring this impact. Clearly, however, a calculation that stops counting at retirement age and implicitly places a zero value on the death of a person of 65 years or over will yield a very different estimate from the economist's estimate of the value to the individual. Even before the point of death, there are different calculations at work addressing different features of the same reality: counting the lost output as a result of the patient's absence from work is not the same as counting the patient's own loss.

None of this is to deny the validity,

importance or policy relevance of accounting, including national accounting. But the information yielded needs to be considered separately from the information on economic costs. Thus, there is a case for bringing to the attention of decision-makers simultaneously, side-by-side, both the economic cost estimates of air pollution and the estimates of its direct impact on GDP.[5] Section 2.10 of this chapter presents some important recent results from the United States on both sets of estimates: the economic cost and the GDP impact.

2.2 The standard method for calculating the cost of mortalities: value of a statistical life (VSL)

This section focuses on the standard method for estimating economic costs and presents new results that have arisen from applying this method to the WHO European Region. It also shows that common variations on the standard method – within the discipline of economics – do not alter these results significantly: the economic cost is similarly large as long as this is indeed the cost being measured.

Present-day economics possesses a standard method by which to measure the cost of mortality at the level of society as a whole: VSL, as derived from aggregating individuals' willingness to pay (WTP) to secure a marginal reduction in the risk of premature death. Despite its unfortunate name, suggesting a monetary judgement on the worth of an individual life, this method is safely grounded in economic first principles, seeking to aggregate the valuations by individuals of the value to individuals.[6]

The algebraic reasoning informing this method is elegant in its simplicity. Suppose that each individual has an expected utility function, EU, relating the utility of consumption over a given period, U(y), and the risk of dying in that period, r, of the form:

$$EU(y, r) = (1 - r)\, U(y).$$

The individual's WTP, to maintain the same expected utility in the event of a reduction in the level of risk from r to r' is the solution to the equation:

$$EU(y - WTP, r') = EU(y, r).$$

VSL is thus the marginal rate of substitution between these two valued items, consumption and the reduction in the risk of dying, such that:

$$VSL = \delta WTP/\delta r.$$

The simplest way to discover the relevant individuals' WTP is – of course – to ask them. A WTP survey is in fact the starting point of the calculation. The OECD describes the basic process of deriving a VSL value from such a survey (OECD, 2012:14):

> *The survey finds an average WTP of US$ 30 for a reduction in the annual risk of dying from air pollution from 3 in 100 000 to 2 in 100 000. This means that each*

5 See, for example, related works by the United States Environmental Protection Agency (EPA) (EPA, 2011a; 2011b), along with current OECD research exploring the subject (OECD, 2014). Unpublished works on local air pollution matters are also informing related decision-making (for example, a report prepared by Elisa Lanzi (OECD Secretariat) for the 2nd ad-hoc technical workshop of the CIRCLE project on costs of inaction and resources scarcity and the consequences for long-term economic growth, held on 2–3 October 2014 at the OECD headquarters in Paris). There is an interesting parallel here with the issue of GDP impacts from public investment projects. In recent years, and for certain high-profile projects, the United Kingdom Department for Transport has reported results in terms of both economic evaluation and national accounts – that is, both cost–benefit results and GDP impacts – in the same document. That said, these calculations have been carefully presented separately, and the reasons for it explained (see, for example, United Kingdom Department for Transport, 2005).

6 For recent expositions on the subject, including the inevitable complexities and caveats, see (inter alia) Biausque (2012), Braathen (2012), Hunt & Ferguson (2010), Hunt (2011), and OECD (2012; 2014). The exposition in the present study borrows heavily from current OECD research (OECD, 2014).

individual is willing to pay US$ 30 to have this 1 in 100 000 reduction in risk. In this example, for every 100 000 people, one death would be prevented with this risk reduction. Summing the individual WTP values of US$ 30 over 100 000 people gives the VSL value – US$ 3 million in this case. It is important to emphasise that the VSL is not the value of an identified person's life, but rather an aggregation of individual values for small changes in risk of death.

This approach yields a simple result for researchers and policy-makers, which contributes to assessing the magnitude of a given problem, in terms of monetized societal value. The economic cost of the impact being studied – in the present case, the cost of mortality from air pollution – is the VSL value multiplied by the number of premature deaths. The economic benefit of a mitigating action becomes the same VSL value multiplied by the number of prevented deaths.

Owing to the multi-year research effort, embodied in a report by the OECD (OECD, 2012) – including its meta-analysis of VSLs starting with 1095 values from 92 published studies – both researchers and policy-makers can now use a set of OECD-recommended values for the average adult VSL. In units of (2005) US$, the recommended range for OECD countries is US$ 1.5 million to US$ 4.5 million, with a recommended base value of US$ 3 million.

This in turn enables the computation of country-specific VSL values for both OECD and non-OECD countries from 2005 onwards. The sections of this chapter that follow present this computation for the countries of the WHO European Region for 2005 and 2010, together with an exploration of some of the equity issues arising in the derivation and use of such country-specific VSLs.

Some words of reflection on these methods by the originator of the WTP approach, Jacques Drèze, throw into sharp relief their underlying motivation; namely, the failure of accounting to recognize the loss of value to the individual. The original case concerned the cost of mortality from road injury, but the point can easily apply to mortality from air pollution or any other environmental factor. Drèze recalls (Dehes, Drèze & Licandro, 2005:8–9):

In 1960, two French engineers were wondering how much should be spent on investments enhancing road safety. So they tried to define the economic value of a life saved. They suggested measuring that economic value by the future income of a potential victim … and stumbled on the question: should the value of future consumption be subtracted, in order to appraise society's net *loss? I realised at once that this very question pointed to the basic flaw of the approach: people want to survive and consume, not starve! Going back to the root of the problem, I introduced what is known today as the "willingness to pay" approach to valuing lives in safety analysis. How much would an individual be willing to pay in order to reduce his probability of accidental death? That is for the individual to decide, given his resources … [and] the subjective importance he attaches to survival… Road safety being a public good, individual willingness to pay should then be aggregated as in the Lindahl-Samuelson theory of public goods.*

The standard method has undergone many developments and refinements over the subsequent half-century but it remains true to its original point of departure: its fidelity to individual valuations of the value to the individual.

2.3 Country-specific VSLs and intra- and international equity

The meta-analysis of VSL studies and VSL values by the OECD (OECD, 2012) yielded a recommended base value for average adult VSL in OECD countries of US$ 3 million for the year 2005 and using (2005) US$. The derivation of country-specific values, and analysis of years other than 2005, involves two main adjustments:

- an adjustment for differences in per-capita income (per-capita GDP) and with the best-estimated income elasticity, in order to derive the value for any given country for the year 2005;
- an adjustment for post-2005 income growth (ΔY) and price inflation (ΔP) in order to derive values for that given country for years following 2005.

Details of the methodology are described elsewhere (OECD, 2012; Braathen, 2012; OECD, 2014), but it is important to recall five elements in particular.

1. The OECD base value of US$ 3 million is the starting point for the calculation, both for OECD countries and for several other (non-OECD) countries, such as China and India (OECD, 2014).
2. The calculation is performed in purchasing power parity (PPP)-adjusted US$ estimates, and not through national currencies, re-converted into PPP-adjusted US$ estimates.
3. These PPP-adjusted US$ estimates reflect those published in the OECD's statistical database for OECD countries and in the World Bank database for non-OECD countries.
4. The income elasticity beta applied is 0.8, being the mid-point of the best estimate of 0.7–0.9 (as established by the OECD) (OECD, 2012), without use of further sensitivity tests with alternative estimates.
5. The income elasticity adjustment is applied not only to the 2005 level but also to its growth in the post-2005 period.

The result for any given country, C, for any given year (here 2010), is thus:

$$\text{VSL C}_{2010} = \text{VSL OECD}_{2005} \text{ x } (\text{Y C}_{2005}/\text{Y OECD}_{2005})^{\beta} \text{ x } (1 + \%\Delta\text{P} + \%\Delta\text{Y})^{\beta}.$$

As already discussed, a VSL value is an aggregation of individual valuations; that is, WTP figures, as elicited from surveys, to secure a marginal reduction in the risk of premature death. It is a fact of life that, in the present day as much as in the past, individuals are differentially endowed with the means with which to make such a trade-off. At one end of the scale, some are obliged to work for their living for a dollar a day; at the other, some hold an inherited fortune, yielding an income of 1 billion dollars per year. All societies have therefore sought to socialize these risks to a greater or lesser extent in the form of public goods; to share the burden of these risks at least partially through the collective treasury, rather than impose it exclusively on the individual's purse at the point of need, in addition to measures designed to redistribute incomes to a greater or lesser extent. It so happens that the level at which this socialization of risks is executed today is, principally, the level of the nation-state. Thus it is most often appropriate to aggregate at the level of country-specific VSLs, rather than at a lower level, such as a neighbourhood or (more realistically) a city or province, or at a higher level, such as the world as whole or (more realistically) a continent-wide union.

The point here is not that the problem of air pollution is, in the nature of things, national: it is not. Rather, the point is that the burden of addressing the problem and bearing the costs of any solution – that is, effecting the sacrifice of some value in consumption in order to secure the greater value of lives saved – is, in the present day, principally the responsibility of national governments.

National-level VSL is an aggregate value, reflecting the level at which the socialization of risks is executed. One consequence is that, as far as the use of VSLs is concerned, the problem of *intra*-national variability in the ability to make the relevant trade-offs is supressed. To the extent that income inequality occurs within countries, within-country variability in WTP also exists; however, national VSLs average out such variability and reflect the monetary valuations made by people in different countries about any good, including risk.

Differences in country-specific VSL values will thus tend to mirror the differences in country-specific per-capita income levels in any given year, as shown in Table 2.1. Because differences in income are very pronounced across the region, so are VSLs, with the highest value around 22 times the lowest.

This result is not a normative judgement on the part of the economists; it is simply recognition of the present-day reality: the citizens of low-income countries execute their relevant trade-offs largely without reference to the resources of high-income ones. The economist's calculation would change if in fact the socialization of risks were devolved to a lower level, or elevated to a higher level.

The latter possibility is not far-fetched, given the role played by EU institutions and legislation in EU countries. Therefore, aside from the calculation of country-specific VSLs based on the OECD data formula for the countries of the WHO European Region (OECD, 2014) and the calculation of the economic costs of air pollution on the basis of these country-specific values, this chapter also presents an alternative calculation of the economic costs of air pollution for EU countries, on the basis of a common EU-wide VSL value.

Finally, it is important to note that the phenomenon of differential rates of growth in per-capita income in fact acts to change the differences in country-specific VSLs. Normally, this entails a movement toward the convergence of VSL values, as the lower income countries catch up with the higher income ones, as becomes apparent by comparison of the many countries in Table 2.1 (for example, the ratio of Germany's VSL to Poland's VSL falls from about 2:1 in 2005 to about 1.5:1 in 2010). However, issues of international equity do exist; they lie in different means and abilities to prevent and respond to environmental threats, including trans-boundary ones, such as air pollution and, even more so, climate change. The variability of VSLs by country reflects the different levels of resources available to deal with environmental risks – an issue that can only be addressed in international discussion of burden-sharing arrangements or the lack thereof.

Table 2.1. Computed country-specific VSL values per country in the WHO European Region, 2005 and 2010

	OECD VSL base value (2005) US$ (millions)	Country-specific VSL (2005) US$ (millions)[1]	Country-specific VSL (2010) US$ (millions)[2]
Albania	3.00	0.83	1.11
Armenia	3.00	0.62	0.83
Austria	3.00	3.28	3.67
Azerbaijan	3.00	0.66	1.45
Belarus	3.00	1.11	2.01
Belgium	3.00	3.17	3.50
Bosnia and Herzegovina	3.00	0.85	1.06
Bulgaria	3.00	1.23	1.77
Croatia	3.00	1.75	2.07
Cyprus	3.00	2.54	2.87
Czech Republic	3.00	2.28	2.75
Denmark	3.00	3.25	3.46
Estonia	3.00	1.86	2.27
Finland	3.00	3.05	3.32

Table 2.1. (continued)

	OECD VSL base value (2005) US$ (millions)	Country-specific VSL (2005) US$ (millions)[1]	Country-specific VSL (2010) US$ (millions)[2]
France	3.00	2.96	3.16
Georgia	3.00	0.55	0.84
Germany	3.00	3.09	3.48
Greece	3.00	2.54	2.82
Hungary	3.00	1.90	2.32
Iceland	3.00	3.39	4.46
Ireland	3.00	3.68	3.75
Israel	3.00	2.44	2.92
Italy	3.00	2.86	3.00
Kazakhstan	3.00	1.11	1.85
Kyrgyzstan	3.00	0.30	0.49
Latvia	3.00	1.54	2.10
Lithuania	3.00	1.65	2.15
Luxembourg	3.00	5.78	6.28
Malta	3.00	2.25	2.65
Montenegro	3.00	1.08	1.45
Netherlands	3.00	3.40	3.76
Norway	3.00	4.34	4.65
Poland	3.00	1.61	2.10
Portugal	3.00	2.28	2.50
Republic of Moldova	3.00	0.39	0.63
Romania	3.00	1.18	1.67
Russian Federation	3.00	1.42	2.40
Serbia	3.00	1.09	1.75
Slovakia	3.00	1.83	2.42
Slovenia	3.00	2.46	2.90
Spain	3.00	2.79	3.06
Sweden	3.00	3.21	3.50
Switzerland	3.00	3.52	3.85
Tajikistan	3.00	0.26	0.44
The former Yugoslav Republic of Macedonia	3.00	1.01	1.26
Turkey	3.00	1.38	2.02
Turkmenistan	3.00	0.69	0.97
Ukraine	3.00	0.78	1.42
United Kingdom	3.00	3.26	3.55
Uzbekistan	3.0	0.34	0.44

Notes. Andorra, Monaco and San Marino are excluded owing to incomplete data. All presented numbers have been rounded up/down after the first two digits.

1 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8.

2 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8, and adjusted for post-2005 income growth and inflation.

Sources: data for OECD and non-OECD countries, respectively, were extracted from OECD (2013) and World Bank (2013).

2.4 Estimating the cost of morbidity

Economics possesses a standard method by which to measure the cost of mortality. However, as was argued at some length by the OECD (OECD, 2014), it does not yet possess a standard method by which to measure the cost of morbidity. Nor do researchers and policy-makers possess anything like a set of OECD-recommended values for the several disease outcomes at issue here.

With regard to morbidity, there is not yet a clear consensus on exactly what outcomes need to be calculated or the values at which they are to be calculated. This is not entirely surprising. As discussed by the OECD (OECD, 2014), a defensible calculation of the costs of morbidity, grounded in economic first principles, is necessarily a more complex exercise than the calculation of the cost of mortality, as these costs are, in reality, plural in several respects.

- Morbidity includes a plurality of endpoints, varying greatly in extent of severity, and complicating enormously the task of eliciting and aggregating individual WTP values.
- Morbidity imposes costs on a plurality of agents: to begin with, the individual who is suffering ill health, but also the many who are involved in the organization and execution of formal and informal care of ill individuals.
- The individual who is suffering ill health suffers a plural loss: not only the pain and suffering imposed by the illness but also the loss of some part of consumption (and leisure) in expending income (and time) in averting and mitigating activities in response to current and prospective morbidities.

Therefore, and without departing from the distinction between economic calculation and other forms of calculation, such as national accounting, it is legitimate to calculate the costs of morbidity in a plural manner, as the sum of separate elements of cost, as listed by Hunt & Ferguson (2010).

- *Resource costs* are represented by the direct medical and non-medical costs associated with treatment for the adverse health impact of air pollution, plus avertive expenditures. That is, all the expenses the individual faces visiting a doctor, ambulance costs, purchasing medicines and other treatments, plus any related non-medical cost, such as the cost of childcare and housekeeping owing to fact that the affected person cannot carry out these tasks.
- *Opportunity costs* are associated with the indirect costs related to loss of productivity and/or leisure time owing to the health impact.
- *Disutility costs* refer to the pain, suffering, discomfort and anxiety linked to the illness.

As already mentioned, research is currently under way on establishing the methodology, and currently available estimates are indicative. As noted by Hunt & Ferguson (2010), Hunt (2011) and the OECD (2014), several issues remain to be resolved, including:

- the need to specify the distinct endpoints to be captured in the cost calculation;
- the necessity to include each of the aforementioned three separate elements of cost (resource costs, opportunity costs and disutility costs;
- the need for consistency between methods for estimating the different cost elements and in particular, the importance of avoiding double-counting;
- the necessity to conduct and complete this complex search in a manageable manner, in order to provide readily useable information.

Nonetheless, as argued by the OECD

(OECD, 2014), these difficulties do not justify abandoning this line research in favour of short cuts that would abandon the principle of evaluating the loss to the individual. As discussed in Chapter 1, over time and with the progress of society, morbidity is likely to become relatively more important. Its relative weight is in part a reflection of society's success in keeping ill people alive, thus saving them from premature death.

For the present, it seems preferable to choose an indicative estimate for the additional cost of morbidity from the most comprehensive recent studies available. Quantitatively, however, this is not necessarily a serious limitation when estimating the economic cost of the BOD of air pollution, because mortality dominates over morbidity. As shown in the previous chapter, in current estimates YLDs are a small fraction of DALYs from air pollution, at about 5% of the sum of DALYs across the WHO European Region. Also, as shown in section 2.5, the most recent estimates of the economic cost of morbidity are below 10% of the overall economic cost of air pollution's health impacts.

2.5 An indicative estimate for the additional cost of morbidity

The chosen indicative estimate for the additional cost of morbidity in the present study is – as shown by the OECD (OECD, 2014) – about 10%. This implies that morbidity constitutes approximately 9%, or < 10%, of the estimated total cost of health impacts from air pollution, with mortality accounting for about 91%, or > 90%, of the total.

The rationale for this choice is set out in the paragraphs that follow, but it bears repeating that this is no more than an additional indicative estimate. The primary and definite estimates on which the calculations in this study rest are those conducted for the economic cost of premature deaths from air pollution.

The recent past has seen the publication of two comprehensive cost–benefit analysis (CBA) studies on air pollution, one on each side of the Atlantic. The first is the evaluation conducted by the United States EPA of the 1990 Clean Air Act Amendments (CAAA) (see EPA, 2011a; 2011b). The second is the set of studies supporting the EU's Clean Air Policy Package (CAPP) (see in particular, Holland (2014), and also the earlier version of that report (Holland, 2012), as well as Amann (2014) and the European Commission (EC, 2013)). Neither the work conducted in the United States nor that from the EU are directly concerned with the problem of estimating the economic cost of morbidity for the WHO European Region. However, the evidence they provide on the share of morbidity costs helps to underpin the indicative estimate chosen here.

The United States EPA (EPA, 2011a) provides a series of estimates of the annual monetized benefits – that is, of the economic benefits from the reduction in mortality and morbidity, as well as the benefits from the reduction in environmental impacts other than health – by target years and also cumulatively. The specific estimates are not relevant here; what is of interest is the distribution of the benefits gained, or, more precisely, its counterpart; that is, the distribution of the economic costs saved. The information in Table 2.2 is extracted from the central estimate in the final analysis for the year 2020.

Table 2.2. Estimated share of mortality costs saved in the United States EPA CAAA CBA for the year 2020

Category	%
PM mortality as a % of health effects	89.5
Ozone mortality as a % of health effects	2.9
Sum of PM mortality and ozone mortality as a % of health effects	92.4
Sum of health effects as a % of total effects	95.0
Sum of PM mortality and ozone mortality as a % of total effects	87.8

Source: extracted from results given in Table 7-2 (in US$) by the United States EPA (EPA, 2011a).

It is evident that mortality has a dominant share in the health effects of air pollution: in this case, 92.4%. Indeed, since health effects have a dominant share in the total effects of air pollution (95% in this case), mortality alone has a dominant share in the latter (87.8%, here).

From this analysis it is permissible to deduce that if morbidity costs were to amount to about 7.5% of the total of health effects, then an addition of approximately 8% to the estimated mortality cost would suffice to provide an indicative estimate of the overall economic cost of health impacts.

In the case of the EU, Holland provides a CBA of various mitigation scenarios, estimated with various methods (Holland, 2014). Again, specific estimates of these benefits are not relevant here.

What is of interest is the distribution of the monetized equivalent of health impacts in the baseline scenario – the set of impacts expected to be obtained under the CLE – and its estimation with the standard method only; that is, with mortality calculated at mean VSL. The information in Table 2.3 is extracted from the CLE baseline for the year 2025:

Table 2.3. Estimated share of mortality costs in the EU CAPP CBA for the year 2025

Category	%
Sum of chronic mortality (30 years +) at mean VSL and infant mortality (0–1 year +) at mean VSL as a % of the total of monetized equivalent of health impacts, if mortality is calculated at mean VSL	91.9

Source: extracted from results given in Table 3.3 (in €) by Holland (2014).

Here, too, it is evident that mortality has a dominant share (91.9%). It is permissible again here to deduce that if morbidity costs amount to about 8% of health impacts, then an addition of about 9% to the estimated mortality cost provides an indicative estimate of the overall economic cost of health impacts.[7]

7 The OECD (OECD, 2014) used an earlier version of this evaluation defined by Holland (Holland, 2012), since Holland's later work (Holland, 2014) was evidently not yet available. As a result, the estimate cited therein was approximately a 9% share for morbidity costs and the formula adopted for the indicative estimate was an addition of about 10% to mortality costs.

Moreover, since the studies cited above are studies of high-income countries and since the evaluation year chosen for the Table 2.2 and Table 2.3 are in the future, rather than current or past years, the result ought to provide an automatic correction to at least one of the sources of the downward bias to estimations of morbidity costs as described in the preceding discussion.

Nonetheless, it remains the case that the information extracted is best used here as a guide – it is too early to suggest that all potentially significant biases to the estimation have been corrected. Used thus, it does provide sufficient guidance in favour of adding 10% to the primary estimate of mortality costs in the calculations that follow.

2.6 The economic cost of health impacts of air pollution in the WHO European Region

Table 2.4 presents estimates of the economic cost of premature deaths from air pollution, per country, for 2005 and 2010. It does so on two counts: from APMP, for which data are available for all Member States of the WHO European Region other than Andorra, Monaco and San Marino; and from the sum of APMP and HAP, for which a default value of zero for HAP is applied to those 24 high-income countries where no deaths are recorded in the data.

Table 2.4. Economic cost of premature deaths from air pollution (APMP and APMP + HAP) per country in the WHO European Region, 2005 and 2010

	Economic cost of premature deaths from APMP US$ (millions)		Economic cost of premature deaths from APMP + HAP US$ (millions)	
	2005[1]	2010[2]	2005[1]	2010[2]
Albania	1 358	1 673	3 622	4 572
Armenia	1 599	2 160	3 398	3 690
Austria	11 957	11 457	11 957	11 457
Azerbaijan	3 377	7 415	5 893	10 042
Belarus	9 296	16 534	12 900	19 865
Belgium	19 559	19 842	19 559	19 842
Bosnia and Herzegovina	1 838	2 146	5 920	7 228
Bulgaria	13 803	16 788	2 182	32 091
Croatia	6 465	6 316	9 844	9 035
Cyprus	819	857	819	857
Czech Republic	19 862	19 321	22 834	20 901
Denmark	5 955	6 283	5 955	6 283
Estonia	351	796	1 867	2 015
Finland	1 179	1 495	1 179	1 495
France	53 031	53 295	53 031	53 295
Georgia	1 636	2 766	5 562	9 127
Germany	154 382	144 715	154 382	144 715
Greece	22 300	22 785	22 300	22 785

Table 2.4. (continued)

	Economic cost of premature deaths from APMP US$ (millions)		Economic cost of premature deaths from APMP + HAP US$ (millions)	
	2005[1]	2010[2]	2005[1]	2010[2]
Hungary	21 839	21 281	41 051	40 859
Iceland	62	96	62	96
Ireland	1 773	2 518	1 773	2 518
Israel	6 227	7 164	6 227	7 164
Italy	98 612	97 193	98 612	97 193
Kazakhstan	12 752	18 585	23 168	29 226
Kyrgyzstan	1 029	1 389	2 597	3 571
Latvia	612	2 404	2 401	3 779
Lithuania	2 314	3 812	5 043	6 041
Luxembourg	1 035	913	1 035	913
Malta	521	602	521	602
Montenegro	519	567	1 072	1 202
Netherlands	26 594	24 644	26 594	24 644
Norway	1 533	864	1 533	864
Poland	47 121	51 870	90 547	101 826
Portugal	7 885	9 205	7 885	9 205
Republic of Moldova	1 688	2 028	2 724	3 208
Romania	30 931	36 109	53 664	62 028
Russian Federation	139 423	225 975	193 176	285 467
Serbia	10 185	12 420	23 019	28 850
Slovakia	8 246	9 134	9 727	10 074
Slovenia	2 489	2 539	3 441	3 226
Spain	42 124	42 951	42 124	42 951
Sweden	3 219	3 641	3 219	3 641
Switzerland	10 471	10 225	10 471	10 225
Tajikistan	722	1 226	2 071	3 199
The former Yugoslav Republic of Macedonia	1 834	2 094	4 061	4 755
Turkey	37 524	56 932	50 639	70 386
Turkmenistan	3 379	4 791	3 629	4 951
Ukraine	59 655	74 935	79 572	94 201
United Kingdom	89 741	83 069	89 741	83 069
Uzbekistan	6 400	8 299	10 303	12 267
Total (of available data)	**1 007 223**	**1 156 118**	**1 258 904**	**1 431 499**

Notes. Andorra, Monaco and San Marino are excluded owing to incomplete data.

1 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8.

2 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8, and adjusted for post-2005 income growth and inflation.

Sources: data on economic indicators for OECD and non-OECD countries, respectively, were extracted from OECD (2013) and World Bank (2013); data on deaths for all countries were extracted from the IHME (2014).

Table 2.5 adds the chosen indicative estimate of 10% to the economic cost of premature deaths, in order to present an indicative estimate of the economic cost of health impacts from air pollution across the WHO European Region (excluding Andorra, Monaco and San Marino) for 2005 and 2010, at an aggregate level.

Table 2.5. Indicative estimate of the economic cost of health impacts from air pollution (APMP and APMP + HAP) across the WHO European Region, 2005 and 2010

	Economic cost of health impacts from APMP if morbidity costs add ≈ 10% US$ (millions)		Economic cost of health impacts from APMP + HAP if morbidity costs add ≈ 10% US$ (millions)	
	2005	2010	2005	2010
Indicative total across the WHO European Region	≈ 1 107 945	≈ 1 271 730	≈ 1 384 794	≈ 1 574 649

Note. Andorra, Monaco and San Marino are excluded owing to incomplete data.
Sources: data for OECD and non-OECD countries, respectively, were extracted from OECD (2013) and World Bank (2013); data on deaths for all countries were extracted from IHME (2014).

In absolute terms, the result is clear: air pollution imposes a large economic cost on the countries of the WHO European Region. As at 2010, the annual cost of premature deaths from air pollution across the countries of the Region stood at US$ 1.4 trillion, and the overall annual cost of health impacts from air pollution stood at US$ 1.6 trillion (see Box 2.1 at the end of this chapter).

This is to be expected for as long as air pollution remains a leading risk factor in premature deaths and as long as economists are willing to record the valuations that individuals report through their stated WTP to reduce the risk of premature death.

In terms of the change over time, the result is more complex. The cost of premature deaths from air pollution increased from 2005 to 2010 despite the decreasing number of deaths: the reduction of approximately 12% in premature deaths from APMP over this period has been attended by a 14.8% increase in estimated cost, and the reduction of 15% in premature deaths from APMP + HAP has been attended by a 13.7% increase in estimated cost. This divergence is the result of the increase in the VSL value for each death outpacing the reduction in the number of deaths. The pace of reduction in premature deaths has been too slow to counter an increase in cost, even if as many as 11 countries (around a quarter of the list) did achieve the desired outcome on both counts.

This is an important result: the increase in VSL values is no economic artefact, but rather the indication of a widespread trend. Increasing affluence brings in its wake a greater ability to secure a reduction in the risk of premature death by means of the requisite sacrifice in consumption.[8] Society has signalled the requisite willingness, so it is for decision-makers to act on this signal.

8 Note that this refers to a greater ability to sacrifice consumption and not to a greater sacrifice. On the contrary, the calculation here assumes a lesser sacrifice: that is, the income elasticity is < 1. Thus, VSL values rise with increasing incomes but are assumed here to rise at a lesser rate. As it happens, although this assumption is based on an extensive body of evidence in high-income countries (see OECD, 2012), there is reason to suppose and evidence to suggest that the assumption is too conservative in the case of low-income countries transitioning to middle- and high-income status; it may be that, across this income range, income elasticity is > 1 and that VSL values increase at a greater rate than assumed here (see Hammitt & Robinson, 2011). If so, the rate of increase in the economic cost of premature deaths from air pollution in at least some of the non-OECD countries of the WHO European Region would be greater than that reported in Table 2.4, while starting from a lower base.

2.7 Economic cost calculation in relation to GDP

These estimates are the result of using a particular method and a particular set of assumptions. However, as shown in the subsequent sections of this chapter, this result is insensitive to the most common variations on this method and these assumptions: as long as the physical toll from air pollution remains as it is, economic analysis will yield the result that the economic cost of this toll is large.

To put the figures in context, it may be useful to describe this cost not only in absolute terms (dollars) but also in relation to the metric by which many countries measure themselves nowadays: GDP. Table 2.6 reports the economic cost of premature deaths from air pollution as a percentage of GDP for each of the countries of the WHO European Region.

Table 2.6. Economic cost of premature deaths from air pollution (APMP + HAP) as a percentage of GDP per country in the WHO European, 2005 and 2010

	Economic cost of premature deaths from APMP + HAP US$ (millions)		Economic cost of premature deaths from APMP + HAP as a % of GDP (at PPP)	
	2005[1]	2010[2]	2005[1]	2010[2]
Albania	3 622	4 572	18.89	16.9
Armenia	3 398	3 690	23.90	19.5
Austria	11 957	11 457	4.19	3.3
Azerbaijan	5 893	10 042	9.80	7.1
Belarus	12 900	19 865	13.82	13.6
Belgium	19 559	19 842	5.65	4.6
Bosnia and Herzegovina	5 920	7 228	24.79	21.5
Bulgaria	26 182	32 091	34.01	29.5
Croatia	9 844	9 035	14.26	10.8
Cyprus	819	857	4.43	3.3
Czech Republic	22 834	20 901	10.03	7.4
Denmark	5 955	6 283	3.22	2.7
Estonia	1 867	2 015	8.32	7.2
Finland	1 179	1 495	0.70	0.7
France	53 031	53 295	2.76	2.3
Georgia	5 562	9 127	30.37	35.2
Germany	154 382	144 715	5.82	4.5
Greece	22 300	22 785	8.00	7.1
Hungary	41 051	40 859	23.64	19.0
Iceland	62	96	0.58	0.8
Ireland	1 773	2 518	1.06	1.3
Israel	6 227	7 164	3.85	3.3
Italy	98 612	97 193	5.73	4.7
Kazakhstan	23 168	29 226	11.00	9.3
Kyrgyzstan	2 597	3 571	23.84	24.0
Latvia	2 401	3 779	8.00	10.2
Lithuania	5 043	6 041	10.35	9.8

Table 2.6. (continued)

	Economic cost of premature deaths from APMP + HAP US$ (millions)		Economic cost of premature deaths from APMP + HAP as a % of GDP (at PPP)	
	2005[1]	2010[2]	2005[1]	2010[2]
Luxembourg	1 035	913	3.31	2.1
Malta	521	602	6.14	5.4
Montenegro	1 072	1 202	20.76	14.5
Netherlands	26 594	24 644	4.41	3.3
Norway	1 533	864	0.70	0.3
Poland	90 547	101 826	17.18	12.9
Portugal	7 885	9 205	3.40	3.2
Republic of Moldova	2 724	3 208	25.73	23.5
Romania	53 664	62 028	26.43	18.8
Russian Federation	193 176	285 467	11.39	9.8
Serbia	23 019	28 850	34.91	33.5
Slovakia	9 727	10 074	10.92	7.6
Slovenia	3 441	3 226	7.20	5.7
Spain	42 124	42 951	3.46	2.8
Sweden	3 219	3 641	1.04	0.9
Switzerland	10 471	10 225	3.60	2.5
Tajikistan	2 071	3 199	19.88	20.3
The former Yugoslav Republic of Macedonia	1 834	4 755	25.31	19.9
Turkey	50 639	70 386	6.48	6.0
Turkmenistan	3 629	4 951	13.20	10.0
Ukraine	79 572	94 201	26.08	26.7
United Kingdom	89 741	83 069	4.30	3.7
Uzbekistan	10 303	12 267	14.54	10.5

Notes. Andorra, Monaco and San Marino are excluded owing to incomplete data.

1 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8.

2 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8, and adjusted for post-2005 income growth and inflation.

Sources: data on economic indicators for OECD and non-OECD countries, respectively, were extracted from OECD (2013) and World Bank (2013); data on deaths for all countries were extracted from IHME (2014); data on GDP at PPP were extracted from World Bank (2013).

Given that so much public debate and public dispute – whether with regard to wages, pensions, schools or hospitals – is concerned with differences between rival options that amount to far less than 1% of GDP (that is, given that the GDP (at PPP) of France or Germany or the United Kingdom is approximately US$ 2 trillion and that the aforementioned public debate and public dispute are primarily concerned with items that amount to less than US$ 20 billion), perhaps the most the remarkable feature of the results reported in Table 2.6 is that, as at 2010, the economic cost of premature deaths from air pollution amounted to less than 1% of GDP in only 4 of the 48 countries of the WHO European Region (for which results are available): Finland, Iceland, Norway and Sweden.

In 22 of the 48 countries for which results are available – almost half the number of countries in the WHO European Region – the economic cost of premature deaths from air pollution is, in round numbers, at or above 10% of GDP. Furthermore, in no less than 10 of the countries of the Region, the economic cost of premature deaths from air pollution is, in round numbers, at or above 20% of GDP.

2.8 Economic cost calculation with a common VSL for EU countries

The rationale of using country-specific VSLs is discussed in section 2.3. The VSL seeks to aggregate the willingness of individuals to sacrifice some part of their consumption to secure a reduction in the risk of premature death. In the world as it is today, this trade-off between consumption and risk reduction is, as a general rule, conducted within the boundaries of each country, with national governments bearing the principal responsibility for effecting this trade-off, by means of legislation, regulation, taxation and public expenditure.

However, as already noted, it would be perfectly possible for this trade-off to be conducted at the level of such supranational entities as the EU. Indeed, the European Commission already bears an important measure of responsibility in this regard, by virtue of its capacity to propose legislation and regulation for Member States to agree, adopt and apply on an EU-wide basis.

Therefore, recognizing the possibility of a future reconfiguration of responsibilities in order to permit the trade-off between consumption and risk reduction to be conducted fully on a EU-wide basis, Table 2.7 calculates the per-country economic cost of premature deaths from air pollution for 2005 and 2010 for the (then) Member States of the EU on the basis of a common VSL – starting with a 2005 base figure of US$ 3.6 million as determined in the research presented by the OECD (OECD, 2012). Table 2.8 adds the chosen indicative estimate of 10% for morbidity to the economic cost of premature deaths in order to present an indicative estimate of the economic cost of health impacts from air pollution across the EU for 2005 and 2010, conducted on the same basis.

As shown in the two tables, this alternative calculation does not alter the key conclusion that the economic cost of air pollution is large. Rather, the cost appears even larger: the overall economic cost of health impacts from the sum of APMP and HAP in 2010 becomes US$ 1.483 trillion for the EU alone, as opposed to the US$ 1.575 trillion reported in Table 2.5 for the WHO European Region as a whole.

Table 2.7. Economic cost (with a common VSL) of premature deaths from air pollution (APMP and APMP + HAP) per EU Member State, 2005 and 2010

	Economic cost of premature deaths from APMP US$ (millions)		Economic cost of premature deaths from APMP + HAP US$ (millions)	
	2005[1]	2010[2]	2005[1]	2010[2]
Austria	13 112	12 839	13 112	12 839
Belgium	22 209	23 292	22 209	23 292
Bulgaria	40 568	39 041	76 950	74 628
Cyprus	1 161	1 229	1 161	1 229
Czech Republic	31 430	28 906	36 133	31 271

Table 2.7. (continued)

	Economic cost of premature deaths from APMP US$ (millions)		Economic cost of premature deaths from APMP + HAP US$ (millions)	
	2005[1]	2010[2]	2005[1]	2010[2]
Denmark	6 600	7 476	6 600	7 476
Estonia	680	1 443	3 614	3 652
Finland	1 390	1 852	1 390	1 852
France	64 497	69 478	64 497	69 478
Germany	180 183	171 026	180 183	171 026
Greece	31 669	33 183	31 669	33 183
Hungary	41 391	37 793	77 802	72 560
Ireland	1 736	2 762	1 736	2 762
Italy	124 240	133 453	124 240	133 453
Latvia	1 432	4 711	5 619	7 406
Lithuania	5 059	7 283	11 025	11 543
Luxembourg	645	597	645	597
Malta	833	936	833	936
Netherlands	28 180	26 951	28 180	26 951
Poland	105 482	101 709	202 696	199 663
Portugal	12 430	15 147	12 430	15 147
Romania	94 370	89 147	163 727	153 137
Slovakia	16 243	15 534	19 160	17 133
Slovenia	3 640	3 603	5 032	4 578
Spain	54 443	57 756	54 443	57 756
Sweden	3 610	4 277	3 610	4 277
United Kingdom	99 165	96 134	99 165	96 134
Total (of available data)	**986 397**	**1 064 559**	**1 247 861**	**1 347 777**

Notes. Only those countries that were Member States of the EU in 2005 and 2010 are included.

1 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8

2 OECD base value of US$ 3 million in 2005, adjusted for differences in per-capita GDP at PPP, with an income elasticity to the power of 0.8, and adjusted for post-2005 income growth and inflation.

Sources: data on economic indicators were extracted from OECD (2013); data on deaths were extracted from IHME (2014).

Table 2.8. Indicative estimate of economic cost (with a common VSL) of health impacts from air pollution (APMP and APMP + HAP) per EU Member State, 2005 and 2010

	Economic cost of health impacts from APMP if morbidity costs add ≈ 10% US$ (millions)		Economic cost of health impacts from APMP + HAP if morbidity costs add ≈ 10% US$ (millions)	
	2005	2010	2005	2010
Indicative total across the EU	≈ 1 085 037	≈ 1 171 015	≈ 1 372 647	≈ 1 482 555

Note. Only those countries that were Member States of the EU in 2005 and 2010 are included.
Sources: data on economic indicators were extracted from OECD (2013); data on deaths were extracted from IHME (2014).

Despite the relative stability of the overall cost estimates, the alteration in the detailed pattern of results for each country when using a common VSL is not insignificant. If it were to be adopted as a policy guide in a world in which nation-states bear the primary responsibility for effecting the trade-off between consumption and risk reduction, the change would not be inconsequential.

For example: whereas Table 2.4 reports for the year 2005 an economic cost of premature deaths from APMP + HAP of about US$ 90 billion in Poland, the equivalent figure in Table 2.7, using country-specific VSLs, is about US$ 200 billion. The data in Table 2.7 therefore show Poland's cost burden to be larger than Germany's, using a common VSLs for EU countries, rather than smaller, as is the case with the country-specific VSL, shown in Table 2.4 (APMP +HAP). Moreover, there is also a significant alteration in the pattern of per-country results with regard to the change over time. Comparing 2010 with 2005, Table 2.4 reports an appreciable increase in the economic cost of premature deaths from APMP + HAP in Poland, reflecting the relatively rapid increase in per-capita incomes and VSL values. On this reading, with its rising incomes, Poland was more able and willing in 2010 to sacrifice consumption in order to secure a reduction in air pollution risk than it had been in 2005. In contrast, with a common VSL, and only a moderate change in its value owing to slow-growing incomes in high-income EU countries, Table 2.7 reports Poland's cost burden as little changed. The signal of a greater ability to pay and a great WTP is therefore lost.

In view of this, the alternative calculation with a common VSL is perhaps best interpreted as a confirmation of the overarching conclusion that the economic cost burden imposed by air pollution is similarly large by any reasonable measure, rather than as a guide to policy outlining details of costs and responsibilities for each individual country.

2.9 Economic cost calculation using values of life-years (VOLYs) lost in lieu of VSLs

The variation presented in section 2.8 is an alternative calculation of VSL, using a single common value rather than differentiated country-specific values. There is another variation to the standard method that merits attention here: the use of units of VOLY (sometimes called the value of a statistical life-year (VSLY)) as an alternative to using units of VSL. As discussed in this section, there are some important theoretical issues involved in the choice between VSLs and VOLYs. However, remarkably – and importantly – there is little practical difference in terms of the outcomes in relation to the matter at hand. The use of VOLYs also serves to confirm the overarching conclusion about the size of the cost burden of air pollution in Europe today and the related policy implications.

First, it needs to be emphasized that the two epidemiological metrics used in these alternative economic calculations – the metric of YLLs (also called LYLs) and that of lives lost, or excess premature deaths – are equally legitimate metrics for counting the cost of air pollution. However, the former may be regarded as a more accurate indication of the mortality impact, given its ability to discriminate how long a premature death is moved forward in time (see COMEAP, 2010).

In the view of this study (as argued in Chapter 1), each of the relevant epidemiological metrics – attributed premature deaths, LYLs/YLLs, YLDs, DALYs/quality-adjusted life-years (QALYs), inter alia – remains a valuable aid to arriving at a full understanding of the impacts of air pollution. However, economists cannot be entirely indifferent to the choice between VSLs and VOLYs: that is, the choice between these alternative translations of the epidemiological metrics into economic metrics.

As a matter of historical record, the use of VSLs – having been originally developed with explicit reference to economic first principles and in particular the principle of individual valuations of the value to the individual – has been established as a standard method for measuring the cost of mortalities since the 1960s. The use of VOLYs emerged much later and has developed somewhat unevenly since the 1990s. In the United States, the United States EPA Science Advisory Board advised in favour of the continued use of VSLs, arguing that "alternative measures, such as the value of a statistical life-year or the value of a QALY, are not consistent with the standard theory of individual WTP for mortality risk reduction" (EPA, 2001:26). More recently, both the United Kingdom Government and the European Commission have conducted calculations using VOLYs (see COMPEAP, 2010, and EC, 2013), although without abandoning VSLs altogether (see Hunt & Ferguson, 2010; Hunt, 2011; Robinson & Hammitt, 2013).

Turning to the present day, it remains the case that most governments across the countries of the OECD and beyond continue to calculate in VSLs rather than VOLYs. This may be explained by some open questions on the methodology underlying the use of VOLYs (discussed by the OECD (OECD, 2014)), including in particular two key points.

1. VOLYs are rarely derived from WTP surveys (Hunt, 2011) – even if it is in principle possible to do so – and they therefore reflect the valuations of external parties, such as health professionals, rather than valuations by representative individuals in the general population.
2. However derived, VOLYs will produce results that differ from, and are inconsistent with, the results given by VSLs: the cost of the premature death of a group of people of a given age will automatically be counted as less than the premature death of a comparable group of younger people with otherwise identical characteristics, since the number of YLLs for the former group will be less than that for the latter. It follows that VOLYs, by counting life-years rather than lives in this calculation, "explicitly places a lower value on reductions in mortality risk accruing to older populations" (Hubbel, 2002:22).

However, how do the different metrics compare? In the final CBA for the EU CAPP, Holland (2014) details the economic benefits – that is, the reduction in economic costs – achievable from reducing air pollution in the EU. The study considers the (monetary) costs and (economic) benefits of a range of progressively ambitious pollution-reducing scenarios relative to the CLE baseline. Table 2.9 shows the estimated outcomes for 2030 for two scenarios: the first involves moving from the baseline to the European Commission's proposal (B7) and the second involves moving beyond the Commission's proposal to the maximum technically feasible reduction (MTFR). Estimates are given for four different metrics: median VOLY, mean VOLY, median VSL and mean VSL.

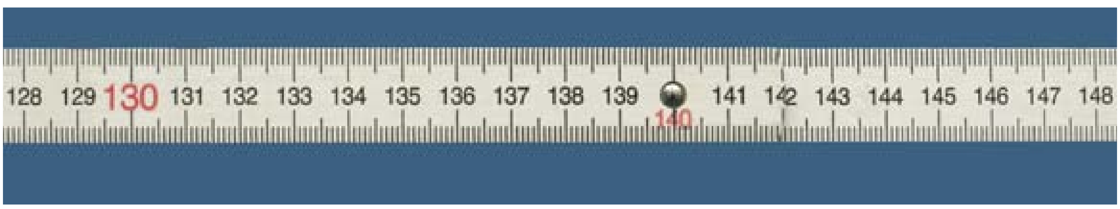

Table 2.9. Estimated net health benefits of alternative scenarios using alternative metrics in the EU CAPP CBA, 2030

Net health benefits in 2030 (28 Member States of the EU)*	€ (millions)	
	CLE – B7	B7 – MTFR
Costs	3 334	47 347
Net benefits with alternative metrics		
Total with median VOLY	35 140	–28 063
Total with mean VOLY	74 437	–8 606
Total with median VSL	70 012	–11 059
Total with mean VSL	135 371	21 002
WHO European Region	509 100	–11.8

Note. *Countries belonging to the EU in 2014.
Source: extracted from Table 5.2 in Holland (2014).

The different metrics produce different results. In moving from the baseline to the European Commission's proposal, the use of a median VSL produces an estimate of benefits that is roughly twice as large as the estimate produced by using a median VOLY, as does the use of a mean VOLY. The use of a mean VSL produces an estimate that is roughly twice as large as that produced by the use of a mean VOLY (and, therewith, roughly twice as large as that produced by the use of a median VSL).

However, each calculation produces large estimated net benefits from the pollution-reduction policies in the European Commission's proposal, within the same order of magnitude, and at many multiples of the monetary costs of these policies. Each calculation delivers the same advice on the decision required with regard to the proposal: namely, an affirmative response.

When the level of ambition moves beyond the European Commission's proposal to the MTFR, the use of alternative metrics results in a difference in the sign of the net benefits (that is, negative in some cases, positive in others), and thereby also a difference in the advice that the analysis offers to decision-makers.

In Holland's research (Holland, 2014), using the most conservative metric (the one yielding the lowest estimate of benefits: median VOLY), the point at which marginal cost equals marginal benefit is at 76% of gap closure. Using the least conservative metric (the one yielding the highest estimate of benefits: mean VSL), marginal cost equals marginal benefit at 92% of gap closure. In other words, all four metrics yield the result that abatement is economically justified up to 76% of gap closure and not beyond 92%.

The area of disagreement is an important matter of debate between specialists, in terms of the precise choice of instrument. However, this should not obscure the wider area(s) of agreement: the cost of air pollution is large; the benefits available from reducing these costs are large; and ambitious pollution-reduction policies are economically justified, including the proposal from the European Commission currently under debate.

2.10 A comparison of economic (welfare) and GDP impact assessments

As defined at the outset, the focus of this chapter is the economic cost to society of air pollution: a term that is interchangeable in standard economic theory with social cost, welfare cost, or welfare loss. It is acknowledged that other standpoints are relevant, including, inter alia, calculations of the financial costs of: defined health impacts, the GDP impacts of air pollution, and measures to mitigate air pollution.

Moreover, while it is argued that these contributions need to be considered separately and that their results should not be conflated, it is also acknowledged that there is a case for bringing to the attention of decision-makers simultaneously the economic cost estimates of air pollution and the estimates of its impact on GDP: separate presentations, but side-by-side. This section does this, using the case of the United States EPA's 2011 evaluation of the US CAAA (EPA, 2011a; 2011b).

The focus of the evaluation is very much on the economic (welfare) analysis: its main purpose is to provide an analysis of the costs and benefits of the 1990 CAAA, "incremental to those costs and benefits achieved from implementing the original 1970 Clean Air Act and the 1977 amendments" (EPA, 2011a:Abstract). As with any such CBA, the economic *benefits* being estimated are the economic costs saved as a result of the intervention – the economic costs of the reduction in premature deaths, the reduction in morbidity, the reduction in other negative environmental impacts, and so on. The *costs* to which these benefits are compared are the costs of implementation; namely, the resources sacrificed as a result of complying with the provisions of the intervention.

The assessment presents results in several categories: costs, benefits, net benefits, benefit–cost (B/C) ratios and costs per premature mortality avoided, with a central estimate as well as high and low estimates, by given target years as well as cumulatively. Table 2.10 presents only the central estimate for the year 2020.

The estimated benefits of US$ 2 trillion comprise the estimated economic costs associated with the loss of lives and health (and other valued things) that air pollution would have imposed in the United States in 2020, but that will now be saved as a result of its mitigation by

Table 2.10. Estimated costs, benefits, net benefits and B/C ratios in the United States EPA's CAAA CBA, 2020

1990 CAAA: central estimate for 2020	US$ (millions) (2006)
Monetized direct costs	65 000
Monetized direct benefits	2 000 000
Net benefits	1 900 000
Benefits divided by cost (B/C ratio)	31:1
Costs per premature death avoided (2006, in US$)	280 000

Source: extracted from results given in Table 7-5 (in US$) by the United States EPA (EPA, 2011a).

means of the 1990 CAAA. It can therefore be compared to the US$ 1.575 trillion estimated in this study as the economic cost of the combined health impacts of air pollution in the WHO European Region in 2010 (see Table 2.5). Bearing in mind that the United States EPA study is relevant to the United States and the year 2020, with its considerably higher VSL values, and the fact that it includes a range of environmental impacts in addition to the impact on human health, the result is indeed comparable, notwithstanding the difference in population.

Given the argument and evidence presented to date, the significant benefits and high B/C ratios are not surprising. The costs of large-scale interventions to reduce air pollution, although they may appear sizeable at first glance, are small relative to the economic benefits gained, whenever these benefits are calculated in a manner consistent with economic first principles. In particular, the costs per premature death avoided, as given in the final row of Table 2.10, are small relative to VSLs.

The United States EPA (EPA, 2011a) also provides, by means of computable general equilibrium modelling, an estimation of the final impacts of the 1990 CAAA on GDP (as well as on consumption and the consumption–leisure trade-off). Here, the costs in question are the compliance *expenditures*, rather than the direct *costs* of compliance. Thus, whereas the latter excludes taxes, since they are a transfer from one party to another, all such expenditures need to be included in order to track the final impacts on GDP. In contrast, the benefits are limited to those included in GDP accounting. Thus, the lives of people that are not part of the labour force, and who are saved from premature death, are not counted.

The results are presented by the United States EPA (EPA, 2011a:Table 8-7, Table 8-8) in two case studies. The first is a cost-only case, which estimates the final impact on GDP of the expenditures related to the CAAA. The second is a labour force-adjusted case, which seeks to include as many of the benefits that can reasonably be included within GDP accounting: changes in labour force from reduced mortality, changes in labour force from reduced morbidity, and savings in medical expenditures. These changes permit an expansion of output in the rest of the economy, alongside a contraction in output in the health care sector itself. The results for the two cases are shown in Table 2.11.

Table 2.11. Estimated GDP impacts presented in the United States EPA's CAAA CBA, 2020

1990 CAAA: estimated GDP impacts for 2020	US$ (billions) (2006)
Cost-only case	
GDP with CAAA	20 202
GDP without CAAA	20 312
Change in GDP	–110
Labour force-adjusted case	
GDP with CAAA	20 202
GDP without CAAA	20 197
Change in GDP	5

Source: extracted from results given in Table 8-7 and Table 8-8 (in US$) by the United States EPA (EPA, 2011a).

Two conclusions follow from these results.

- Counting the GDP-relevant benefits of pollution reduction can yield a net positive impact on GDP for ambitious interventions designed to achieve significant reductions in air pollution: in the case of the 1990 CAAA, this is a net addition to the United States GDP of US$ 5 billion in the year 2020.
- The net positive impact on GDP is trivial when compared to the economic benefits of reduced pollution, as defined in standard economic theory and as embodied in the standard method for valuing the loss of life: in this case, this is a net economic benefit of US$ 1.9 trillion in the year 2020.

An objection could be raised that the relatively meagre GDP impacts constitute an argument for better accounting of these impacts and better accounting of GDP itself. Indeed, the United States EPA concludes with a discussion of these analytical limitations (EPA, 2011a). As noted earlier, research is currently being undertaken to overcome these limitations.

Nonetheless, it remains the case that the economic benefits from reduced pollution will dwarf the positive change in GDP, as long as:

- GDP continues to be designed as a measure of the output of society's economic actors – a measure that society surely needs to possess for many and various reasons – and not a measure of the welfare of all the individuals who constitute that society;
- economic costs and benefits continue to be defined as they are in standard economic theory; namely, as the value lost and the value gained by individuals, established by their individual valuations.

Box 2.1. Putting trillions into context

1 trillion = 1 million million = 1 000 000 000 000

Estimates in the trillions of dollars can be illustrated by making simple comparisons to other figures, as detailed below.

- Approximate 2013 GDP figures according to the World Bank (World Bank, 2015) are: Germany US$ 3.7 trillion, Russian Federation US$ 2.1 trillion, Spain US$ 1.4 trillion, and Israel US$ 0.3 trillion.
- The total global health spending in 2009 was US$ 5.1 trillion (Bloom et al., 2011).
- Globally, the cost of illness was estimated at US$ 2.5 trillion for mental illness and US$ 0.9 trillion for cardiovascular disease in 2010 (Bloom et al., 2011).
- The total amount of overseas development assistance delivered since the mid-1990s is less than US$ 2 trillion (Bloom et al., 2011).

3. Policy implications: towards an evidence-based approach

3.1 The need for action and the need for reflection

The statement of evidence on the problem of air pollution, presented earlier – globally, a premature death toll in the millions, with an economic cost in the trillions; and in the WHO European Region, a toll of about 663 000 premature deaths, approximately 13 million DALYs, and an estimated economic cost of about US$ 1.6 trillion – is, ipso facto, a compelling argument for action to mitigate the problem.

However, the case for action is not new. Governments and supranational authorities from across the world have received advice from their own agencies and from supranational agencies such as WHO, the OECD and the European Commission, and have acted on it to a greater or lesser extent. In the EU in particular, there is a vast body of evidence and recommended actions that have been documented in the course of the development of the EU's CAPP and in the processes related to it.[9]

Consider, for example, the quantification of the case for action in Holland's final CBA for CAPP (Holland, 2014), in moving from the CLE baseline scenario to the European Commission proposal (B7) for strengthened regulations on emission controls (Table 3.1).

Table 3.1. Estimated costs, benefits, net benefits and B/C ratios in the EU CAPP CBA, 2030

Commission proposal (B7): estimate for 2030	€ (millions)
Benefits over baseline (at mean VSL)	138 705
Net benefits (at mean VSL)	135 371
Benefits over baseline divided by cost (B/C ratio)	42:1

Source: extracted from Table 5.2 and Table 5.4 in Holland (2014).

Note, moreover, that the benefits displayed here are health benefits only (see Holland, 2014). The CAPP itself addresses a wider range of air pollution impacts, including damage to: agriculture and forestry, in the form of reduced crop

9 See, in particular, Amann (2014), Holland (2014), WHO Regional Office for Europe (2013a; 2013b), and the European Commission impact assessment (EC, 2013).

yields and fish stocks; landscape; the built environment, and so on (see EC, 2013). It therefore yields a wider range of pollution-reduction benefits. These are, as noted in the Introduction, matters that lie outside the remit of this study, but they are very much a part of the challenge posed by air pollution.

In any case, it is now clear that the toll imposed by air pollution is much more serious than was previously understood. Indeed, the very fact that such a remarkably high B/C ratio as that shown in Table 3.1 remains (42:1), requiring further action, is itself proof of the insufficiency of the interventions conducted to date. On the other hand, as documented in Chapter 1, it is also a fact that Europe has succeeded in reducing air pollution and its toll – not by nearly enough, but by more than has been achieved elsewhere. Europe has accomplished this more with regulatory instruments that have defined air quality standards[10] than with other instruments in the policy arsenal. There is therefore a need for better understanding of regulatory instruments and their place within the arsenal: a better understanding of how to maximize their effectiveness and when to complement them along with other instruments.

Mitigating the toll on life and health imposed by air pollution is a multifaceted task: indeed, it is arguably not a singular task at all, but rather the sum of a number of separate tasks. Potential health benefits in the form of enabling more people to lead longer lives and with better quality of life can be, and are, secured through several, separate and often unrelated channels. Moreover, some of these channels need not in fact involve any attempt to reduce air pollution itself. For example, health programmes that result in the early identification and treatment of patients suffering from heart disease could reduce the number of those who are at higher risk from air pollution, and in turn reduce the health toll of air pollution, without addressing its root causes.

In addition, there are several measures attempting to reduce exposure to air pollutants, rather than acting to reduce emissions. Consider, for example, recent research from the United States on the optimal design of cycle paths in relation to roads, or the optimal design of bus and tram stops in order to enable waiting users to face away from road traffic rather than toward it (Grabow et al., 2012; Figliozzi & Monsere, 2013).

Although these measures play a role in reducing the effects of air pollution, the largest benefits are to be expected from measures that reduce emissions. Important examples of this come from the estimate for the United States of about US$ 2 trillion in benefits gained by 2020 as a result of the 1990 CAAA, of which 95% are health benefits (see Table 2.2 and EPA, 2011a), and the estimate for the EU of about €140 billion in health benefits that would be available by 2030 if the European Commission's proposal on the CAPP were to be adopted (see Table 3.1, as well as Holland (2014) and the European Commission (EC, 2013)).

It follows that these large-scale, multi-sector, as-comprehensive-as-possible initiatives to reduce air pollution at source are indeed appropriate actions. It also follows that well-designed policy instruments possess per se important attributes that promote action, providing means to promote innovation, information exchange and monitoring. Moreover, and evidently, they also demonstrate that the world does indeed possess enough knowledge to attack air pollution at source and to reduce it and its toll on life and health: that is good news.

10 Examples include the European Air Quality Directive (European Parliament & Council of the European Union, 2008), which addressed the emissions of air pollutants; the Convention on Long-Range Transboundary Air Pollution; and a set of EU directives addressing emissions from large combustion plants, waste incineration plants, road vehicles and ships.

3.2 Sector-specific technical evidence and its limits

As reported in the preceding chapters, important recent evidence exists from epidemiology and economics perspectives on the resulting toll on health and its value, in the form of millions of premature deaths and DALYs globally, and the trillions of dollars. In order to inform effective policy-making, detailed, documented evidence is needed on the sectoral sources of air pollution's impacts and costs. In Europe, as elsewhere, much data are available on the sources of air pollution, including evidence on the sources of air pollutant emissions, and also, albeit less extensively, on their concentrations and on the exposure of populations thereto. In particular, the EEA continually monitors and regularly reports on trends for emissions from the main pollutants for each of the EU Member States and for the EU as a whole (see, for example, EEA, 2014).

The final report of the Clean Air for Europe (CAFE) Programme estimated the actual contribution of road transport to NO_x emissions for the year 2000 baseline to be 45% – and also its potential reduction to be 30% by 2020, with a mitigation policy scenario (see Amann et al., 2005:Fig. 4.8, Fig. 4.9). Subsequently, the EEA recorded that road transport's share of NO_x emissions had fallen to 33% by 2010 (see EEA, 2012:Fig. 4.1), while adding the warning that this overall reduction had been vitiated in part by "the increased proportion of NO_x emitted directly as NO_2 from the exhaust of more modern diesel vehicles" (see EEA, 2012:32). Rafaj, Amann & Siri (2014) later reported that road transport has disappointed expectations, owing to the higher than projected penetration of diesel.

Such research on the sectoral sources of pollutant emissions provides much useful evidence, but is not in itself an answer to the question of the sectoral sources of pollution's impacts and costs: how many premature deaths, DALYs and dollars can be attributed to the various sectors is not yet known, and a full estimation of the sectoral shares of air pollution impacts and costs across the WHO European Region cannot be presented here, as the required evidence base is not available.

However, the evidence that does exist indicates that, across Europe, commercial, institutional and household fuel combustion, transport, industrial emissions, energy production, agriculture, and waste and solvent product use are known to be the largest contributing factors (EEA, 2013b). Establishing the relative magnitude of these sources, and analysing available remedial policy strategies and options are key ingredients for improving air quality in Europe. The rest of this report is dedicated to exploring this matter further.

3.3 Estimating the main sectoral sources of air pollution impacts and costs

As argued by the OECD, with regard to ambient PM, the available literature suggests that road transport's share of the economic cost of premature deaths – when properly calculated – is likely to be about 50% across the EU, albeit not in each Member State (OECD, 2014).

An important turn-of-the-century study covering Austria, France and Switzerland estimated road traffic-generated air pollution to be responsible for 54% of the economic cost of air pollution's health impacts, including the cost of both mortality and morbidity, in the three countries taken together (Sommer et al., 2000). The technical evidence on relevant developments since then – including (a) the reduction in transport-sector emissions as documented by the EEA

(EEA, 2013a) (noted in Chapter 1), (b) the partial reversal of this downward through dieselization, and (c) the general growth in the transport sector – supports the plausibility of an estimate of about 50% for the EU today.

As far as their contribution to air pollution is concerned, diesel vehicles are more harmful than petrol vehicles. In contrast to petrol vehicles, diesel vehicles have not shown significant reduction in NO_x emissions since the 1990s. Exhaust emissions from such vehicles are lower for carbon monoxide, non-methane volatile organic compounds and PM, but may be substantially higher for NO_x. The fraction of NO_x emitted as NO_2 by diesel vehicles is high – at around 25–30%, as opposed to a few percent for petrol vehicles – and has shown a variable rather than downward trend over the years (Carslaw & Rhys-Tyler, 2013; Carslaw et al., 2011). The decrease in NO_x emissions (30% between 2003 and 2012) is greater than the fall in annual mean NO_2 concentrations (approximately 18%). This is attributed primarily to the increase in NO_2 emitted directly into the air from diesel vehicles, plus the increasing numbers of newer diesel vehicles. However, owing to tax incentives favouring diesel over petrol, the recent past has witnessed a continuing shift from petrol to diesel vehicles. Holland (2014) counts this missing consideration as one of the most important limitations of the CAPP CBA.

The most recent studies that offer estimates broken down by sector also support an estimate of about 50% responsibility (directly or indirectly) for the damage inflicted by the road transport sector in EU countries. A 2013 study from the Massachusetts Institute of Technology (MIT) (Caiazzo et al., 2013) calculates sector shares of premature deaths from air pollution across the United States (see also Chu (2013) and Dedoussi (2014) for the further development of this line of research). The breakdown is not the same as for Europe, but this is unsurprising, given the long-established higher share attributed to power generation in the United States relative to the EU, and the correspondingly lower share for road transport and other sectors.[11] An earlier MIT study for the United Kingdom also supports the validity of the estimate of a share of about 50% responsibility for the road transport sector in EU countries. The study provides a range of estimates, both nominal and corrected, of premature deaths per year in the United Kingdom by fuel combustion sector, along with some estimates for London in particular (shown in Table 3.2) (Yim & Barrett, 2012).

In the discussion of these estimates, Yim and Barrett (2012) make two further caveats of relevance to this study. The first relates to modelling technology: note should be taken that that the road transport estimate in particular is likely to be an underestimate, as the peaks in roadside

Table 3.2. Share of combustion sectors as a percentage of combustion emissions-related premature deaths in the United Kingdom and in London, 2005

Sector	United Kingdom ≈ share of premature deaths (%)	London ≈ share of premature deaths (%)
Road transport	40	50
Other transport	20	20
Power generation	20	15
Other sectors	20	15
All sources	100	100

Sources: extracted from Table 1 (and related discussion) presented by Yim & Barrett (2012).

11 Further discussion exists on the differences between the two sides of the Atlantic, presented by Caiazzo et al. (2013).

$PM_{2.5}$ may not be accurately represented owing to the model resolution. The second relates to toxicology: it should be taken into account that potentially significant unquantified uncertainty is the differential toxicity among PM species, and the outcome of this is that the health impact of road transport is likely to be further underestimated.

However, it is difficult to judge whether the share of approximately 50% attributed to road transport in EU Member States might also apply to the rest of WHO European Region. It is perhaps more likely that a full study encompassing the countries to the east of the EU would establish a contraction for the share contributed by road transport and a corresponding expansion of the share attributable to the sum of all others sectors identified in Table 3.2: namely, other transport, power generation and other sectors.

With regard to HAP, by definition, household fuel combustion can be assumed to account for 100% of the impact. As far as its share in the totality of premature deaths from APMP and HAP is concerned, three points are worth noting.

1. From the evidence presented in the GBD-2010 Study (IHME, 2014), as recorded in Chapter 1 (see in particular Table 1.3), HAP accounted for about 25% of the sum of the premature deaths from APMP and HAP in the WHO European Region as a whole in 2010 (164 231 out of 662 769). It accounted for around 50% of the sum, and sometimes more than 50%, in a number of WHO European Region Member States.
2. These figures likely underestimate the real share of premature deaths from APMP and HAP, partly because of less complete data collection in the areas in which HAP is predominant and partly because a default value of zero is assigned for 24 high-incomes countries for which no data on premature deaths from HAP were recorded.
3. There is some evidence to suggest that HAP has been increasing since the onset of the economic downturn in 2009 (also noted by the European Commission (EC, 2013)).

Proceeding with the available evidence-based estimate of a 25% share attributable to HAP in the sum of premature deaths from APMP and HAP in the WHO European Region and, therewith, a 25% share attributable to household fuel combustion as a sectoral source of this sum, the shares of premature deaths from air pollution attributable to the main responsible sectors (as recorded in Table 1.3), and the economic cost thereof (as recorded in Table 2.4), are shown here in Table 3.3 and Table 3.4.

Table 3.3. Indicative estimates of sector shares in premature deaths from air pollution in the WHO European Region, 2010

Sector	(≈) Share of premature deaths from APMP (%)	(≈) Share of premature deaths from HAP (%)	(≈) Share of premature deaths from APMP + HAP (%)
Household fuel combustion	0	100	25
Road transport	50	0	37.5
The remainder (other transport power generation, and other sectors)	20	15	
All sources	100	100	100

Sources: extracted from text and tables presented earlier in this report.

Table 3.4. Indicative estimates of sector shares in the economic cost of premature deaths from air pollution in the WHO European Region, 2010

Sector	(≈) Share of economic cost of premature deaths from APMP (%)	(≈) Share of economic cost of premature deaths from HAP (%)	(≈) Share of economic cost of premature deaths from APMP + HAP (%)
Household fuel combustion	0	100	20
Road transport	50	0	40
The remainder (other transport power generation, and other sectors)	50	0	40
All sources	100	100	100

Sources: extracted from text and tables presented earlier in this report.

These findings suggest that in the WHO European Region road transport and household fuel combustion combined account for the majority of the impacts and costs. No other single source, such as industry, power generation, agriculture, transport (other than motorized road transport) contributes as much. It is therefore a matter for concern that there is evidence to suggest instances of recent and continuing regress rather than progress in precisely these two sectors; in one case due to increasing diesel penetration and other due to the effects of the economic downturn in relatively poorer regions (see EC, 2013). Serious reflection on the policy response to air pollution, and on how best to sharpen that response, should pay particular attention to these two sectors.

3.4 The logical framework for correction: "pricing + investment + regulation"

A striking feature of the record of pollution abatement policies over the last several decades – whether in Europe, the United States, or elsewhere – has been its predominant reliance on regulatory interventions to impose new limits and standards across multiple sectors of the economy (see the European Commission (EC, 2013) and the United States EPA (EPA, 2011a) for summary reviews of this record).

Yet, as a general rule, economists tend to prioritize the policy of "getting prices right" in addressing the problem of externalities. In reflecting on past and future policy, this divergence merits attention. Economics holds that the general welfare is maximized when the price paid for consuming a product – each good or service, including each trip by road, rail, sea and air – is equal to the additional cost it imposes on all. However, as a result of various market imperfections, market prices can deviate sharply above or below this point; and when this happens, the gain to the winner – the producer in the former case, or the consumer in the latter – is less than the loss to the rest of society. The general welfare is thus reduced.[12]

For example, a consumer may gain by purchasing and driving a cheaper but more polluting car, but the loss to society can far outweigh that gain. Therefore, in the presence of externalities such as pollution, economists have generally

12 The argument is spelt out and fully referenced in Roy (2008), from which this brief statement here borrows freely.

recommended a tax to raise the marginal price of the product for the user up to the marginal cost imposed by its use (or equivalent monetary incentives) and thereby “price out” welfare-reducing consumption that would otherwise be “priced in”. In moving thus from the original price to the corrected price, there is a net gain to society.

Within this framework, pricing and investment have vital, complementary roles to play. The question of pricing logically comes prior to the question of investment, as the wrong answer to the former will generate wrong answers to the latter, and quite possibly at a significant cost to the welfare of future generations. The schedule of demand that follows from the wrong set of prices is not the same as the schedule of demand that would follow from a corrected set of prices. A correction to prices is likely to alter the composition, location, scale and timing of the investment required to meet future demand.

Regulation, too, has a vital, and also complementary role to play. Economists generally recognize that there is a strong case for both initiating and maintaining regulations that pass a cost–benefit test; for example, higher mandatory standards for new vehicles. However, economists also recommend a correction of prices, for example via differentiated taxes on more polluting vehicles and fuels in order to shift consumption patterns to cleaner vehicles and fuels, until the point at which polluting cars and vehicles are retired entirely from use. At this limiting point, regulatory controls assume the role of a watching brief over a corrected market, for example to ensure that manufacturers do not reintroduce the production of polluting vehicles.

This framework, however, has important limitations, as exemplified by the CBA for the CAPP (described earlier), with a B/C ratio of 42:1. Such a disproportionate ratio is itself evidence that something has gone wrong with the system of price signals faced by consumers and producers, the schedule of demand and supply following from it, and the investment, production and sales decisions made in response to this schedule. It is only at the end of this sequence, and in consequence of it, that the outcomes can be bad enough to offer the possibility of good returns from further corrective action. Such a possibility should therefore direct the attention of policy-makers to the beginning of the sequence.

Thus described, the policy framework for correcting externalities recommended by economics can be summarized using the formula: pricing + investment + regulation. Today, however, the evidence gained from the story of air pollution and its mitigation suggests that there is a case for rethinking this framework – not so much its logical sequence as its chronology, and not in order to abandon it but to understand it anew and to apply it more effectively.

3.5 The evidence on the efficacy of pricing, investment and regulation

Much evidence exists to show that pricing works effectively as a policy instrument, when correctly devised and applied. Sector-specific relevant examples include the introduction of the congestion charging scheme in central London, which succeeded in sharply reducing the number of kilometres driven by vehicles and thereby reducing not only congestion but also fuel consumption and air pollution (Roy, 2014). Policies also include reductions in company-car subsidies in a few countries that have reduced the number of kilometres driven and thereby reduced the several and various attendant externalities (Le Vine & Jones, 2012).

At the same time, the evidence suggests that such examples of corrections to prices have been too limited and too localized to correct distorted prices in the transport sector; that is, the under-pricing of road use relative to the costs it imposes and relative to its potential substitutes. An example is given by the OECD; it predates recent evidence on air pollution and therefore understates the extent of the distortion (OECD, 2014:76):

> *A benchmark study for the UK Department for Transport in 2001 (Sansom et al., 2001) found that, in 1998, the ratio of revenues to marginal social costs in road transport was in the range of 0.36 to 0.50 – that is to say, road users were, on average, paying one third to one-half of the costs that their trips imposed on society. Passenger and freight rail users were paying, respectively, just below and just above marginal costs. Updating for 2004 in 2007 (United Kingdom Department for Transport (DfT), 2007), the Department found that the ratio of revenues to marginal social costs for passenger cars was 0.15. As before, the ratio for buses and commuter rail, the most comparable segment of passenger rail, was close to unity.*

One of the adjustments to prices in this sector, intending to respond to the problem of climate change, was the tax differential in favour of diesel, which actually worsened the problem of air pollution (Upton, 2013; Harding, 2014), as it increased the diesel-vehicle share of the total vehicle market. The European Commission's review of current policy also emphasizes this point, stating that "[T]he consequences of the less than hoped for effects of the vehicle standards relating to diesel passenger cars and light-duty vehicles have been exacerbated by national taxation policies favouring diesels" (EC, 2013:25).

Moreover, another misconceived response to the problem of climate change may have served to distort prices in such a way as to worsen the problem of air pollution. With regard to household fuel combustion, the European Commission's review notes: "[T]he problem is not only continuing coal use, but also increase in biomass use, driven partly by renewables policy" (EC, 2013:32).

As is clear from Table 3.3 above, for the WHO European Region household fuel combustion ranks (along with road transport) as one of the two main sectoral sources of the health impact of air pollution. It is also clear, therefore, that the role of any price distortions acting to encourage the use of more polluting forms of household energy is an issue that is in need of additional focused research.

The evidence on investment is more encouraging. Fortunately, it is not the case that public and private investors have continued uninterruptedly to invest to meet the demand resulting from distorted prices. This is partly because governments – recognizing that current market prices have not been corrected – have sometimes explicitly used shadow prices in their ex ante evaluations, such as estimates for the costs of externalities that have not been corrected in the market price. This is partly because both public and private investors have sometimes assumed that the near future would bring a correction and have made their investment decisions in anticipation of it.

Finally, regulation has succeeded in reducing air pollution, especially in Europe. As evidenced in Chapter 1 (Table 1.2), the 53 Member States of the WHO European Region (taken together) did achieve a greater reduction in premature deaths from air pollution, and a greater rate of reduction, than that recorded for the 34 OECD countries (taken together) for the period under study. As noted earlier, and detailed by the European Commission (EC, 2013) and elsewhere, pollution abatement policies have relied principally on regulation.

This has not been flawless, however. The policy failure with regard to diesel has been a failure of regulation as a well as a failure in terms of pricing, since the compliance regime has failed to prevent a significance divergence between the test cycle performance, which secures regulatory approval, and the actual on-road performance of the approved vehicles. As the European Commission describes it (EC, 2013:24):

> *[T]he problem is due in part to the poor*

representativeness of the standardised test cycle used for type approval in the EU … and the weakness of in-service conformity testing. Under the current regime an engine type has to meet the type-approval requirements when tested according to the test cycle, but under normal driving conditions the real emissions can be much higher.

The result is that "while the NO_x emission limit values for diesel passenger cars have been tightened by approximately a factor of 4 from 1993 to 2009 (Euro 1 to Euro 5), the estimated average NO_x emissions in real driving conditions have slightly increased" (EC, 2013:24). Clearly, this is mainly a matter of regulation and not of pricing; however, the tax differential in favour of diesel has added to the problem, leading consumers to shift from petrol to diesel vehicles.

In any case, the record of regulation in this field – including both its successes and failures – is positive. Thus, it seems reasonable to conclude that the instances of failure are arguments for refining and strengthening the scope for stringent regulation, rather than arguments for weakening it.

3.6 The chronological framework for correction: "regulation + investment + pricing"

The evidence on the actual record of pollution abatement policy and the various instruments used to date does not falsify the theoretical proposition that the general welfare *is* maximized when prices are equal to marginal costs, nor the recommendation that they *should* be equalized. Nor does this evidence undermine the priority accorded to pricing in the logical sequence described earlier: it remains true that both investments and regulations are necessarily obliged to act on a schedule of demand and supply that is itself shaped by the schedule of prices, whether it is right or wrong.

However, the chronological sequence is another matter. Suppose the evidence noted above to be not simply a case of human error but the result of intractable time lags in developing and implementing tax reform, and/or information gaps which also impose critical time lags, and further suppose that the problem does not apply in the same way or to the same degree in the case of regulation.

For example, mandatory vehicle standards have been in place for long enough to have become a fact of life: the relevant actors expect them to be maintained and tightened over time (even if some attempt elaborate ways of evading the effects of the tighter limits). However, national road pricing, for example, is not in place: it is not a matter of adjusting the actual prices in the light of new information, but of bringing the scheme into effect against the force of inertia.

Moreover, the information requirements are not the same: witness the incompleteness of the evidence base on the sector sources of air pollution. The information gaps here are not necessarily critical to the design of regulation (even if they impact on the precision of the CBAs). For example, setting and enforcing a more rigorous compliance regime and with tighter standards for new diesel vehicles does not need to wait for the determination of the precise contribution of diesel vehicles to marginal costs: it is sufficient to know that standards need to be tightened further. Setting and implementing a full schedule of corrected prices through purchase taxes, fuel taxes and road-user charges, however, does require a greater degree of precision in knowledge. In short, if Europe is to act on the problem immediately and continuously, rather than at a designated date in the future, it needs to begin at the place in which it finds itself today, rather than waiting for future information.

A relatively successful, albeit imperfect regulatory regime on air quality in Europe, and a relatively good knowledge of its own points of weakness have resulted in substantial progress in terms of health impacts and costs, even in the absence of a price system capable of taking full account of externalities. In view of the persistence of the problem of air pollution in Europe, however, correcting distortions in taxes and subsidies remains highly desirable.

To pursue this goal, operating in the anticipated period of time until a full correction of prices can be achieved, there is a case for conceiving the chronological framework of correction following the approach: regulation + investment + pricing, as set out in the list below.

- First, maintain and strengthen existing and planned regulatory controls on air pollution and the associated compliance regimes, refining these as required.
- Next, develop and extend guidance on using shadow prices in relevant investment decisions – a procedure which does not require the same level of precision as the setting of actual prices.
- Then, over a period of years, close the information gaps required to prepare a model of fully corrected prices, and pursue full implementation.
- Throughout this period, adjust the uncorrected prices as the requisite information becomes available – beginning immediately, with the undoing of those changes that are known to have acted in the wrong direction, such as any tax preferences given to diesel over petrol.

Recalling the evidence on the sectoral sources of air pollution's impacts and costs in the WHO European Region, the chronological framework suggested above can provide some practical guidance on where and how the policy response to the problem could be strengthened.

Assuming implementation of the measures proposed in the EU impact assessment policy, with the geographical scope covering the whole of the WHO European Region rather than the EU alone (but with the technical scope restricted to impacts on public health rather than the full range of environmental impacts), the implications for additional policy action that follow from the analysis are shown in Table 3.5.

Table 3.5. Implications for additional policy action regarding the main sectoral sources of air pollution in the WHO European Region, 2010

Sector	Additional policy action
Household fuel combustion	CAPP + • focused research effort and, as necessary: • removal of subsidies for polluting fuels • appropriate guidance on investment in substitutes • a longer term correction to relevant prices
Road transport	CAPP + • a more rigorous compliance regime • removal of tax advantage for diesel • appropriate guidance on investment in substitutes • a longer term correction to relevant prices
The remainder (other transport power generation, and other sectors)	CAPP + • continued research and monitoring and, as necessary: • new regulatory/investment/pricing options

Sources: extracted from text and tables presented earlier in this report.

Nothing in the chronological framework proposed in this section relieves the urgency of the fact that prices need to be fully corrected in order to maximize the general welfare. The case for pricing reform, hitherto pursued with limited success, must continue to be pursued. Information gaps will need to be closed and new models will need to be developed. Interdisciplinary work and cross-sectoral action are needed to allow evidence and expertise to support the existing process of reform in terms of regulation and investment decision-making.

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