The Cost of Air Pollution

HEALTH IMPACTS OF ROAD TRANSPORT

OECD

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Foreword

Local air pollution, and the health problems it causes, have received increased attention in many parts of the world, often because of specific incidents in major cities. However, over the last few years, the evidence-base has improved significantly, and now demonstrates that the health impacts of local air pollution, particularly from road transport, are much larger than previously thought. Drawing on this improved evidence-base this study estimates the economic cost of the health impacts of air pollution from road transport – on a global scale, but with special reference to People's Republic of China, India and the OECD member countries.

After the preparation of this book was finished, the World Health Organization published new information showing that 3.7 million people died globally because of outdoor air pollution in 2012; a further increase from the 3.4 million mortalities in 2010 that this book is based on.

The book was prepared by Dr Rana Roy, who in turn wishes to acknowledge the able research assistance provided by Mr Stuart Baird. Jenny Calder of the OECD Secretariat contributed to the preparation of the final manuscript, and Nils Axel Braathen of the OECD Secretariat oversaw the implementation of the project.

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List of abbreviations

AQG	Air Quality Guidelines
B/C	Benefit-cost ratio
CAFÉ	Clean Air for Europe Programme
CBA	Cost-benefit analysis
COI	Cost-of-illness
CPI	Consumer price index
CSE	Centre for Science and Environment
DALY	Disability-adjusted life years lost
DfT	UK Department for Transport
DKK	Danish krona
EEA	European Environment Agency
ESCAPE	European Study of Cohorts for Air Pollution Effects
EU	European Union
EUR	Euro
EVs	Electric vehicles
GBD	Global Burden of Disease
GDP	Gross domestic product
HEAT	Health economic assessment tool
HEI	Health Effects Institute
IARC	International Agency for Research on Cancer
IIASA	International Institute for Applied Systems Analysis
LYL	Life years lost
MIT	Massachusetts Institute of Technology
MTFR	Maximum technical feasible reduction
NO_x	Nitrogen oxides
NPV	Net present value
PM	Particulate matter
PPP	Purchasing power parity
PV_b	Present value of benefits
PV_c	Present value of costs
QALY	Quality-adjusted life years lost
TSAP	Thematic Strategy on Air Pollution
UK	United Kingdom
USD	United States dollar

US EPA	United States Environmental Protection Agency
VOLY	Value of a life year lost
VSL	Value of a statistical life
VSLY	Value of a statistical life year
WHO	World Health Organization
WTP	Willingness to pay
YLL	Years of life lost

Executive summary

Outdoor air pollution kills more than three million people across the world every year, and causes health problems from asthma to heart disease for many more. This is costing OECD societies plus People's Republic of China and India an estimated USD 3.5 trillion dollars a year in terms of the value of lives lost and ill health, and the trend is rising. But how much of the cost of those deaths and health problems is due to pollution from cars, trucks and motorcycles on our roads? Initial evidence suggests that in OECD countries, road transport is likely responsible for about half the USD 1.7 trillion total.

Air pollution in OECD countries has fallen in recent years, helped by tighter emission controls on vehicles, but has increased in China and India as rapid growth in traffic has outpaced the adoption of tighter emission limits. The switch to more polluting diesel vehicles in many countries in part to combat climate change has also added to pollution effects, threatening to arrest the downward trend in emissions from road transport in OECD countries.

Over the five-year period from 2005 to 2010, there was an overall increase of about 4% in the number of premature deaths globally caused by outdoor air pollution – with an improvement in the OECD world being offset by a larger deterioration in the rest of the world.

These figures, based on new technologies for measuring pollution and improved analysis of health data, are far higher than those from previous studies of premature death and illness from air pollution. Calculating the economic cost of these health impacts, and how much is due to air pollution from road transport, requires estimating the value of lost lives or lost quality of life in the case of illness. There is a standard method for calculating the cost of lost life, but not for loss of health. Hence this study adds to the mortality cost a 10% margin for loss of health (morbidity), based on the best available evidence in recent studies.

It is now possible to give a better calculation of the health impacts of air pollution and of the associated economic cost. Available evidence and methodology suggest that about 50% of that cost in OECD countries is specifically attributable to road transport, although more work needs to be done to provide a robust calculation for the road transport share.

Main findings

- The number of deaths due to outdoor air pollution fell by about 4% in OECD countries between 2005 and 2010, while the number of years of life lost fell even further. But while 20 of the 34 OECD countries achieved progress, 14 did not.
- The number of deaths due to outdoor air pollution in China rose by about 5%, although years of life lost increased by only about 0.5%. China has arguably succeeded in slowing the increase in the effect of air pollution on health, since a reduction in exposure to pollution will have a greater effect on years of life lost than on the number of deaths.
- India registered an increase of about 12% in the number of deaths and about 3% in years of life lost. Although the number of deaths in India is only just over half the number in China, the trend in India is increasing faster.
- The cost of the health impact of outdoor air pollution in OECD countries, both deaths and illness, was about USD 1.7 trillion in 2010. Available evidence suggests that road transport accounts for about 50% of this cost, or close to USD 1 trillion.
- The best available estimate of the economic cost of the health impacts of outdoor air pollution in China and India combined is larger than the OECD total – about USD 1.4 trillion in China and about USD 0.5 trillion in India in 2010. There is insufficient evidence to estimate the share of road transport in these figures but even if it is less than half, it nonetheless represents a large burden.

Main recommendations

- A defensible calculation of the *economic cost* of health impacts must be based on economic first principles. This means *continuing* the use of the standard method for calculating the cost of mortality – the Value of Statistical Life (VSL) as derived from individuals' valuation of their willingness to pay to reduce the risk of dying.
- Indicative estimates suggest that morbidity would add 10% to the mortality cost figures, but work is needed to complete a standard method of calculating morbidity costs in a manner consistent with the standard method for calculating mortality costs.
- A defensible calculation of the economic cost of the *health impacts of air pollution* must base itself on the new body of epidemiological evidence made possible by recent innovations in monitoring and modelling technology.

- A defensible calculation of the economic cost of the health impacts of air pollution *from road transport* must base itself on sector-specific evidence consistent with the new body of epidemiological evidence. This necessitates a renewal of the sector-specific evidence base. In the meanwhile, it is possible to provide indicative estimates only of road transport's share in the overall cost.
- Governments should maintain strong regulatory regimes, particularly strict vehicle standards. Given the size of the economic cost of the health effects of air pollution, the benefits of reducing that burden could easily outweigh the monetary cost of investments in more ambitious programmes to reduce pollution.
- Governments should also rethink their approach to appraising policy moves, such as the regulatory and tax settings that facilitated the shift to diesel vehicles. Importantly, there is also a need to ask how it is that the appraisal process has hitherto failed to secure the passage of a range of policy proposals for example in relation to public transport that could have reduced air pollution – and how to rectify this in future.

Chapter 1

Defining the economic cost of health impacts

This chapter begins with a restatement of the economic first principles informing the "valuation" of life and health and, therewith, the "cost" of mortalities and morbidities. It shows that a standard method is available by which to measure the cost of mortality – the "value of statistical life" (VSL). While there is work to be done in order to establish standard measurement methods regarding morbidity, it is possible to proceed with an indicative estimate of the additional cost imposed by morbidities drawn from the best available evidence.

This study reports on the economic cost of the health impacts of air pollution from road transport – on a global scale but with special reference to People's Republic of China (hereafter "China"), India and the OECD world.

Any report on the "economic cost" of impacts on human health, be it from air pollution or any other source, involving as it does a "valuation" of life and of health, needs to explain as clearly as possible what precisely is meant by the terms "value" and "cost". This is a non-trivial task. For the use of these terms is frequently misunderstood.

The world is not yet free of the illusion that the wealth of the world subsists in gold (or some other form of money): the "chrysohedonistic illusion". Even though an explicit rejection of this view characterises the founding works of economic science in the mid-eighteenth century following through to today,[1] long after gold has given way to paper money, it is all too frequently supposed that what economists really mean by "value", or by "cost", is a given sum of money.

It is therefore as well to begin by stating that this is not so: money is not the thing being measured but the instrument with which we measure it. Of course, money plays several roles wherever it is present; and rival schools of economic thought hold rival views on the roles that it plays. In the context of the present analysis, however, and irrespective of these otherwise rival views, all economists can agree that money serves here merely as a common unit of account, an imperfect instrument with which to measure certain non-monetary phenomena: namely, the several various items that all of us as individuals "value" in the ordinary sense of the word.[2]

So, what is it that we as individuals value and that economists as observers seek to measure? They include:

- consumption – and, with it, 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 – and the sacrifice of some leisure in the act of labour in order to secure consumption
- health – and the sacrifice of some part of consumption in order to secure health
- life – and the sacrifice of some part of consumption in order to preserve it.

"Value" as used here – also called "utility" – is simply a measure of these items that we all value in the ordinary sense of the word; and "cost" is a measure of their loss, absolutely or as a means of securing other valuable items. 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*.

1.1. Mortality: The value of statistical life

In the case of the ultimate impact on health – mortality – economics today possesses a singular, and singularly elegant, *standard method* by which to measure the cost of this impact from a given source: that is to say, to measure the loss of the valued item – life – at the level of society as a whole. This is the "value of statistical life" (VSL), as derived from aggregating individuals' willingness to pay (WTP) to secure a marginal reduction in the risk of premature death.

OECD (2012) describes the basic process of deriving a VSL value from a WTP survey:

> The survey finds an average WTP of USD 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 individual is willing to pay USD 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 USD 30 over 100 000 people gives the VSL value – USD 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 (OECD, 2012).

As such, the economic cost of the impact being studied becomes 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 lives saved.

In addition, following an extensive research effort led by the OECD (OECD, 2012; Biausque, 2010; Braathen, 2012; Hunt and Ferguson, 2010; Hunt, 2011), including a rigorous meta-analysis of VSL studies (OECD, 2012), starting with 1 095 values from 92 published studies, both researchers and policy makers now possess a set of OECD-recommended values for average adult VSL. In units of 2005 USD, the recommended range for OECD countries is USD 1.5 million – 4.5 million, the recommended base value is USD 3 million.

The remit of this study is to apply these VSL values to the problem at hand: the problem of the health impacts of air pollution from road transport. There is, however, a need to pause to add a few words on the meaning and purpose of the standard method. For this in turn sets sharp limits to what can and cannot be done in this report. In particular, it shows up the folly, not to say absurdity, of attempting to *combine* the standard method with alternative methods of calculating the "costs" of mortality that have an entirely different meaning and purpose.

The reasoning informing the standard method is simple enough and may be simplified even further for the purpose of presentation as follows (Biausque, 2010; OECD, 2012). 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 the marginal rate of substitution between these two valued items, consumption and the reduction in the risk of dying, such that:

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

For the present, the two main points to note are these. First, the value that the standard method seeks to capture is the value (in this case, the value of the reduction in the risk of dying) *to the individual*; it is not, for example, the value of postponed revenue to the undertaker or the value of higher pension expenditure by the government. And second, the task of the economist is one of aggregating valuations *by individuals* at their marginal rates of substitution; it is not one of imposing valuations from above.

It is worth recalling here the words of Jacques Drèze, the originator of the standard method, in reflecting on its origins in an interview more than forty years later:

> 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 (Dehes, Drèze, and Licandro, 2005).

It follows that alternative methods of calculating the "cost" of mortality which seek neither to capture the value to the individual nor to register and aggregate the valuations by individuals cannot substitute for the standard method; nor can they be simply combined with the standard method to produce composite estimates.

This is not to deny that these alternative methods can offer interesting policy-relevant information. But that information needs to be treated separately from the information yielded by the standard method. To do otherwise is almost a category error.

For example, an incidence of pollution that results in the premature deaths of working-age people has an impact on the national accounts through the loss of output and wages; those responsible for studying and forecasting gross domestic product (GDP) changes have an interest in measuring this impact. Clearly, however, a calculation that stops counting at retirement age and places a zero value on the death of a person of 65 years is not counting the same thing as the standard method. It should not occasion surprise that this national-accounts' measure of the "cost" of mortality frequently produces very different estimates to those produced by the standard method.[3]

Similarly, the attempt to derive "WTP values" and "VSL values" from "revealed preference" rather than "stated preference" – for example, by reference to wage levels in dangerous jobs – can reveal interesting information on the degree of bargaining power, or the lack thereof, possessed by particular segments of the workforce.[4] What they do not reveal is what is registered by the standard method: the valuation by individuals of their WTP to reduce the risk of death.

As shown below, these issues of compatibility also have a bearing on the valuation of morbidity. But so far as concerns the valuation of mortality, the conclusion drawn here is simple. The standard method, safely grounded as it is in the first principles of economic science, will suffice for the task at hand; the rest can be set aside.

1.2. Morbidity: In search of a standard method

Economics today does *not* possess a singular, let alone singularly elegant, standard method by which to measure the cost of morbidity from a given source: that is, to measure the loss of the valued item, health. Nor do researchers and policy makers possess anything like a set of OECD-recommended values for the several and various morbidities that can arise from a given source.[5]

In part, this lack reflects the current state of research and its limitations. As noted below, there are two lines of research in this field. There is a reasonably well-established tradition of developing a plural rather than singular method of calculating the various costs of morbidities – but this has not yet arrived at a clear consensus on exactly what needs to be calculated or the values at which they are to be calculated. There is also a more recent line of research which seeks to arrive at a composite cost estimate – but this is nowhere near a state of maturity sufficient to generate either a consensus on method or a set of agreed values across the OECD world.

This lack also reflects a material difference in the subject matter of the two fields. There is a material difference between the "cost of mortality" and the "costs of morbidity" – or rather, several material differences. For the latter item is, in reality, *plural* in several respects.

Whereas mortality is, in the nature of things, a singular and well-defined endpoint, morbidities entail a *plurality of endpoints* – indeed, a very large range of endpoints, varying greatly in the extent of severity, and complicating enormously the task of eliciting and aggregating individual WTP values.

In addition, whereas the cost of mortality is, in an immediate and unconditional sense, borne by the individual who dies, a case of morbidity can entail the imposition of costs on *a plurality of agents* – to begin with, the individual who is suffering ill-health and the many who are involved in the organisation and execution of formal and informal care of the one who is ill.

Finally, the individual who is suffering ill-health suffers *a plural loss of utility*: 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 insofar as morbidity imposes a loss in utility on a plurality of agents as well as a plural loss of utility on the one who is ill – and without departing in the least from the distinction between economic calculation and other forms of calculation, such as national accounting that is so critical to a correct understanding of VSLs – it is entirely legitimate to calculate the costs of morbidity in a plural manner: as the sum of separate elements of cost.

In a more or less recent paper for the OECD, Hunt and Ferguson (2010) set out the elements of this sum:

> The economic costs of the health impacts of air pollution can then be given by the sum of three different categories:
>
> 1. *Resource costs*: 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 with visiting a doctor, ambulance, buying medicines

and other treatments, plus any related non-medical cost, such as the cost of childcare and housekeeping due to the impossibility of the affected person in doing so;

2. *Opportunity costs*: Associated with the indirect costs related to loss of productivity and/or leisure time due to the health impact;
3. *Disutility costs*: Refer to the pain, suffering, discomfort and anxiety linked to the illness.

It should be noted that the "loss of productivity" referenced above, and regardless of exactly how it is estimated, should be read here as the loss of income and hence consumption for the affected person and the affected person's household – as distinct from the loss of valued-added in the employer's accounts or in the national accounts. In this manner, each of these elements as well as their sum can be defined in conformity with the economic first principles set out in this chapter.

Unfortunately, this line of research has not yet had time to establish itself as a standard method, with a high degree of agreement on the definition of the elements to be calculated and the values at which they are to be calculated. There are several issues that need to be resolved, including but not restricted to the following (Hunt and Ferguson, 2010; and Hunt, 2011):

- the definition of distinct endpoints – without which WTP values make little sense since the disutility of the pain and suffering involved in "illness" can range from trivially low to very high;
- the need for consistency between methods for estimating the different cost elements;
- the obvious need to avoid double-counting;
- but also, and just as importantly, the need to be comprehensive – in particular, the need to include WTP values for disutility, rather than restrict the definition of costs to "resource costs" and "opportunity costs" alone, and to include both lost income and lost leisure in opportunity costs rather than restrict the definition of opportunity costs to lost income alone.

Nonetheless, this is a line of research that is safely grounded in economic first principles and should in the fullness of time be able to deliver the goods: that is, a standard method to calculate the costs of morbidity.

What is more unfortunate is that the search for a standard method has taken a turn in quite another direction, one which might never arrive at a destination that is capable of winning general agreement. This is the attempt to arrive *pari pasu* at a *composite* cost estimate of morbidity *and mortality*.

The reasoning informing this approach is as follows. The epidemiological literature can and does estimate mortality not only in terms of the number of premature deaths but also in terms of the years of life lost (YLLs) or life years lost (LYLs): that is, adjusting for the age profile and also the pre-existing condition of those impacted by mortality. The same literature can, and sometimes does, estimate morbidity not only in terms of its multiple endpoints but also in terms of "quality-adjusted life years lost" (QALYs) – or, alternatively described, "disability-adjusted life years lost" (DALYs). Given this, if economists could arrive at a "value of a life year lost" (VOLYs) (sometimes described as "value of a statistical life year" – VSLY), they could derive values for QALYs as a co-efficient of VOLYs – and therefore determine a measure of the "economic cost" of morbidity as a co-efficient of the "economic cost" of mortality. Once this task is achieved, policy makers could be relieved of the burden of applying VSLs derived from WTP surveys as a measure of the economic cost of mortality.

Now if this approach were well-founded, then the recent meta-analysis of VSLs and related research effort by the OECD to establish recommended values – not to mention more than 50 years of progress in economic science since the pioneering work of Jacques Drèze – could well become redundant. There is, however, good reason to suppose that it is not well-founded.

First, as a matter of record, it should be noted – as indeed is noted in an important early paper for the US Environmental Protection Agency (US EPA) (Hubbell, 2002) – that the original interest of policy makers in the use of QALYs was as "an alternative method that can account for morbidity effects as well as losses in life expectancy, *without requiring the assignment of dollar values to calculate total benefits*". And as the US EPA Science Advisory Board advised at the time: whilst there was merit in using QALYs and therefore VOLYs in certain contexts and for certain purposes, "alternative measures, such as the VSLY or the value of a QALY, are *not consistent with the standard theory of individual WTP for mortality risk reduction*" (Hunt and Ferguson, 2010; and Hunt, 2011).

Of the many ways in which the new approach can violate the letter and spirit of the standard theory, the following deserve special mention:

- Non-monetised QALYs, however useful they are to health professionals, reflect their valuations of the morbidity suffered by others – not valuations by representative individuals in the general population – and this will necessarily flow through into their monetisation.
- VOLYs are rarely derived from WTP surveys even today (Hunt, 2011) – even if it is in principle possible to do so – and therefore also reflect the valuations of external parties.
- However they are derived, VOLYs will necessarily produce results that differ from, and are inconsistent with, the results given by VSLs: the cost of the death of a group of people of a given age will automatically be counted as less than the death of a comparable group of younger people with otherwise

identical characteristics since the number of LYL for the former group will be less than that for the latter.

- Whether monetised or not, QALYs can involve an element of "double jeopardy" (Hubbel, 2002) as described in Hubbel (2006): "If the QALY loss is determined based on the underlying chronic condition and life expectancy without regards to the fact that the person would never have been in that state without long term exposure to elevated air pollution, then the person is placed in double-jeopardy. In other words, air pollution has placed more people in the susceptible pool, but then we penalize those people in evaluating policies by treating their subsequent deaths from acute exposure as less valuable, adding insult to injury, and potentially downplaying the importance of life expectancy losses due to air pollution."
- The combination of counting LYL, rather than lives lost, and carrying through pre-existing conditions means that the VOLY-QALY approach "explicitly places a lower value on reductions in mortality risk accruing to older populations with lower quality of life" (Hubbel, 2002).

Now it would be dogmatic to conclude that the search for a composite method will necessarily fail to resolve these issues in a manner that is compatible with economic first principles. It is clear, however, that this search has not arrived at such a destination and cannot today offer a set of values that are in any way compatible with the OECD-recommended values for VSLs that this report is tasked to apply.

Against this background – the availability of a singular standard method for calculating mortality costs, a well-founded search for a plural method for calculating morbidity costs which is not yet complete, an also-incomplete search for a singular method which may be fatally flawed – the approach adopted in this report is to concentrate on the task at hand. As such, the study reports on both mortality and morbidity *impacts* of air pollution but calculates *costs* for mortality only, and using only the OECD-recommended values for VSLs – and then adds to this only a provisional indicative estimate of the additional cost imposed by morbidity.

It follows that if the OECD and its member-governments wish to calculate the economic costs of air pollution's impact on morbidity on a par with the calculation of the economic costs of air pollution's impact on mortality offered below, it is necessary to build an economically robust evidence-base on morbidity on a par with the economically robust evidence-base on mortality established in OECD (2012).

1.3. The dominance of mortality costs over morbidity costs

As is indicated below and in the discussion in Chapter 2, the costs of morbidity are large. As a result, it would indeed be advisable to capture more precisely these costs and their constituent parts in order to develop more effective interventions to reduce them. But mortality costs are, and necessarily so, *much larger*. In any defensible calculation of "economic costs" properly defined, mortality dominates over morbidity as a share of the total economic cost of health impacts from air pollution.

The most recent OECD report to address this point sums it up as such: "overall health costs are dominated by the cost of premature mortality; the order of magnitude changes vary significantly between morbidity and mortality." (Hunt, 2011 and the discussion following Table 2.1.)

This finding has been established for a long time. *Inter alia*, Hunt (2011) cites a 1996 report estimating morbidity costs at 15-45% of total costs, with mortality costs accounting for 55-85%. More recent research, with more accurate values, tends to attribute a much higher share to mortality costs. Hunt (2011) cites the 2010 study by the US EPA of the benefits of the 1990 Clean Air Act Amendments, attributing 93% of the benefits to reductions in mortality (Hunt, 2011, Table 2.6).

This last point, the progressive attribution of a larger share of the total to mortality, is best shown by concentrating on a single programme and its progress. From Hunt and Ferguson (2010), we can extract the following data on an early iteration of the Clean Air for Europe (CAFÉ) Programme, showing the effects of adding in, first, non-mortality WTP values and, next, mortality WTP values.

Table 1.1. **CAFÉ Programme cost-benefit analysis (CBA), with and without WTP values**

Benefits in reduced damage costs	EUR billions, 2005	As a % of programme cost
Medical cost	0.38	
Lost production cost	3.06	
Crop losses	0.33	
Materials	0.19	
Total	**3.96**	**56**
Adding in non-mortality WTP		
Non-mortality WTP	10.40	
New total	**14.36**	**202**
Adding in mortality WTP		
Mortality WTP	29.09	
Grand total	**43.45**	**612**

Source: Data reported in Hunt, A. and J. Ferguson (2010), *A review of recent policy-relevant findings form the environmental health literature*, OECD, Paris.

If valued by the individual's WTP, the benefits in reduced mortality account for 67% of the grand total. And WTP values account for 72% of the remainder. In short, mortality costs dominate morbidity costs; and the values for (dis)utility dominate the values for resource costs and opportunity costs.

The most recent CBA for the Thematic Strategy on Air Pollution (TSAP) (Holland, 2012), which builds upon the CAFÉ Programme, estimates the baseline damage costs as follows:

Table 1.2. **TSAP cost-benefit analysis (CBA), with mortality in VOLYs and VSLs**

Baseline health impacts from air pollution in year 2030 (%)	
All mortality – LYL – in median VOLY – as a % of the total (with median VOLY)	69
All mortality – LYL – in mean VOLY – as a % of the total (with mean VOLY)	84
All mortality – number of deaths – in median VSL – as a % of total (with median VSL)	83
All mortality – number of deaths – in mean VSL – as a % of total (with mean VSL)	91

Source: Data extracted from Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, corresponding to International Institute for Applied Systems Analysis (IIAC) Thematic Strategy on Air Pollution Report #7, EMRC.

On the basis of the OECD-recommended approach in OECD (2012) – calculating with mean VSLs – mortality costs claim a *91% share* of total costs in this European research, close to the *93% share* of total benefits reported for reductions in mortality in the US EPA study. In addition, the VSL values used in Holland (2012) pre-date the higher VSL values recommended in OECD (2012); applying the latter would yield a result *above 91%*.

Hence, the most recent evidence suggests that morbidity costs add to the total by *around 10% of the cost of mortality as given by mean VSLs*. And this is the estimate carried over as a provisional indicative estimate in the calculations of Chapter 2.

The further development of the plural method of calculating morbidity costs, including a more comprehensive calculation of WTP values, may well raise morbidity's share. But it is not credible to suppose that it would raise that share above that of mortality.

If despite this weight of evidence in the specialist literature, non-specialists are sometimes inclined to suppose that morbidity costs, and especially medical costs, are the dominant share of the economic costs of health impacts, it is only because of critical ambiguities in the use of the term "costs".

For example, a consultants' report for the US EPA from the turn of the century, reporting on "asthma costs" for 1997 (Chestnut, Mills and Agras, 2000), shows "direct costs" (medical expenditures in the treatment of illness) to be greater than "indirect costs", and "morbidity costs" to be greater than "mortality costs". But this is only because "indirect costs" are defined here as being "the market value of lost productivity (e.g., wages)". The authors themselves clearly warn that this is *not* the appropriate measure.[6] But to no avail: even today, that paper is sometimes used to question this critical scientific finding of the dominance of mortality costs.

And yet: how could economic science find otherwise? In the language of economics, cost is not a sum of money; cost is the loss of what we value. We value consumption, leisure, health and life. Jacques Drèze says: "People want to survive and consume, not starve!" To this should be added: "People want to live, in health if possible, in sickness if need be. In sickness and in health, people want to live!"

It is only from the contrary perspective of an ancient chrysohedonism, predating not only the 50 years' of progress in valuation since the early work of Jacques Drèze, but also the 250 years' of progress in the understanding of value since Francois Quesnay and Adam Smith – only from this perspective of "counting the King's money" – that medical expenditures can loom larger than life. Economic science provides a very different calculation.

Notes

1. To keep it manageable, the referencing in this report is restricted to items published in the twenty-first century. But the veracity of this claim – that is, the universal rejection of chrysohedonism by all major schools of economics from the mid-eighteenth century to the present day – can be checked easily enough by consulting inter alia the works of Francois Quesnay, Adam Smith, David Ricardo, Karl Marx, Leon Walras and Kenneth Arrow.
2. This is also described as "use value" as distinct from "exchange value" in the language of the classical economists and as "utility" in neo-classical and present-day economics.
3. To repeat: this is not to say that the impact on GDP is not interesting or that it should be left unreported. But it needs to be reported separately; and so do the reasons for that separation. There is a parallel here with the issue of GDP impacts of public investment projects. In recent years, in the case of certain high-profile projects, the UK Department of Transport has reported results in terms of both economic evaluation and national accounts: that is, both cost-benefit results and GDP impacts. But it has taken care to present these calculations separately and to explain the reasons for it. See for example *UK Department for Transport (UK DfT)* (2006).
4. See for example the recent paper by Qin, Li and Lui (2013) on how workers' *lack* of bargaining power in certain sectors, including especially agriculture, can distort the results.
5. On the current state of research on the costs of morbidity, see in particular Hunt and Ferguson (2010) and Hunt (2011).

6. See Hunt (2011), where the authors warn as follows: "It should be noted that COI [cost-of-illness] estimates are a useful measure of financial burden of disease, but they do not measure the monetary value of the full effect of disease on the welfare of the population and are therefore insufficient for a full cost-benefit analysis of public policies aimed at reducing morbidity or mortality. Willingness to pay (WTP) is the more appropriate measure of the change in welfare in cost-benefit analysis, because it reflects not just the financial effect but also the value people place on the effect on quality of life and longevity…. In addition, there is substantial evidence that WTP for reductions in mortality risk far exceed the expected value of lost earnings, which is the COI measure of the financial effect of premature mortality…"

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Chapter 2

Reviewing the evidence on and calculating the cost of the health impacts of air pollution

This chapter reviews the extensive new epidemiological evidence that has become available since the WHO's 2010 Global Burden of Disease study. It tabulates health impacts from ambient particulate matter and ambient ozone pollution – including deaths, years of life lost (YLLs), and disability adjusted life years lost (DALYs) – for all OECD countries plus China and India. This chapter also provides a new calculation of the economic cost of deaths from ambient air pollution for all OECD countries plus China and India, along with an additional indicative estimate for the cost of morbidities.

The statistical data for Israel are supplied by and under the responsibility of the relevant Israeli authorities. The use of such data by the OECD is without prejudice to the status of the Golan Heights, East Jerusalem and Israeli settlements in the West Bank under the terms of international law.

2.1. Improved reporting versus real changes in impacts and costs

The evidence reviewed in this chapter is, wherever possible, on a global scale. But parts of the discussion, and more especially the calculations, are conducted with special reference to People's Republic of China (hereafter "China"), India and the 34 member countries of the OECD. In an immediate sense, this restriction is the result of constraints in data availability. But the restriction is not eccentric: each of these three population blocks constitutes just under one-fifth of the world's population and together they make up its majority.

On a global scale as well as in the case of each major country or group of countries, the best available evidence today suggests that the health impacts of outdoor air pollution, including from road transport, are *considerably greater* than previously reported.

This is primarily a consequence of the *improved reporting* of the health impacts of air pollution, through the use of more advanced monitoring technology – in particular, the use of remote-sensing satellite technology in place of ground monitoring stations (Brauer et al., 2012; Evans et al., 2012; Amann, Klimont and Wagner, 2013) – and through the development of a more comprehensive and rigorous methodology for assembling and analysing the epidemiological data, as embodied in the Global Burden of Disease (GBD) 2010 study, and published in a series of papers in *The Lancet* in December 2012 and in a number of follow-up papers thereafter (Box 2.1).[1]

The net result is that a far larger number of premature deaths – that is, a far larger *share* of the *given* number of premature deaths[2] – is now classified by epidemiologists as being attributable to "ambient particulate matter (PM) pollution".[3]

GBD 2010 reports a global death toll from PM pollution for the year 2010 that is *four times greater* than the figure reported for the year 2000 in the World Health Organization's GBD study for 2000 (Figure 2.1). It is also more than two times greater than the figure reported for the year 2010 itself in the *OECD Environmental Outlook to 2050* (OECD, 2012a).

These ≈ 3.2 million deaths represent a significant toll; and, at ≈ 6% of the global total of premature deaths, a significant share of the GBD.

The toll on life and limb can also be expressed in terms of years of life lost (YLL) due to premature mortality and disability-adjusted life years (DALYs) – that is, the sum of years lost due to premature deaths *and* years lived in

Box 2.1. **The new epidemiological evidence-base on air pollution**

This study draws on an epidemiological evidence-base on air pollution that is markedly different to that used in most previous studies. The new evidence-base – and in particular the tabulations of deaths, years of life lost (YLLs) and disability-adjusted life-years lost (DALYs) produced in the Global Burden of Disease Study 2000 study and reproduced here – incorporates several critical scientific breakthroughs, of which the two most relevant are these:

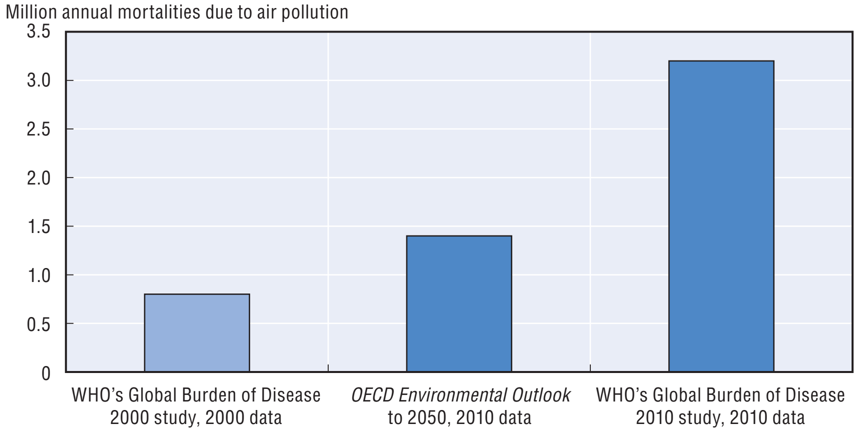
It makes use of a more advanced monitoring technology for measuring emissions and ambient concentrations of pollutants technology – in particular, the use of remote-sensing satellite technology in place of ground-based monitoring stations.

- For a further discussion of the matter, see Amann, Klimont and Wagner (2013), "Regional and Global Emissions of Air Pollutants: Recent Trends and Future Scenarios", *Annual Review of Environment and Resources*, Vol. 38, *http://environ.annualreviews.org*; Brauer et al. (2012), "Exposure Assessment of the Global Burden of Disease Attributable to Outdoor Air Pollution", *Environmental Science and Technology*, Vol. 46, *http://dx.doi.org/10.1021/es2025752*; and Evans et al. (2012), "Estimates of global mortality attributable to particulate air pollution using satellite imagery", *Environmental Research*, Vol. 120, *http://dx.doi.org/10.1016/j.envres.2012.08.005*.

It employs a more comprehensive and rigorous methodology for assembling and analysing the epidemiological data. The result is a more accurate assignment to each disease and each risk factor of its share in the given number of premature deaths – which in turn happens to entail a higher share being assigned to ambient air pollution than in most previous studies.

- As is highlighted in Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease: Generating Evidence, Guiding Policy*, *www.healthmetricsandevaluation.org/gbd/publications/policy-report/global-burden-disease-generating-evidence-guiding-policy*: "GBD was created in part due to researchers' observation that deaths estimated by different disease-specific studies added up to more than 100% of total deaths when summed. The GBD approach ensures that deaths are counted only once." And again ibid.: "To ensure that the number of deaths from each cause does not exceed the total number of deaths estimated in a separate GBD demographic analysis, researchers apply a correction technique called CoDCorrect. This technique makes certain that estimates of the number of deaths from each cause do not add up to more than 100% of deaths in a given year."

Figure 2.1. **Estimates of deaths from ambient particulate matter (PM) pollution**



Source: Data reported in or extracted from Cohen et al. (2004), "Urban air pollution", in Ezzati et al. (eds.) (2004), *Comparative quantification of health risks: Global and regional burden of disease due to selected major risk factors*, World Health Organization, Geneva, and Cohen et al. (2005), "The global burden of disease due to outdoor air pollution", *Journal of Toxicology and Environmental Health, Part A*, 68:1-7, 2005; OECD (2012), *OECD Environment Outlook to 2050: The Consequences of Inaction*, OECD Publishing, *http://dx.doi.org/10.1787/9789264122246-en*; and, for the final column, Lim et al. (2012), "A comparative assessment of burden of disease and injury attributable to 67 risk factors and risk clusters in 21 regions, 1990-2010: a systematic analysis for the Global Burden of Disease Study 2010", *The Lancet*, Vol. 380, pp. 2224-60, and Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

StatLink http://dx.doi.org/10.1787/888933012807

disability (for a definition of the key terms of art used in GBD2010, see Institute for Health Metrics and Evaluation, 2013b). On these counts, too, there has been a significant revision since the previous assessment. Whereas the share of global DALYs attributed to ambient PM pollution in the previous assessment was only 0.4%, the share attributed to this risk factor in the present assessment is 3.1% (Lim et al., 2012).

The economic costs of these health impacts as estimated in this report are also considerably higher than previously estimated in consequence of acts of improved reporting – the improved reporting of health impacts achieved in GBD 2010 as well as the improved reporting of VSLs achieved in OECD (2012b), both of which enter into the new calculation of costs.

These higher numbers for impacts and costs in consequence of improved reporting of impacts and costs are objectively *distinct* from, and should in presentation be *distinguished* from, real changes in impacts and costs – that is, the actual increases (or reductions) in the burden of impacts and costs borne by individuals and societies from year to year.

While the recent reporting improvements represent a significant breakthrough for science and it is important that future policy work should proceed on the basis of the new evidence-base rather than the old, the focus of what follows in this chapter is on capturing the real changes in impacts and costs that have obtained in the recent past and are likely to obtain in the prospective future.

Here, the record is more complex. As is explained in detail below, for the five-year period from 2005 to 2010, a relatively modest overall increase in the health impacts of air pollution is recorded as measured by the number of premature deaths, with YLLs and DALYs little changed – with an improvement in the OECD world as a whole (including in most, but not all, OECD countries) being offset by a larger deterioration in the rest of the world as a whole (including especially in China and in India).

The share of global DALYs attributed to ambient PM pollution in GBD 2010 shows little variation over the years from 1990 to 2000 to 2010, at ≈ 3%.

Attending this is a larger overall increase in the economic cost of these impacts – not only in the major emerging economies of China and India and in the rest of the world, but also in most OECD countries and in the OECD world as a whole.

It is important to note that there is a sense in which the health impacts of air pollution can increase (or decrease) without an actual worsening (or improvement) in air pollution itself as measured in physical metrics – and the cost of these impacts can increase (or decrease), even if the impacts themselves are unchanged.

As such, the natural course of economic development will tend to reduce the absolute toll, and the relative share of premature deaths, claimed by the characteristic diseases of poverty. The share of all other diseases will rise. And beyond a certain point, the characteristic diseases of the process of development will also be eclipsed by the rise of what might be called the diseases of affluence.

The change over time in the relative rankings of some selected "risk factors" in the GBD 2010 study[4] – when plotted on a sufficiently long time scale, such as over the twenty-year period from 1990 to 2010 – serves to make the point:

Table 2.1. **Selected risk factors ranked by attributable burden of disease in 1990 and 2010**

Risk factor	Ranking in 1990	Ranking in 2010
Childhood underweight	1	8
Ambient PM pollution	6	9
High body-mass index	10	6

Source: Data extracted from Lim et al. (2012), "A comparative assessment of burden of disease and injury attributable to 67 risk factors and risk clusters in 21 regions, 1990-2010: A systematic analysis for the Global Burden of Disease Study 2010", *The Lancet,* Vol. 380, pp. 2224-60.

The same point is seen more clearly still by comparing the relative rankings of the same selected risk factors across different regions:

Table 2.2. **Selected risk factors ranked by attributable burden of disease in selected regions in 2010**

Risk factor	Ranking in…			
	Central sub-Saharan Africa	South Asia	East Asia	Australasia
Childhood underweight	1	4	38	37
Ambient PM pollution	14	6	4	26
High body-mass index	18	17	9	1

Source: Data extracted from Lim et al. (2012), "A comparative assessment of burden of disease and injury attributable to 67 risk factors and risk clusters in 21 regions, 1990-2010: A systematic analysis for the Global Burden of Disease Study 2010", *The Lancet,* Vol. 380, pp. 2224-60.

The risk factor of "childhood underweight", a characteristic of poverty, holds the top rank in Central sub-Saharan Africa and a high rank in fast-developing but still-poor South Asia (India and its neighbours) – but is no longer a significant risk factor in fast-developing but no-longer-poor East Asia (predominantly, China) or in affluent Australasia (Australia and New Zealand). The risk factor of "high body-mass index', a characteristic of affluence, not to say excessive affluence, is of little importance in Africa or South Asia, of some importance in East Asia, and the single-most important risk factor in Australasia.

The high rank occupied by "ambient PM pollution" in both South Asia and East Asia (see also the regional analyses in Institute for Health Metrics and Evaluation et al., 2013a, and Institute for Health Metrics and Evaluation et al., 2013b) – but not in Central sub-Saharan Africa or Australasia – can be said to be the outcome of two related processes. On the hand, there is the strong and growing presence of air pollution, a function of economic growth and in particular, the expansion of industry, energy generation, and road transport. On the other hand, there is also the diminishing presence of the old diseases of poverty and the near-absence of the new diseases of affluence, a process that would serve to raise the ranking of air pollution, even if the level of air pollution itself were constant.

Nonetheless, the increase in the health impacts of outdoor air pollution reported in this book is real, irrespective of whether or not it is the outcome of a worsening of air pollution itself. An increase in the number of premature deaths from, say, cancers triggered by air pollution is a real increase in the death toll from air pollution – even if it is the result of the victims having benefited from a lower toll from childhood malnutrition before succumbing to the effects of air pollution. This change in relative rankings is no sleight-of-hand: it is a genuine signal to a society and economy emerging from classical poverty to attend to risk factors that now represent a genuinely greater risk.

In much the same vein, the economic cost of the said health impacts can also increase without an increase in the impacts themselves – as the willingness to pay (WTP) to reduce the risks of dying from air pollution rises in line with the rise in incomes. Once more, this is a genuine signal to society. In an ever-more affluent society, the individual's willingness to sacrifice that extra unit of consumption so as to reduce the risk of dying from air pollution – to sacrifice that extra unit of consumption which latter is also, as often as not, the route to an increase in body-mass index – is an entirely rational response.

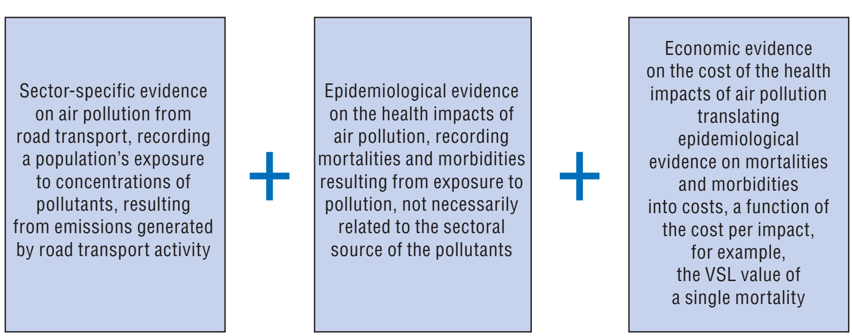
The real change in impacts and costs: Three links in the chain

The object of this study, the "economic cost of the health impacts of air pollution from road transport" is the end-point of a causal chain – and mirroring it, a chain of analysis – that may be schematically represented as in Figure 2.2:

Figure 2.2. **The economic cost of the health impacts of air pollution from road transport: Three links in the chain**

As such, the full evidence-base required for the final calculation necessarily draws on three separate evidence-bases, as shown in Figure 2.3:

Figure 2.3. **The economic cost of the health impacts of air pollution from road transport: Three bases of evidence**



If these evidence-bases were relatively stable, and if all other were equal, then, on the basis of historical precedent, the pattern of change in any given country might be broadly predictable as follows:

- An initial increase in air pollution from road transport as a result of higher levels of vehicle ownership and vehicle kilometres driven – gradually offset, and more than offset, by stricter pollution controls, in particular, progressively tighter regulatory standards for vehicle emissions.
- A similar pattern of a rise and fall in health impacts – with a lag reflecting the current incidence of mortalities and morbidities resulting from the past incidence of exposure to pollution as well as the change in the relative rankings of risk factors as described in the preceding discussion.
- A similar pattern of a rise and fall in economic costs – with a lag reflecting the predictable rise in WTP values as described in the preceding discussion.

The evidence reported below does not necessarily contradict this broad pattern.

Nonetheless, what can be reported at the present moment is complicated by the uneven development of the three relevant evidence-bases. The significant improvement in the epidemiological evidence-base established in the very recent past permits a new and robust calculation of the health impacts of air pollution. And the improvement in the economic evidence-base permits therewith a new and robust calculation of the economic cost of the health impacts of air pollution, at least in respect of mortalities. However, although there is a long-established tradition of sector-specific research on the issue of air pollution from road transport, the sector-specific evidence-base has not yet caught up with the developments in the epidemiological evidence-base: it is therefore not yet possible to provide an equally robust calculation of the share of the health impacts of air pollution, and the economic cost thereof, that is specifically attributable to road transport.

As is detailed below, all other things are not equal on at least one critical point: the changing share of diesel vehicles in the overall road transport vehicle fleet. And since new evidence on the health impacts of diesel is a critical part of the new epidemiological evidence-base, the phenomenon of "dieselisation" also complicates any calculation of the changing share of road transport in the overall health impacts of air pollution and the economic cost thereof.

Therefore, the procedure adopted in this chapter is to report as follows:

- a discussion of the available evidence on trends in regard to air pollution from road transport – but without attempting to calculate sector-specific pollution levels and their changes over time;

- a discussion, and also a tabulation, of the health impacts of air pollution and their changes over time[5] – but without regard to sector;
- a discussion, and also a calculation, of the economic cost of the health impacts of air pollution, and their changes over time[6] – with definitive estimates for mortality costs and indicative estimates for morbidity costs for the reasons provided earlier – and again without regard to sector;
- a discussion of the available evidence on road transport's share of the above economic cost, with no more than an indicative estimate of this share – and without attempting to calculate the change over time.

It follows therefore that, along with the need for further research on morbidity costs in a manner that is consistent with the established calculation of mortality costs, there is a clear need for further research on the sector-specific shares of the health impacts of air pollution and the economic cost thereof, in a manner that is consistent with the newly improved epidemiological and economic evidence-bases that are now available.[7]

2.2. Air pollution from road transport

If all other things were equal, then the pattern of economic and social development witnessed over recent decades would provide grounds for guarded optimism on the outlook for air pollution from road transport.

Economic growth brings with it a rise in vehicle ownership and vehicle kilometres driven. It also brings with it a societal demand for stricter pollution controls: in particular, progressively tighter regulatory standards for vehicle emissions, as witnessed in the progression in Euro vehicle classes in the European Union (EU) (and its equivalents in the United States), and the gradual if uneven adoption of these Euro standards (or its US equivalents) by countries around the world.

One might therefore expect that, in the course of time, in each country, the tightening of vehicle standards will eventually overtake the increase in vehicle kilometres driven, and turn the trajectory of road traffic-generated air pollution downward.[8] Indeed, one might also expect that, in the course of time, this downward trajectory will be further strengthened by a levelling off or even a reduction in vehicle kilometres driven, especially where it is encouraged by policy interventions promoting a mode shift to public transport (as well as cycling and walking) and by supportive investments therein (the best-known example of this being London: the data in Le Vine and Jones, 2012).

If tested against a suitably long time scale, this pattern has indeed come to pass in much of the OECD world. The most recent reports on air quality for the United States and the EU – by the American Lung Association, the US

Environmental Protection Agency (US EPA), the European Environment Agency (EEA), and others (see *inter alia* American Lung Association, 2013; EPA, 2013; EEA, 2013; Amann, Klimont and Wagner, 2013) – show a downward trend in emissions of pollutants, including from road transport, over the period of the last two decades, including the last full decade. Here, as Amman, Klimont and Wagner (2013) report "the evolution of air pollutant emissions has effectively decoupled from economic growth".

But all other things are not equal. In the world of the "emerging economies", which represents the majority of the world's population, the end of the upward trajectory of road traffic-generated air pollution is not yet in sight. In some cases because of the pace of growth of vehicle ownership, in some cases because of the lagged adoption of Euro vehicle standards (Amann, Klimont and Wagner, 2013) – and sometimes because of both factors – the increase in vehicle kilometres continues to outpace reductions in emissions per vehicle kilometre. And where unattended by sufficient mitigating action, rapid urbanisation continues to result in increased exposure of people to concentrations of pollutants.

In the OECD world as well as in the rest of the world, there is now another major complicating factor: the impact of diesel and the changing share of diesel vehicles in the overall road transport fleet. Depending on the pace of "dieselisation", there is a real risk that the downward trend in road traffic-generated air pollution discernible in most OECD countries could be arrested and reversed and that the still-upward trend evident in much of the rest of the world would be strengthened. This is a factor that needs to be highlighted in the survey below.

Air pollution in OECD countries

Given that the EU has set the pace on vehicle emission standards over the last two decades – and given that, as is detailed later, it is EU member states that have registered the sharpest reductions in deaths and disabilities from air pollution – EU data can serve duty here as a guide to the overall trend in the 34 OECD countries in this important respect: a slowdown, arresting or reversal of the positive trend in the EU may be taken as a warning to the rest of the OECD world.

To begin with the positive: for the decade from 2002 to 2011, the EEA (EEA, 2013) records an overall improvement in the trend of pollutant emissions, with reductions in emissions of primary PM – by 14% for PM_{10}, by 16% for $PM_{2.5}$ – and in emissions of its main precursor gases, 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. And thanks in large part to the introduction of progressively tighter emission limits for Euro 4 vehicles in 2005 and Euro 5 vehicles in 2009, the reduction in transport-specific emissions – 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.

However, this largely positive story also contains several less positive sub-plots:

- The link between ground-level emissions and ambient concentrations is highly complex. Thanks to the improvements in monitoring and modelling technology, it is now clear that this relatively rapid reduction in emissions has been attended by a much slower reduction in ambient concentrations.
- The percentage of the EU urban population exposed to PM_{10} levels exceeding the binding EU limit, and WHO's Air Quality Guidelines (AQG) limit, has actually increased since 2008. The EEA reports that, as at 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 AQG limit (EEA, 2013). Hence, the recent headline in *The Guardian*: "More than 90% of people in European cities breathe dangerous air" (*The Guardian*, 2013). Hence, too, the current collaboration between WHO and the EU in reviewing the evidence informing the EU limit (WHO, 2013a; WHO, 2013b; Henschel and Chan, 2013).
- The reported falls in transport emissions do not include non-exhaust emissions from road traffic (for example, from tyre and brake wear); clearly, these add to the real total and should be added to the reported total. In addition, being outside the EU regulatory standards regime, non-exhaust emissions are likely to constitute an increasing share of the total (EEA, 2013).
- Exhaust emissions under real-world driving conditions often exceed test-cycle limits and this excess may have a greater impact than assumed (EEA, 2013).
- Finally, and perhaps most critically for the future trend of emissions, there is the story of diesel.

There are four main points to be made here in regard to diesel:

1. So far as concerns their contribution to air pollution as distinct from their contribution to climate change, diesel vehicles are the more harmful by far. For example, of the exhaust emissions from all vehicles in London in 2009, 91% of $PM_{2.5}$ and 95% of NO_2 were attributable to diesel vehicles (Moore and Newey, 2012).
2. In contrast to the clearly downward trend recorded for petrol vehicles, including petrol hybrids, the most recent analysis of the subject (Carslaw and Rhys-Tyler, 2013) shows that "emissions from diesel vehicles of all types

have not shown significant reductions in NO_x for the past two decades…". Moreover, the fraction of NO_x emitted as NO_2 by diesel vehicles is not only very high – at around 25-30% as opposed to "only a few per cent" in the case of petrol vehicles – but has shown a variable rather than downward trend over the years (Carslaw and Rhys-Tyler, 2013; Carslaw et al., 2011).

3. Partly as a perverse consequence of policy and tax settings designed to combat climate change, the recent past has witnessed a continuing shift from petrol to diesel vehicles (Anable and Bristow, 2007) – that is, in the context of the present study, from less-polluting to more-polluting vehicles.
4. Therefore, if diesel vehicles continue to claim an increasing share of the vehicle market without reducing their emissions significantly, there is a real risk of an arresting and reversal of the downward trend in emissions from road transport. Indeed, there are some early signs of just such a reversal, in some cities in Europe, in the very recent past: from around 2011 onward (Carslaw et al., 2011).

These features of the record to date in Europe – the relatively high pollutant emissions from diesel vehicles and the failure to date to bring about a significant reduction in emissions for successive classes thereof – have been confirmed by research in other OECD countries, including in the United States and including on the part of those who maintain an optimistic view of the prospects for significant emissions reductions for *future* classes of diesel vehicles (Fujita et al., 2011; Health Effects Institute, 2012). So, too, has the emergence of a shift to diesel vehicles and, with it, the first signs of a slowdown in the downward trend of emissions from road transport, and the prospect of a possible reversal of this trend in the near future – not only in the cities at the heart of Europe but also in e.g. New Zealand (Kuschel, Bluett and Unwin, 2012).

Air pollution in China, India and the rest of the world

Beyond the OECD, in China, India and much of the rest of the world, the problem of "dieselisation" is one that compounds prevailing upward trends in emissions rather threatening to arrest and reverse a downward trend. Nonetheless, the problem is very much present: indeed, the term itself is borrowed from the literature in India (Centre for Science and Environment [CSE], 2013a).

The upward trends visible in China, India and the rest of the world are by no means uniform; nor do they arise from the same combination of factors in every case.

China, for example, has a relatively strict vehicle standards regime compared to most emerging economies, with targets for introducing and progressing through the Euro vehicle classes at a pace that is not too far behind the EU itself – with Euro 4 (adopted in the EU in 2005) being adopted in Beijing in 2008 and in Shanghai and Guangzhou in 2010, and with Euro 5 (adopted in the EU in 2009) being adopted in Beijing in 2012 (Amann, Klimont and Wagner, 2013, Figure 2; Ministry of Environmental Protection of the People's Republic of China, 2010; Wang and Hao, 2012). In addition, China has as of yet held a strong ambition to achieve future reductions in traffic-generated emissions by means of an accelerated shift to electric vehicles (EVs) (Ji et al., 2012).

Here, the first and foremost driver of the still-upward trend in traffic-generated emissions is the historically unprecedented growth of traffic itself, a function of the historically unprecedented economic growth achieved by China. In the three years from the end of 2008, China's car population effectively doubled from around 50 million to around 100 million, overtaking Japan as the second largest car population in the world (Wu et al., 2012). Unsurprisingly, this has led to a new and widespread societal demand for considerably more stringent regulatory standards than the ones already in place (*People's Daily Online*, 2011; Ouyang, 2013; Shang et al., 2013) – and to which the Government of China has now responded (*The Lancet*, 2013).

This may well lead to new directions as well as a faster pace of change. For even if China were to escape the worst effects of dieselisation through a switch to EVs, its reliance on coal-generated electricity will limit the capacity of EVs to reduce overall pollutant emissions, even if they do succeed in reducing emissions from traffic (Ji et al., 2012).

India's pace of adoption of Euro vehicle classes is also more ambitious than most emerging economies – with Euro 4 being adopted in 13 major cities in 2009 (Amann, Klimont and Wagner [2013], Figure 2). But this has not stopped a continuing worsening of pollution in recent years, with 50% of the 180 cities monitored in 2010, and 60% of the population of these cities, exposed to PM_{10} levels that are now designated as "critical" (CSE, 2013a; CSE, 2003b). Here, the prominent drivers include both the growth in traffic attending India's rapid economic growth and the problem of dieselisation – as well as traffic management systems that manage to deliver some of the severest congestion levels in the world.[9]

In the case of India's neighbours in South Asia, as well as further afield in Asia and Africa, the problem is that vehicle standards in too many countries remain stuck at Euro 1 or at best Euro 2 (Amann, Klimont and Wagner, 2013, Figure 2). There is little in place to offset increased emissions from increased traffic when traffic does begin to increase, as it doubtless will one day, at something approaching the pace of China and India.

Thus, for China, India and the rest of the world, for many and varying reasons, the upward trend in traffic-generated air pollution that has been a feature of the recent past is also likely to continue to feature in the immediate future. What is less clear is whether the downward trend discernible in the OECD countries in the recent past is likely to continue into the future – or whether it has already been arrested and reversed.

2.3. Health impacts of air pollution

Irrespective of the uncertainties in the sector-specific evidence-base, the epidemiological evidence-base can provide today a reasonably robust reading of the health impacts of outdoor air pollution from all sources over a period of years leading up to 2010. This is what is provided in the GBD 2010 study – in the series of papers published in *The Lancet* in December 2012 and in the follow-up papers thereafter (Lim et al., 2012; Institute for Health Metrics and Evaluation, 2013b) and in the data sets and tools available at the Institute of Health Metrics and Evaluation website (Institute for Health Metrics and Evaluation, 2013a).

Several key findings from an earlier date have prefigured the findings of December 2012 and thereafter. Importantly, in regard to establishing air pollution as a risk factor in lung cancer, earlier studies on the incidence of cancer amongst pollution-affected non-smokers (Beelen et al., 2008), and non-smoking miners in particular (Silverman et al., 2012), as well as the use of satellite-derived assessments (Brauer et al., 2012; Evans et al., 2012; Fajersztajn et al., 2013), have prefigured the most recent findings in *The Lancet* (Raaschau-Nielsen et al., 2013) and paved the way for the successive decisions by the International Agency for Research on Cancer (IARC) to classify diesel as a definite (Group 1) carcinogenic (IARC, 2012; Benbrahim-Tallaa et al., 2012) and outdoor air pollution as “a leading cause of cancer deaths” (IARC, 2013).

The new understanding of air pollution as a leading cause of cancer – when coupled with a fuller understanding of its role as a factor in myocardial infarction (Bhaskan et al., 2011) and in heart failure (Shah et al., 2013) and indeed a better understanding of the full range of its impacts on respiratory health (Laumbach and Kipen, 2012) – now permits a more comprehensive audit of the health impacts of air pollution, one that goes far beyond the old issue of asthma.

The GBD 2010 data sets include three risk factors under the heading of “Air pollution”, each of these being quantified without reference to the sectoral source. These are:

- “Ambient PM pollution”.
- “Household air pollution from solid fuels”.
- “Ambient ozone pollution”.

The first and third of these are relevant to the total of which traffic-related air pollution is a part; the second is not. What is tabulated and then calculated below are the impacts and costs of these two items, “Ambient PM pollution” and “Ambient ozone pollution”, combined under the heading of “Ambient air pollution”. The road transport share is then estimated on the basis of a reading of the literature.

Of the many other risk factors itemised in GBD 2010, there is one which is, arguably, also relevant: “Occupational exposure to diesel engine exhaust”. But given that it is not possible to separate transport and non-transport occupations within the data sets, and given that the literature does not offer a clear guide, this item is left unreported, albeit at the risk of under-estimating the full impact.

Both health impacts and economic costs are reported for two years: 2005 and 2010. This permits not only a reading of the *state* of things for the latest year for which the data is available but also a reading of the *trend* of recent years. What it cannot do is to provide a reading of the state of things today, as at the first half of 2014, and of how the trend has progressed since 2010. As a result, if the downward trend in the OECD countries has indeed been arrested as a result of dieselisation, and the upward trend in the rest of the world strengthened further, this is not something that can be captured in the tables below – though it is of course highly unlikely that the dieselisation to date could have been succeeded in reversing the downward trend for the OECD across the board as distinct from the trend related to road transport alone.

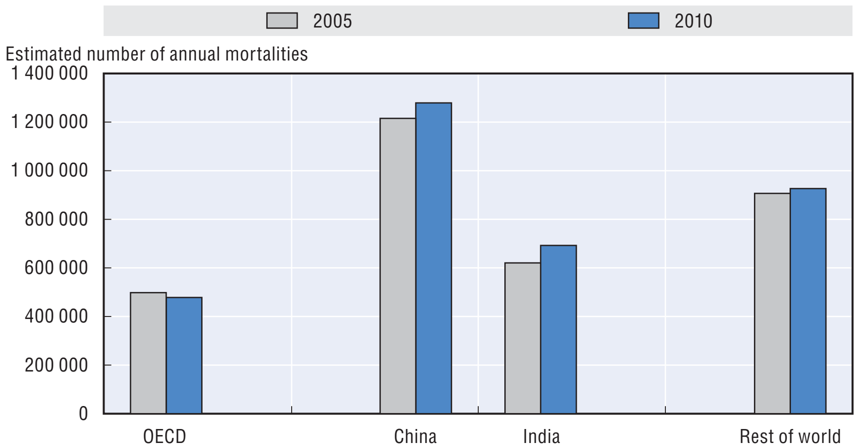
What is captured here first and foremost is that, on a global scale, the death toll from air pollution is high, much higher than was previously assumed (Laumbach and Kipen, 2012, Table 3). On a global scale, the toll continues to climb: the composite of a reduction in the OECD world, more than offset by a larger increase in the rest of the world.

More precisely,[10] 2010 witnessed a total of ≈ 3.376 million deaths from ambient air pollution, an increase of over 135 000 relative to 2005; the composite of a reduction of ≈ 20 000 in the OECD countries, offset by an increase of ≈ 20 000 in the rest of the world other than China and India, and more than offset by an increase of over 135 000 in China and India (Figure 2.4). In percentage term, the global death toll increased by ≈ 4% over this period.

Arguably, part of the continuing increase in the global death toll represents a predictable lagged effect, reflecting the *current* incidence of mortalities and morbidities resulting from the *past* incidence of exposure to pollution. This is seen more clearly in the comparison of the absolute number of deaths to the number of life years lost (YLL) and the number of DALYs (Table 2.3). On these last two counts, the toll from air pollution appears to have peaked.

Figure 2.4. **Deaths from ambient air pollution**

Total number of deaths from ambient particulate matter (PM) and ozone pollution by region in 2005 and 2010



Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle. *http://viz.healthmetricsandevaluation.org/gbd-compare/.*

StatLink http://dx.doi.org/10.1787/888933012826

Table 2.3. **Deaths, YLLs and DALYs from ambient air pollution in 2005 and 2010**

Global total

Health impacts from ambient PM pollution and ambient ozone pollution	2005	2010	Change from 2005 to 2010 (%)
Deaths	3 240 129	3 375 977	4.2
YLLs (years of life lost)	75 306 340	74 829 050	-0.6
DALYs	78 658 710	78 619 250	-0.1

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle. *http://viz.healthmetricsandevaluation.org/gbd-compare/.*

But it would be inadvisable to draw too hasty a conclusion. As is shown later, YLLs and DALYs continue to increase in China and India. The apparent global peaking is largely reducible to the downward trend now discernible in the OECD world. Elsewhere, it is, at best, a case of a slowdown in the rate of increase in YLLs and DALYs, not a peak that has been crossed.

Similarly, as is shown below, the global average YLL per mortality, at 23 years in 2005 and 22 years in 2010, disguises non-trivial differences between the major regions, the average being higher in China than in the OECD and higher still in India.

On one point, however, the experience across the world is indeed similar: DALYs impose a relatively small, though increasing, addition to YLLs. Once more, there are non-trivial differences between the major regions. But nowhere does this addition exceed 10%. In the epidemiological audit of the health impacts of air pollution, as in the economic calculation that seeks to translate these impacts into costs, mortality dominates morbidity by far.

Health impacts in OECD countries

Tables 2.4-2.6 report per-country deaths, YLLs and DALYs from ambient outdoor air pollution in 2005 and 2010 for each of the OECD member countries.

As was noted above, this recent period has witnessed a reduction in the total number of deaths for the OECD countries as a whole. As is shown in Table 2.4, the reduction in deaths by 2010 relative to 2005 was ≈ 20 000, or ≈ 4% in percentage terms. And this reduction has been attended by a still larger reduction in YLLs and DALYs. Considered on this scale, this is quite clearly a record of progress in governmental- and non-governmental efforts to reduce the health impacts of air pollution, be it by prevention or by cure.

It is important to note, however, that it is not a uniform record of progress. As Table 2.4 records, 20 of 34 countries achieved a reduction in deaths; that is to say, 14 did not. Most European OECD member countries achieved greater or smaller reductions. Of the OECD's non-European members, the United States and Israel both achieved a reduction, though by much less in percentage terms than the best achievers in Europe; all the rest – Canada, Mexico and Chile in the Americas, Japan and Korea in Asia, as well as Australia and New Zealand – registered an increase in deaths from 2005 to 2010.

The pattern is broadly similar for YLLs as it is for deaths but with some important differences. As Table 2.5 records, the average YLL per mortality dropped from 16 years to 15 for the OECD countries from 2005 to 2010. The reduction in YLLs, at ≈ 8%, outpaced the reduction in deaths, at ≈ 4%. And at least some countries, including for example Japan and Korea, succeeded in registering a reduction in YLLs despite registering an increase in deaths. But it remains the case that most of the countries registering an increase in deaths also registered an increase in YLLs.

Table 2.6 shows that, for the OECD countries, DALYs added 8% to the total of YLLs in 2005, 9% in 2010. Thus, the pace of reduction in total DALYs, at 7%, outpaced the reduction in deaths but not the reduction in YLLs. Most of the countries that failed to achieve a reduction in deaths also failed to achieve a reduction in DALYs. So far as concerns comparisons within the OECD world, the above information on YLLs and DALYs does not change the *basic* pattern observable in the record of deaths (Institute for Health Metrics and Evaluation (2013a). The majority of OECD countries did make progress. A minority did not.

Table 2.4. **Deaths from ambient air pollution in OECD countries**

Total number of deaths from ambient PM and ozone pollution	2005	2010
Australia (+)	882	1 483
Austria	3 773	3 240
Belgium	6 341	5 811
Canada (+)	6 989	7 469
Chile (+)	1 329	1 398
Czech Republic	8 811	7 096
Denmark	1 929	1 886
Estonia (+)	191	538
Finland (+)	402	450
France	18 457	17 389
Germany	51 155	42 578
Greece	9 054	8 346
Hungary	11 712	9 376
Iceland (+)	19	22
Ireland (+)	528	713
Israel	2 656	2 548
Italy	36 314	34 143
Japan (+)	61 173	65 776
Korea (+)	21 127	23 161
Luxembourg	184	150
Mexico (+)	17 954	21 594
Netherlands	8 050	6 741
New Zealand (+)	220	294
Norway	393	225
Poland	29 679	25 091
Portugal (+)	3 623	3 842
Slovak Republic	4 543	3 805
Slovenia	1 044	900
Spain	16 182	14 938
Sweden (+)	1 048	1 077
Switzerland	3 085	2 744
Turkey (+)	28 045	28 924
United Kingdom	28 345	24 064
United States	112 721	110 292
OECD total	**497 958**	**478 104**

Note: Countries registering an increase in deaths over this period are identified thus: (+).
Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

Table 2.5. **Years of life lost (YLL) from ambient air pollution in OECD countries in 2005 and 2010**

Total number of YLL from ambient PM and ozone pollution	2005	2010
Australia (+)	13 048	20 631
Austria	51 406	42 492
Belgium	91 965	81 226
Canada (+)	105 853	108 151
Chile (+)	24 684	25 367
Czech Republic	139 944	107 859
Denmark	27 352	26 402
Estonia (+)	3 218	5 224
Finland (+)	6 404	6 785
France	263 810	239 531
Germany	723 908	588 833
Greece	125 497	113 199
Hungary	204 908	155 625
Iceland (+)	271	302
Ireland (+)	8 228	10 890
Israel	38 220	35 103
Italy	449 986	405 093
Japan	853 899	827 509
Korea	374 944	373 785
Luxembourg	2 872	2 217
Mexico (+)	377 739	448 436
Netherlands	123 632	98 707
New Zealand (+)	3 477	4 450
Norway	5 371	2 984
Poland	527 605	424 174
Portugal	52 978	52 572
Slovak Republic	77 542	62 935
Slovenia	16 342	13 041
Spain	228 175	200 810
Sweden	13 526	13 320
Switzerland	38 485	32 490
Turkey	724 885	714 847
United Kingdom	413 108	339 411
United States	1 827 157	1 727 891
OECD total	**7 940 439**	**7 312 212**

Note: Countries registering an increase in YLLs over this period are identified thus: (+).
Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

Table 2.6. **Disability-adjusted life years lost (DALYs) from ambient air pollution in OECD countries in 2005 and 2010**

Total number of DALYs from ambient PM and ozone pollution	2005	2010
Australia (+)	14 342	22 867
Austria	56 529	47 203
Belgium	102 949	91 436
Canada (+)	117 418	121 034
Chile (+)	26 742	28 111
Czech Republic	146 284	112 349
Denmark	29 659	28 824
Estonia (+)	3 328	5 425
Finland (+)	6 813	7 322
France	296 209	270 827
Germany	787 727	644 359
Greece	132 853	120 128
Hungary	212 918	162 393
Iceland (+)	295	336
Ireland (+)	8 873	11 920
Israel	43 307	40 604
Italy	500 154	452 474
Japan	952 111	935 296
Korea (+)	403 888	409 700
Luxembourg	3 163	2 447
Mexico (+)	401 234	475 869
Netherlands	136 561	110 589
New Zealand (+)	3 761	4 863
Norway	5 821	3 234
Poland	550 147	443 957
Portugal	56 471	56 387
Slovak Republic	80 593	65 616
Slovenia	17 550	14 126
Spain	247 959	220 668
Sweden	14 480	14 440
Switzerland	43 714	37 166
Turkey	747 339	739 145
United Kingdom	444 279	368 749
United States	1 994 473	1 906 741
OECD total	**8 589 944**	**7 976 605**

Note: Countries registering an increase in DALYs over this period are identified thus: (+).
Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

Health impacts in China

Ambient air pollution takes a greater toll in China than in any other country: with just under one-fifth of the world's population, China accounts for just under two-fifth of the global death toll. In the terms of the GBD classification, "ambient PM pollution" occupies a higher ranking as a risk factor in "East Asia" (predominantly, China) than anywhere else, accounting here for ≈ 12% of deaths and ≈ 6% of DALYs (Institute for Health Metrics and Evaluation et al., 2013a), as against ≈ 6% of deaths and ≈ 3% of DALYs for the world as a whole (Institute for Health Metrics and Evaluation, 2013b).

As is shown below in Table 2.7, in the period from 2005 to 2010, China registered an *increase* in the total number of deaths: by ≈ 64 000, or ≈ 5% in percentage terms.

Table 2.7. **Deaths from ambient air pollution in China in 2005 and 2010**

Total deaths from ambient PM and ozone pollution	2005	2010
China	1 215 180	1 278 890

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare.* Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/.*

On the other hand, the legacy of the past can be seen in the order of magnitude difference between the ≈ 5% increase in deaths and ≈ 0.5% increase in YLLs shown below in Table 2.8. As noted earlier, the current incidence of mortality in any given year reflects in part the past incidence of exposure to pollution. A reduction in exposure – to begin with, a reduction in the pace of increase in exposure – will therefore bear down on the number of YLLs more than on the number of deaths.

Table 2.8. **YLLs from ambient air pollution in China in 2005 and 2010**

Total YLLs from ambient PM pollution and ambient ozone pollution	2005	2010
China	24 440 869	24 584 438

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare.* Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/.*

DALYs added 4.5% to the total of YLLs in 2005 and more than 5% in 2010. The pace of increase in total DALYs was ≈ 1%.

Table 2.9. **DALYs from ambient air pollution in China in 2005 and 2010**

Total DALYs from ambient PM pollution and ambient ozone pollution	2005	2010
China	25 540 289	25 877 829

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

These numbers permit the conclusion that China has succeeded in slowing the increase in the health impacts of air pollution. But reducing these impacts is a task that has yet to be achieved.

Health impacts in India

India's death toll from ambient air pollution is, in one respect, less alarming than China's: the absolute *levels* are considerably lower. With just over 90% of China's population, India registers a death toll from ambient outdoor air pollution that is just over 50% of China's toll, as is shown in Table 2.10.

On the other hand, the *increase* in the death toll in India over the period from 2005 to 2010 is considerably more alarming: an increase of ≈ 72 000, or ≈ 12% in percentage terms, as against China's increase of ≈ 64 000, or ≈ 5%.

Table 2.10. **Deaths from ambient air pollution in India in 2005 and 2010**

Total deaths from ambient PM and ozone pollution	2005	2010
India	620 622	692 425

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

In this respect, it could be argued that India's lower absolute death toll reflects simply the fact of its lesser advancement along the road of industrialisation and motorisation and that the toll is likely to climb toward China's levels as India advances further along the same road. The current state of emissions controls (including especially outside the transport sector), the levels of compliance achieved and the prospects for further strengthening all suggest that the rate of increase in the death toll in India is likely to outpace that of China (Amann, Klimont and Wagner, 2013). In any event, so far as concerns the currently observable trend, the direction of change in India is further away than it is in China from the direction achieved by the OECD countries.

Over the same period, India's YLLs have also climbed considerably faster than China's: an increase of ≈ 3%, as against ≈ 0.5% in China. But what is perhaps more striking is that the average YLL per mortality in India as at 2010 stood at 26 years, as against 19 in China and 15 in OECD countries. These are non-trivial differences. In India, the long process of bearing down on the level of exposure and the number of YLLs and therefore the number of deaths is still in its infancy.

Table 2.11. **YLLs from ambient air pollution in India in 2005 and 2010**

Total YLLs from ambient PM pollution and ambient ozone pollution	2005	2010
India	17 678 291	18 219 353

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

DALYs in India added just under 4% to total YLLs in 2005 and just over 4% in 2010, the pace of increase over this period being ≈ 3.5%. Once again, the lower additional value for DALYs places India further away from the OECD countries than is China. A higher additional value is not only a reflection of people living more years in ill-health; it is also, and importantly, a reflection of ill people succeeding in staying alive.

Table 2.12. **DALYs from ambient air pollution in India in 2005 and 2010**

Total DALYs from ambient PM pollution and ambient ozone pollution	2005	2010
India	18 358 012	19 007 178

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*.

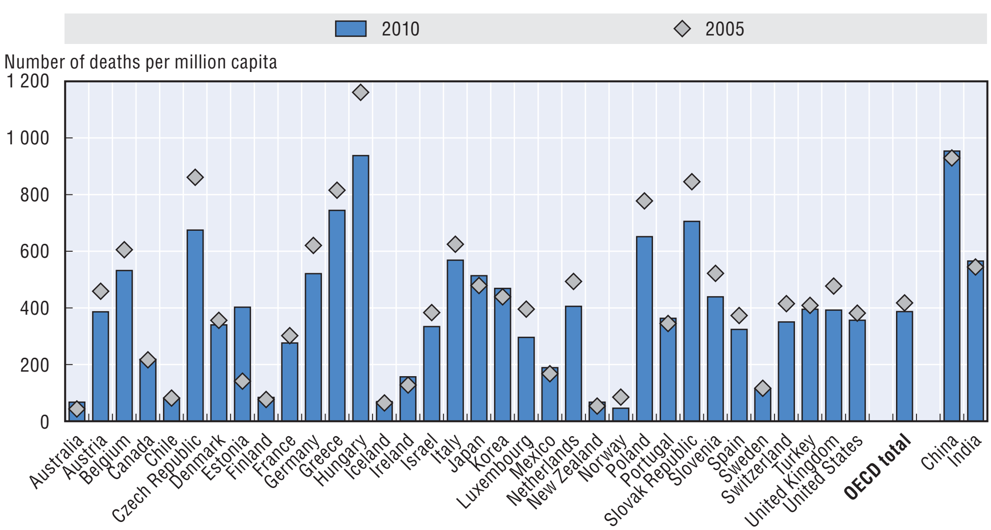
In regard to the challenge of achieving a sustained reduction in the health impacts of air pollution, it seems safe to say that India has further to travel than does China.

Finally, if China, India and the OECD member countries constitute not only a majority of the world's population but also a larger majority of those who are impacted by air pollution – ≈ 75% of the global death toll from ambient air pollution in 2010, according to Table 2.4 – there still remains the question of where the *rest of the world* is headed. But given that something like half the population of this "rest of the world" is situated in the other emerging economies of Asia, and at a level of development that is closer to India than to China, it is difficult to avoid the conclusion that they too are also likely to continue to contribute for a period to the continuing increase in the global toll.[11]

Mortalities from air pollution per million capita

Figure 2.5 illustrates deaths per million capita in each of the OECD countries, as well as China and India. Unsurprisingly, the numbers vary greatly – the result of a host of factors, including physical and economic geography, the historical legacy of industry, energy and transport patterns and the historical development of effective regulatory controls. It is nonetheless noteworthy that the trend of the change from 2005 to 2010 is much the same: a record of overall but not uniform progress across the OECD countries, including an increase in deaths per million capita in 12 of the 14 countries registering an increase in the absolute number of deaths. In China, there was a smaller increase in mortalities per million capita than described above

Figure 2.5. **Deaths from ambient air pollution in OECD countries, China and India, per million capita, in 2005 and 2010**



Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*. Population Statistics from *http://dotstat.oecd.org/Index.aspx*.

StatLink http://dx.doi.org/10.1787/888933012845 regarding the absolute number of deaths between 2005 and 2010; an increase of ≈ 2.5% – but an increase nonetheless. A similar development was found for India: a smaller increase (of ≈ 4%) when measured per million capita than when measured in absolute terms; but on this count, too, the deterioration in the result is greater than that recorded in China.

2.4. Economic cost of the health impacts of air pollution

As has been argued in the preceding discussion, there is now a robust epidemiological evidence-base on the health impacts of air pollution, even if there is work to be done in identifying the separate contributions of the various sectors to these impacts in a manner that is consistent with this evidence-base. And as was argued in the opening chapter, there is now a robust economic evidence-base on the calculation of the cost of mortalities, even if there is work to be done to complete the calculation of the multiple costs of morbidities in a manner that is consistent with this evidence-base.

It is therefore now possible to arrive at an estimate of the economic cost of the health impacts of air pollution that is reasonably robust, even if it is somewhat incomplete. As such, it is possible now to establish definite estimates of the cost of mortality resulting from air pollution for the countries in question – the 34 countries of the OECD as well as China and India – and indicative estimates for the additional impost of morbidity costs. It is also possible to advance, if somewhat tentatively, an indicative estimate of the share of road transport in these estimates of mortality costs as well as in the sum of mortality costs and the indicatively estimated morbidity costs.

Set out in this manner, what follows below also points to what remains to be done: namely, the completion of the calculation of morbidity costs, the completion of the calculation of road transport's contribution to mortality and morbidity costs, and the application of this completed protocol to any and all the world's countries as required.

For the present, the first step is to establish country-specific values for the economic cost of each life lost: that is, to establish country-specific VSLs by the formula recommended in OECD (2012b). The original formula is elaborated and illustrated below in Box 2.2. For the purpose of the present study, it is adapted in a slightly amended form, as is detailed in Box 2.3.

The following formula and the below tables need to be read with some important caveats.

The first concerns the very reason for country-specific VSLs. As argued at the outset of this report, a VSL value is meant to be an aggregation of individual valuations: an aggregation of individuals' WTP, as communicated through WTP surveys, to secure a marginal reduction in the risk of premature death. In the world as we know it, individuals are differentially endowed with

Box 2.2. **Calculating country-specific VSLs: Adjustment factors and illustrative example**

A preliminary internal WHO paper provided a useful summary of the set of adjustment factors used in calculating country-specific VSLs with the formula recommended in OECD (2012b) – along with an illustrative example, calculating the VSL value for Denmark in 2011. The material below has been slightly amended, with minor errors corrected, and reformatted for ease of presentation.

VSL EU	The Value of a Statistical Life (VSL) of the average of EU27 countries. According to OECD (2012b), the best estimate of the VSL is USD 3.6 million, with the range of USD 1.8-5.4 million (in 2005).
Y C	Gross Domestic Product (GDP) per capita at the purchasing power parity (PPP), in 2011. The GDP is converted to international dollars using the PPP rates.
Y EU	The average GDP per capita of EU27 countries at PPP, in 2011. This value equals USD 32 220.
β	Income elasticity of VSL. It measures the percentage increase in VSL for a percentage increase in income. The maximum value estimate is 0.8 and the minimum value estimate is 0.4.
PPP	Purchasing power parity-adjusted exchange rate, in 2005. PPP is the number of units of a country's currency required to buy the same amounts of goods and services in the domestic markets as US dollar would buy in the US.
%ΔP	The percentage increase in consumer price from year 2005 to 2011. This is measured by consumer price index (CPI) that reflects the inflation or changes in the cost to the average consumer of acquiring a basket of goods and services.
%ΔY	The percentage change in real GDP per capita growth from year 2005 to 2011. This is derived from real GDP per capita annual growth.

Calculate VSL in 2011 for Denmark, using the VSL of 3.6 million (USD in 2005)…

VSL in 2011 is calculated as:

$$\text{VSL 2011} = \text{VSL EU 2005} \times (\text{Y C/Y EU})^{\beta} \times \text{PPP} \times (1 + \%\Delta P + \%\Delta Y)$$

Find the values of each adjustment factors for Denmark…

$$\text{VSL 2011} = (\text{USD 3.6 million}) * (1.21) * (8.59 \text{ DKK/USD}) * (1.12) = 42.0 \text{ DKK}$$

Calculate the percentage change in real GDP per capita growth (%ΔY) and the percentage increase in consumer price in real (%ΔP) from 2005 to 2011 of Denmark…

Inflation and real GDP per capita annual growth of Denmark

	2005	2006	2007	2008	2009	2010	2011
Real GDP per capita annual growth	2.16	3.06	1.13	-1.37	-6.17	1.13	0.63
CPI	100						114

Box 2.2. **Calculating country-specific VSLs: Adjustment factors and illustrative example** *(cont.)*

The %ΔP from 2005 to 2011 is calculated as the difference between CPI in 2011 and CPI in 2005 (baseline) (i.e. 114-100 = 14% or 0.14)…

Calculate the real GDP per capita growth Index in 2011 as follows…

$$\text{Index } 2011 = 1.0306 * 1.0113 * 0.986 * 0.938 * 1.0113 * 1.0063 = 0.982.$$

Then, %ΔY is the difference between Index in 2011 and Index in 2005…

$$0.982 - 1 = -0.018 \text{ or } -1.8\%.$$

Source: OECD (2012), *Mortality Risk Valuation in Environment, Health and Transport Policies*, OECD Publishing, Paris, *http://dx.doi.org/10.1787/9789264130807-en*; and an internal WHO document.

Box 2.3. **Calculating country-specific VSLs: The formula applied in this book**

The OECD formula employed in the calculations below differs from the version detailed in Box 2.2 in three ways.

- The first is that the base value for the VSL calculation used here is the OECD's base value for the OECD group of countries as a whole, USD 3 million – not the base value for the EU group of countries, USD 3.6 million.
- The second is that the calculation is conducted entirely in PPP-adjusted USD estimates as published in the OECD's statistical database – not through national currencies and their reconversion into USD estimates at PPP.
- The third is that the income elasticity adjustment is applied not only to the 2005 level but also to its growth in the post-2005 period.

Thus, the formula used below is as follows:

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

the means with which to make such a trade-off: some work for their living for a dollar a day, some inherit a fortune yielding an unearned income of a billion dollars a year. Human societies without exception have sought to socialise these risks to a greater or lesser extent in the form of public goods – that is, in addition to measures designed to redistribute incomes to a greater or lesser extent. And it so happens that the level at which this socialisation of risks is executed today is the level of the nation-state.

It is for this reason, and this reason alone, that it is appropriate to aggregate at the level of country-specific VSLs – rather than at a lower level, say, that of a neighbourhood or, more realistically, a province, or at a higher level, say, that of the world as whole or, more realistically, a continent-wide Union of states.

It follows that differences in country-specific VSL values will tend to mirror the differences in country-specific per capita income levels in any given year. Thus it is that, in the base year of 2005, the VSL value for Germany is around double that of the VSL value for Poland. But of course the people of Mazovia (Mozawieckie) enjoy a higher level of per capita income than the people of Brandenburg – indeed, they enjoy today a higher level of per capita income than the EU average. It is rather the level of aggregation, reflecting the level at which the socialisation of risks is executed in reality, that results in a levelling down of the preferences of individuals in Warsaw and a levelling up of the preferences of individuals in Rostock to the point where the former falls below that of the latter. This is not at all a normative judgement on the part of the economist but simply a recognition of present-day reality.

It also follows that the economist's calculation would change if the socialisation of risks were in fact devolved down to a lower level or elevated up to a higher level. If for example there were to be a devolution down to the provincial level, then the calculation would deliver a higher VSL value for Mazovia than for Brandenburg. If on the one hand "an ever closer union" of the people of Europe were to eventuate one day, then the calculation would deliver a single VSL value for the EU.

Irrespective of any such institutional changes in the hypothetical future, there is of course the actual phenomenon of differential rates of growth of per capita income acting to change the differences in country-specific VSLs. In the usual case, this will entail a movement toward the convergence of VSL values as the relatively poor countries "catch up" with the relatively rich by means of the adoption of more advanced techniques of production. As such, it is unsurprising that the ratio of Germany's VSL to Poland's VSL falls from ≈ 2:1 in 2005 to ≈ 1.5:1 in 2010.

These observations apply *a fortiori* to the difference in VSL values between the United States, the technological lead country, and China and India. It should not occasion surprise that, in 2005 and also in 2010, the U.S. VSL value is a relatively high multiple of the VSL value for China and an even higher multiple of the VSL value for India, reflecting as it does the current large difference in per capita income levels.

But nor should it occasion surprise that the gap between the United States on the one hand and China and India on the other is narrowing at a faster rate than the gap between Germany and Poland. For whereas per capita GDP in the United States was at almost exactly the same level in 2010 as it was in 2005, it rose by ≈ 65% in China and ≈ 40% in India over this same period (OECD 2013; and World Bank, 2013). As has been clear for some decades, China and India are now deeply immersed in a process of development aimed at establishing a convergence in per capita income levels with the now advanced countries[12] – or more precisely, aimed at *re-establishing* the convergence that prevailed for most of the two millennia of the common era (Maddison, 2001 and 2003). It follows that, from year to year, over the next decades, all related numbers will change, including VSLs.

There is a final caveat to be entered. Given the level of aggregation involved, it is obvious that estimates of country-specific VSL values are no more than estimates; they cannot be precise and any pretence at precision would be inappropriate. Nonetheless, these values are a factor in the mathematical calculation of the overall result – that is, the economic cost of the health impacts of air pollution – and must therefore be entered into the relevant equations precisely. For the sake of transparency, the values reported in the tables below are therefore reported just as they are entered in the calculation – as whole dollars – before discussion in the text in appropriately larger units. This should not be read as an instance of "pretended precision": it remains the case that these estimates of VSL values are estimates.

Economic costs in OECD countries

The application of the OECD formula yields the VSL values for the OECD member countries for the years 2005 and 2010 as shown in the third and sixth columns of Table 2.13. And the application of those values to the epidemiological evidence reported earlier yields the estimates of the economic cost of deaths from ambient air pollution in each of the OECD member countries for the years 2005 and 2010, as shown in the fourth and seventh columns of Table 2.13.

As shown above, the economic cost of deaths from ambient air pollution for the OECD countries increased by ≈ 7% over the five years from 2005 to 2010. The cost of the death toll continues to climb even as the toll itself has ceased to climb. Notable too is the absolute figure: a sum of ≈ USD 1.6 trillion, which is, by any measure, a large sum.

As argued earlier, there is work to be done in completing the calculation of the costs of morbidities. But to complete the present discussion it is now appropriate to provide an indicative estimate of what this might add to the economic cost of mortalities as calculated in Table 2.13.

As noted in the closing section of the opening chapter, the most recent international research, in particular, the research in support of the EU's Thematic Strategy on Air Pollution (TSAP), suggests that morbidity costs add to the total by around 10% of the cost of mortality as given by mean VSLs (Maddison, 2001 and 2003; Holland, 2012). If applied to the new OECD-recommended values for VSLs, the supplement to be added would fall; on the

Table 2.13. **Economic cost of deaths from ambient air pollution in OECD countries in 2005 and 2010**

	2005			2010		
	No. of deaths, n (from Table 2.4)	VSL 2005 in USD millions[1]	n x VSL, 2005 USD millions	No. of deaths, n (from Table 2.4)	VSL 2010 in USD millions[2]	n x VSL, 2010 USD millions
Australia	882	3.380	2 981	1 483	3.925	5 821
Austria	3 773	3.283	12 386	3 240	3.670	11 892
Belgium	6 341	3.170	20 104	5 811	3.504	20 361
Canada	6 989	3.397	23 742	7 469	3.657	27 312
Chile	1 329	1.505	2 000	1 398	1.923	2 688
Czech Republic	8 811	2.275	20 045	7 096	2.749	19 508
Denmark	1 929	3.248	6 266	1 886	3.456	6 519
Estonia	191	1.860	355	538	2.269	1 221
Finland	402	3.052	1 227	450	3.319	1 494
France	18 457	2.960	54 633	17 389	3.155	54 863
Germany	51 155	3.085	157 788	42 578	3.480	148 182
Greece	9 054	2.535	22 951	8 346	2.824	23 570
Hungary	11 712	1.899	22 247	9 376	2.316	21 715
Iceland	19	3.388	64	22	4.456	98
Ireland	528	3.677	1 942	713	3.751	2 674
Israel	2 656	2.440	6 480	2 548	2.922	7 445
Italy	36 314	2.857	103 764	34 143	2.995	102 274
Japan	61 173	3.031	185 426	65 776	3.068	201 813
Korea	21 127	2.404	50 783	23 161	3.027	70 117
Luxembourg	184	5.779	1 063	150	6.283	942
Mexico	17 954	1.483	26 631	21 594	1.811	39 109
Netherlands	8 050	3.397	27 349	6 741	3.761	25 353
New Zealand	220	2.621	577	294	2.937	864
Norway	393	4.337	1 704	225	4.650	1 046
Poland	29 679	1.608	47 729	25 091	2.098	52 631
Portugal	3 623	2.284	8 273	3 842	2.499	9 603
Slovak Republic	4 543	1.828	8 302	3 805	2.418	9 202
Slovenia	1 044	2.462	2 570	900	2.898	2 608
Spain	16 182	2.785	45 074	14 938	3.059	45 691
Sweden	1 048	3.210	3 364	1 077	3.502	3 771
Switzerland	3 085	3.516	10 846	2 744	3.851	10 566
Turkey	28 045	1.381	38 725	28 924	2.024	58 548
United Kingdom	28 345	3.258	92 345	24 064	3.554	85 524
United States	112 721	4.088	460 751	110 292	4.498	496 145
OECD total	497 958		1 470 487	478 104		1 571 170

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

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; and OECD (2013), *OECD.Stat Extracts, http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*.

StatLink http://dx.doi.org/10.1787/888933012864

other hand, the supplement to be added could rise if the calculation of morbidity costs were to incorporate WTP values as fully as they should. For the present, therefore, it is not unreasonable to proceed with the assumption of a supplementary value of ≈ 10%.

Applying this supplementary value to the results established in Table 2.13, gives an indicative estimate of the full economic cost of the health impacts from ambient outdoor air pollution including mortalities.

Table 2.14. **Indicative estimate of the economic cost of health impacts from ambient air pollution including morbidities in OECD countries in 2005 and 2010**

	2005		2010	
	Mortality costs, USD millions (from Table 2.13)	Mortality + morbidity costs, USD millions if add-on ≈ 10%	Mortality costs, USD millions (from Table 2.13)	Mortality + morbidity costs, USD millions if add-on ≈ 10%
OECD countries	1 470 487	≈ 1 617 536	1 571 170	≈ 1 728 287

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*. Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; OECD (2013), *OECD. Stat Extracts. http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*; and Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, Corresponding to International Institute for Applied Systems Analysis (IIAC) Thematic Strategy on Air Pollution Report #7.

In sum, the health impacts of ambient air pollution as at 2010 imposed on the OECD member countries an economic cost of ≈ USD 1.7 trillion – rounded up, an economic cost approaching USD 2 trillion.

It is in light of *these* results that OECD governments need to assess the significance of any discrepancies that might obtain between the VSLs used in their national assessments and the per-country VSLs recommended by the OECD. In the United States, official guidance recommends a somewhat higher value (for example, US Department of Transportation, 2013); elsewhere, there are governments that use somewhat lower values. But in view of the magnitude of the damage imposed by the health impacts of air pollution, it is difficult to believe that small variations in VSL values could have any significant bearing on the case in favour of the mitigating actions being considered today in the United States, the EU, and elsewhere.

Economic costs in China

The same procedure as was applied above for the OECD countries can now be applied for China. The first step is to establish VSL values for 2005 and 2010 using the OECD-recommended method (the third and sixth columns of Table 2.15). The second step is to calculate the economic cost of deaths from

Table 2.15. **Economic cost of deaths from ambient air pollution in China in 2005 and 2010**

	2005			2010		
	No. of deaths, n (from Table 2.7)	VSL 2005 in USD millions[1]	n x VSL 2005 USD millions	No. of deaths, n (from Table 2.7)	VSL 2010 in USD millions[2]	n x VSL 2010 USD millions
China	1 215 180	0.610	741 019	1 278 890	0.975	1 246 713

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

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare,* Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; and OECD (2013), *OECD.Stat Extracts, http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*.

StatLink http://dx.doi.org/10.1787/888933012883

ambient outdoor air pollution for these two years (the fourth and seventh columns of Table 2.15).

The last step in the calculation, in Table 2.16, is to provide an indicative estimate of the full economic cost of health impacts assuming that the supplementary value for morbidity costs were ≈ 10%. But it is important to emphasise the indicative nature of this last calculation, given that the supplementary value of ≈ 10% is derived from EU research on EU data.

Table 2.16. **Indicative estimate of the economic cost of health impacts from ambient air pollution including morbidities in China in 2005 and 2010**

	2005		2010	
	Mortality costs, USD millions (from Table 2.15)	Mortality + morbidity costs, USD millions if add-on ≈ 10%	Mortality costs, USD millions (from Table 2.15)	Mortality + morbidity costs, USD millions if add-on ≈ 10%
China	741 019	≈ 815 121	1 246 713	≈ 1 371 384

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare,* Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; OECD (2013), *OECD. Stat Extracts, http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*; and Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, Corresponding to IIASA Thematic Strategy on Air Pollution Report #7, EMRC.

The 60% increase in China's VSL in the short period from 2005 to 2010 is noteworthy. But in view of China's rate of growth over the same period, including its resilience in the face of the intervening global recession, it is not surprising.

It does, however, contribute to the 70% increase in the economic cost of deaths.

China's economic cost burden on this count is comparable to that of the OECD countries – it was so in 2010, and is doubtless more so today.

In the absence of data, the EU-derived supplementary value of ≈ 10% may be added to provide an indicative estimate of the full economic cost. But note that China's DALYs added 5% to China YLLs, as against 9% in OECD countries. It should therefore be emphasised again that the indicative estimate provided is no more than indicative.

What is not doubted, however, is that the full economic cost burden for China is large.

Economic costs in India

The same procedure can now be repeated for India. The first step, in the third and sixth columns of Table 2.17, is to establish VSL values. The second step, in the fourth and seventh columns of Table 2.17, is to calculate the economic cost of deaths. Table 2.18 provides an indicative estimate of the full economic cost of health impacts.

Table 2.17. **Economic cost of deaths from ambient air pollution in India in 2005 and 2010**

	2005			2010		
	No. of deaths, n (from Table 2.10)	VSL 2005 in USD millions[1]	n x VSL 2005 USD millions	No. of deaths, n (from Table 2.10)	VSL 2010 in USD millions[2]	n x VSL 2010 USD millions
India	620 622	0.375	232 736	692 425	0.602	416 704

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

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*, Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; and OECD (2013), OECD.Stat Extracts. *http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*.

StatLink http://dx.doi.org/10.1787/888933012902

As shown above, India with its rapid rate of growth over this five-year period also registers a 60% increase in VSL. And when combined with an increase in deaths that exceeds the increase in China, it helps to deliver an increase in the economic cost of deaths greater than that in China – in this case, an 80% increase in the cost burden.

In regard to the use of the EU-derived supplementary value of ≈ 10% to provide an indicative estimate of the full economic cost, note that India's DALYs added 4% to India's YLLs, as against 9% in OECD countries. Once more, therefore, it should be emphasised that the indicative estimate provided below is no more than indicative.

Table 2.18. **Indicative estimate of the economic cost of health impacts from ambient air pollution including morbidities in India in 2005 and 2010**

	2005		2010	
	Mortality, USD millions (from Table 2.17)	Mortality + morbidity, USD millions if add-on ≈ 10%	Mortality, USD millions (from Table 2.17)	Mortality +morbidity, USD millions if add-on ≈ 10%
India	232 736	≈ 256 010	416 704	≈ 458 734

Source: Data extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare,* Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; OECD (2013), *OECD.Stat extracts, http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*; and World Bank (2013), World Data Bank, World Development Indicators, *http://databank.worldbank.org/data/home.aspx*; and Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, Corresponding to IIASA Thematic Strategy on Air Pollution Report #7, EMRC.

The level of the full economic cost burden in India at present is clearly less than it is in China or in OECD countries. But the direction in which it is heading is a matter of concern.[13]

2.5. Road transport's share of the above economic cost

The calculations above are calculations of the economic cost of the health impacts *of outdoor air pollution* – not of the economic cost of the health impacts of air pollution *from road transport*. It is not possible here to provide a definite estimate of this last item.

The problem is not the absence of estimates: on the contrary, there is a well-established tradition of research on the external costs of road transport, including the external costs of road traffic-generated air pollution and its health impacts. The problem is that these estimates are not comparable with, and consistent with, the epidemiological and economic evidence-base from which the calculations above are drawn. In particular, and as noted earlier, the sector-specific evidence-base has not yet had time to catch up with the recent, highly significant developments in the epidemiological evidence-base.

An authoritative assessment published in late 2011, and calculating for 27 European countries – the then 27 EU member states minus Cyprus[14, 15] and Malta plus Norway and Switzerland – estimated the economic cost of air pollution from road transport for these "EU27" countries in year 2008 at a total of EUR 50 610 million (CE Delft, INFRAS and Fraunhofer ISI, 2011). From Table 2.13 above, the economic cost of deaths alone in 24 European OECD countries – the 21 EU member-states plus Iceland, Norway and Switzerland, and hereafter called the EU24 – is estimated at USD 661 308 million. The two evidence-bases are not compatible. If the new epidemiological evidence is correct, the old economic estimate must be set aside: the share of road transport in this higher total cost cannot possibly be anywhere near as low as would be required to approach EUR 50 610 million.

What can be provided on the basis of the available literature is an indicative estimate of the *share* of the economic cost as calculated in this report that is likely to be attributable to the pollution generated by road transport – at least in regard to OECD countries where there is an available literature on road transport's share of pollutant emissions, health impacts and economic costs. This is what is provided below: indicative estimates for the OECD countries, beginning with estimates for the EU24, and some additional general remarks in regard to China, India and the rest of the world.

Estimates of road transport's share in OECD countries

The available literature, read with care, suggests that, in the EU24, road transport's share of the economic cost, properly calculated, is likely to be ≈ 50%.

A turn-of-the-century study covering Austria, France and Switzerland, originally prepared for the WHO and now part of the OECD environmental database,[16] estimated road traffic-generated air pollution to be responsible for 54% of the economic cost of air pollution's health impacts in the three countries taken together. In the period since the turn of the century, transport-sector pollutant emissions in the EU have fallen faster than pollutant emissions overall (Sommer et al., 2000; EEA, 2013). Other things being equal, this is likely to have reduced road transport's share. On the other hand, road transport's share is likely to be higher by a clear margin in some countries in the EU24 than in the three covered in this study. A more recent UK study (Moore and Newey, 2012) suggests that, as at 2008, road vehicles were responsible for 80% of PM emissions in London and 42% of PM emissions in Manchester. And recall that the years since then have seen a continuing shift to diesel.

The most recent relevant study, a Massachusetts Institute of Technology (MIT) study published in November 2013 (Caiazzo et al., 2013; Chu, 2013), calculates the shares of the various sectors in the total deaths from air pollution in the United States in 2005. This suggests that road transport's share of the total was 28% longer than at present or, alternatively, 34%, if the denominator is restricted to power generation, industry, aviation, and road, marine and rail transport.[17] Given that this is an estimate of the share *in the United States* – and given the long-established higher share claimed by power generation in the United States relative to the EU (see also the discussion of the differences between the two sides of the Atlantic in Caiazzo et al., 2013), and the correspondingly lower share left to be claimed by road transport – these findings are compatible with the suggested estimate of ≈ 50% as road transport's share *in the EU24*.

As such, and with due regard for the indicative nature of the estimate, the numerical results of this estimated share for the EU24 may now be estimated as follows:

Table 2.19. **Indicative estimate of road transport's share of the economic cost of deaths from ambient air pollution in EU24 in 2010**

	Economic cost of deaths from ambient air pollution, USD millions (from Table 2.13)	Share of economic cost attributable to road transport if road transport share ≈ 50%
EU24	661 308	≈ 330 654

Source: Data in middle column extracted from Institute for Health Metrics and Evaluation (2013a), *The Global Burden of* Disease (GBD) *Visualizations: GBD compare*, Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; and OECD (2013), *OECD.Stat Extracts*, *http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*.

Table 2.20. **Indicative estimate of road transport's share of the economic cost of health impacts from ambient air pollution including morbidities in EU24 in 2010**

	Economic cost of health impacts incl. morbidities, USD millions (from Table 2.14)	Share of economic cost attributable to road transport if road transport share ≈ 50%
EU24	≈ 727 439	≈ 763 720

Source: Data in middle column extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare,* Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; and OECD (2013), *OECD.Stat Extracts. http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*; and Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, Corresponding to IIASA Thematic Strategy on Air Pollution Report #7, EMRC.

And if – to repeat, if – the estimate of ≈ 50% drawn from EU data can be provisionally applied to the OECD countries as whole, and noting that the share is likely to be lower in the United States even if it is higher in some other non-EU OECD countries, the numerical results of this estimated share for the OECD countries may now be estimated as follows:

Table 2.21. **Indicative estimate of road transport's share of the economic cost of deaths from ambient air pollution in OECD countries in 2010**

	Economic cost of deaths from ambient air pollution, USD millions (from Table 2.13)	Share of economic cost attributable to road transport if road transport share ≈ 50%
OECD countries	1 571 170	≈ 785 585

Source: Data in middle column extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare*, Institute for Health Metrics and Evaluation, Seattl,. *http://viz.healthmetricsandevaluation.org/gbd-compare/*; and OECD (2013), *OECD.Stat Extracts, http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*.

Table 2.22. **Indicative estimate of road transport's share of the economic cost of health impacts from ambient air pollution including morbidities in OECD countries in 2010**

	Economic cost of health impacts incl. morbidities, USD millions (from Table 2.14)	Share of economic cost attributable to road transport if road transport share ≈ 50%
OECD countries	≈ 1 728 287	≈ 864 144

Source: Data in middle column extracted from Institute for Health Metrics and Evaluation (2013), *The Global Burden of Disease (GBD) Visualizations: GBD compare,* Institute for Health Metrics and Evaluation, Seattle, *http://viz.healthmetricsandevaluation.org/gbd-compare/*; OECD (2013), *OECD.Stat Extracts. http://stats.oecd.org/Index.aspx?DatasetCode=SNA_TABLE1#*; and Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, Corresponding to IIASA Thematic Strategy on Air Pollution Report #7, EMRC.

Estimates of roads transport's share in China, India and the rest of the world

It would be drawing too long a bow to apply this EU-derived estimate to China, India and the rest of the world: the difference in material circumstance is too large.

Indeed, there are some good reasons to suppose that road transport's share in the cost of the health impacts from air pollutions would be lower. These include:

- lower levels of per capita vehicle ownership, even in China;
- especially in China and in India: weaker regulatory controls on sectors other than transport, given the relatively strong vehicles standards regime in place, with its lagged adoption of Euro vehicle classes (Caiazzo et al., 2013; Amann, Klimont and Wagner, 2013);
- the extent of dependence on coal in power generation.

What can be said, however, is that a relatively lower share of the ≈ USD 2 trillion in economic costs borne by China and India thanks to air pollution is nonetheless a large burden on these countries. Mutatis mutandis, the same holds for the rest of the world.

Notes

1. See in particular, Lim et al. (2012) and Institute for Health Metrics and Evaluation (2013b). And see Cohen et al. (2004; 2005) – and the argument therein on the need for a better estimation of air pollution impacts.
2. In Institute for Health Metrics and Evaluation (2013b): "To ensure that the number of deaths from each cause does not exceed the total number of deaths estimated in a separate GBD demographic analysis, researchers apply a correction technique called CoDCorrect. This technique makes certain that estimates of the number of deaths from each cause do not add up to more than 100% of deaths in a given year."

3. Two very recent papers, building on the new monitoring and modelling technology, have leaned toward slightly higher and slightly lower numbers: Caiazzo et al. (2013), and Silva et al. (2013), respectively. But this does not alter the fact of the significant upward revision since GBD 2000 and need not detain the argument.
4. Note that these estimates for 1990 are on the basis of the new methodology: Lim et al. (2012).
5. Drawing primarily on the GBD 2010 evidence-base: in particular, Institute for Health Metrics and Evaluation (2013a) and Institute for Health Metrics and Evaluation (2013b).
6. Drawing primarily on GBD 2010 evidence-base plus the OECD's own evidence-base: in particular, OECD (2012a) and OECD (2013).
7. That is: consistent with GBD 2010 epidemiological evidence-base and the OECD economic evidence-base, as embodied in the sources cited in the preceding footnotes.
8. To be sure, the production of vehicles also generates pollution – though not to the extent generated by road traffic. It follows that the production of these new vehicles with tighter emissions standards will add to pollution at the various sites in the production process, including the process of extraction of raw materials used in vehicle production – though not to the extent that it subtracts from pollution on the roads.
9. See Amann, Klimont and Wagner (2013), Figure 2. Also Ministry of Environmental Protection of the People's Republic of China (2010), Wang and Hao (2012), and Banerjee et al. (2012). It is important to note that increased congestion contributes directly to an increase in air pollution: On this point, see *inter alia* European Commission (2011a) and more especially European Commission (2011b).
10. For ease of presentation, the reporting here is for the central estimate only, not the range. Figure 2.1 showed a total of 3.2 million deaths globally in 2010, while the total given here is close to 3.4 million deaths. The difference is that the present numbers include deaths caused by ambient ozone pollution, in addition to the deaths caused by ambient PM pollution covered in Figure 2.1.
11. For more direct evidence on the experience of other Asian economies, see Institute for Health Metrics and Evaluation (2013c). See also Miraglia, Saldiva and Böhm (2005), and Yaduma, Kortelainen and Wossnik (2012), respectively, for earlier country-specific research on the most populous countries elsewhere, namely, Brazil and Nigeria.
12. See *inter alia* Wilson and Purushothaman (2003) and Roy (2006), including the commentary on the former in the latter. As was argued in Roy (2006), the actual rate of convergence outpaced what was predicted in the Goldman Sachs model: "in this model, Chinese growth was expected to dip below 8% in 2005 and then fall gradually to 5% per year by around 2020. The actual outcome in 2005 was 9%. Indian growth was expected to climb to 6% in 2005 and then run at > 5% but < 7% per year through to 2050. In fact, India has been growing at above 7% per year since 2003." The argument in Roy (2006) applies *a fortiori* today.
13. And it clearly is a matter of concern in India: see inter alia CSE (2013a) and CSE (2013b).
14. Note by Turkey: The information in this document with reference to "Cyprus" relates to the southern part of the Island. There is no single authority representing

both Turkish and Greek Cypriot people on the Island. Turkey recognises the Turkish Republic of Northern Cyprus (TRNC). Until a lasting and equitable solution is found within the context of the United Nations, Turkey shall preserve its position concerning the "Cyprus issue".

15. Note by all the European Union Member States of the OECD and the European Union: The Republic of Cyprus is recognised by all members of the United Nations with the exception of Turkey. The information in this document relates to the area under the effective control of the Government of the Republic of Cyprus.
16. See Sommer et al. (2000). Importantly, the study calculates mortality and morbidity costs on a sound theoretical basis, with WTP-derived VSLs and WTP-derived components for morbidity costs.
17. See Caiazzo et al. (2013), Table 4, and also the commentary in Chu (2013). And note that "heating and cooking emissions" are not part of the total of "ambient air pollution" as defined in the present report and as derived from the GBD database.

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Chapter 3

Rethinking appraisals to mitigate the health impacts of air pollution from road transport

This chapter explores some of the policy implications of the cost burden of the health impacts of outdoor air pollution. It argues the need to maintain strong regulatory regimes – in particular, a strict vehicle standards regime – but also the need the rethink the regulatory and tax settings for diesel vehicles. This chapter also shows that the benefits of reducing the economic cost burden imposed by air pollution could easily outweigh the monetary costs of investments in more ambitious mitigation programmes, and that it is necessary, generally speaking, to rethink the approach to investment appraisals.

Given the size of the economic cost burden imposed by the health impacts of air pollution and the high share of it attributable to road transport – and even if this high share is relatively lower in some places than others – there are several straight-forward implications that follow in regard to the appraisal of policy interventions and investments designed to mitigate these impacts.

First, the findings recorded here confirm the need for, and the success of, strong regulatory regimes, in particular, a strict vehicle standards regime as exemplified *par excellence* in the European Union (EU). On the whole, and excepting the issue of "dieselisation" as noted above and also below, this regime has worked. It has succeeded in bringing down emissions, and consequent deaths and disabilities, in the OECD world, and especially in the EU component of it (as shown above in Tables 2.4-2.6).

They also send a signal to non-EU OECD countries, and to non-OECD countries, to consider the case for accelerating their envisaged timetables for catching up with the EU in the progressive introduction of the successive Euro vehicle classes.

Beyond the interventions already in place, the size of the economic cost burden imposed suggests that the benefits of reducing *these* costs could easily outweigh the monetary costs of investments in more ambitious mitigation programmes.

For example: the earlier-referenced cost-benefit analysis (CBA) for the Thematic Strategy on Air Pollution (TSAP) (Holland, 2012) estimates the benefits of progressively more ambitious scenarios, up to and including the "maximum technical feasible reduction" (MTFR), against their progressively higher monetary costs, expressed both in EUR millions per year and as a benefit-cost (B/C) ratio (Table 3.1 and Figure 3.1). Shown below are the estimates using mean value of statistical life (VSL), but these values pre-date the higher OECD-recommended values. The least-cost, least-ambitious scenario delivers benefits of more than EUR 50 billion at a B/C ratio of 142; the most-expensive, most ambitious scenario delivers benefits of more than EUR 150 billion at a B/C ratio of 3.8.

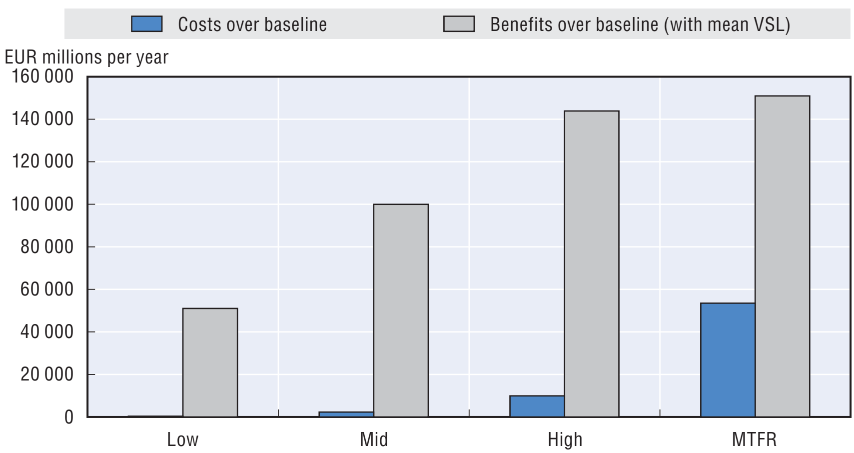
At a more local level and in more detailed ways, the new findings on the health impacts of air pollution can feed into the appraisal of several and various interventions, including on such topics as the optimal design of cycle paths in relation to roads and the optimal design of bus and tram stops, so as

Table 3.1. **TSAP CBA; costs, benefits and B/C ratios**

	Policy scenarios for 2030			
	Low	Mid	High	MTFR
	In EUR millions per year			
Costs over baseline	362	2 316	9 913	53 526
Benefits over baseline (total with mean VSL)	51 029	99 981	143 867	150 972
	Expressed as a B/C ratio			
Benefits over baseline (total with mean VSL)	142	44	15.5	3.8

Source: Data extracted from Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, corresponding to International Institute for Applied Systems Analysis (IIAC) Thematic Strategy on Air Pollution Report #7, EMRC, Table 5.1 and Table 5.2.

Figure 3.1. **TSAP CBA; costs and benefits**



Source: Data extracted from Holland (2012), *Cost-benefit Analysis of Scenarios for Cost-Effective Emission Controls after 2020*, Version 1.02, November 2012, corresponding to IIASA Thematic Strategy on Air Pollution Report #7, EMRC, Table 5.1 and Table 5.2.

to enable waiting users to face away from road traffic rather than toward it (Grabow et al., 2012, and Figliozzi and Monsere, 2013).

That said, the findings of this report also suggest the need to rethink the approach to appraisals in several particulars.

As detailed earlier, regulation has not succeeded in achieving significant reductions in nitrogen oxides (NO_x) emissions from successive classes of diesel vehicles (see above, sections under 2.2). At the same time, the now widespread tax differential in favour of diesel vehicles (on the question of how widespread this phenomenon is, see *inter alia* Upton, 2013) has helped to promote a shift to diesel within the overall vehicle fleet, with adverse consequences for air pollution and its health impacts. There is a clear need to re-think this issue within the appraisals that inform and influence decisions on regulation and taxation, and also, so long as regulatory and tax settings remain unchanged, within the appraisals that inform and influence decisions on wider interventions and investments in the field.

Not unrelatedly, there is a need to link the appraisal of policies on air pollution and policies on climate change. This is not simply or mainly because climate change policy may have had a hand in promoting the shift to diesel (see above, and Anable and Bristow, 2007). It is also because of the sizeable co-benefits available from tackling each with an eye on the other – not to mention the increasing convergence of the monitoring and modelling technology required to capture these two problems in the knowledge-base.*

There is also a clear need to revisit the numbers informing decisions on both pricing and investment in road transport, and especially in relation to its main alternative, rail.

It has been widely acknowledged for some time that, across a large part of the OECD world as well as elsewhere, relative prices between the modes are badly distorted. As a result, investment decisions are also acknowledged as being distorted by virtue of the demand schedules following from these distorted relative prices (Roy, 2008). For example, 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 (UK 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.

The new evidence on the external costs of road transport, as registered in the new evidence on the health impacts of air pollution reported in the present book, adds urgency to the need to revisit this issue – and to act it.

The final point is that there is a more subtle work of rethinking that is required. The extraordinary high net benefits and benefit cost-ratios in the TSAP CBA reported above suggest that something has gone wrong in the decision-making process: present investment proposals with extraordinarily

* On the related issues of air pollution and climate change policy co-benefits and scientific convergence, see *inter alia* Shindell et al. (2011a); Shindell et al. (2011b); Silva et al. (2013); and West et al. (2013).

Table 3.2. **Car, bus and rail revenues in relation to marginal social costs in Great Britain**

	GBP per passenger kilometre, 2004 data		Revenues/costs
	Marginal social costs (including external costs)	Revenues (fares, vehicle excise duty, fuel duty and VAT)	
Car	0.141	0.021	0.15
Bus	0.11-0.137	0.123	0.90-1.12
Rail	0.117-0.126	0.107	0.85-0.91

Source: Data extracted from DfT (2007), *Delivering a Sustainable Railway: Summary of Key Research and Analysis*, Department for Transport, London, *www.dft.gov.uk*.

high benefits suggest a past failure to invest in projects with ordinarily high benefits, high enough to have deserved passage.

As argued in Roy (2008), the optimal investment rule can be stated quite simply: proceed with investment if, and only if, it offers a positive net present value at the chosen discount rate, such that the present value of its discounted future streams of benefits exceeds the present value of its discounted future streams of costs:

$$NPV = PV_b - PV_c = b_0 - c_0 + \frac{b_1 - c_1}{(1+r)} + \frac{b_2 - c_2}{(1+r)^2} + \dots + \frac{b_n - c_n}{(1+r)^n} > 0$$

where NPV is net present value, PV_b is the present value of benefits, PV_c is the present value of costs, r is the discount rate, and n is the final year of evaluation.

And since the calculation of net present value incorporates the sacrifice of present consumption – and so long as each link in the chain is calculated accurately – it follows that:

> Strictly speaking, there is no *economic* constraint to – and no economic case against – the optimal investment rule … [to] proceed if, and only if, NPV > 0. To fail to invest in projects that offer positive net present values, calculated on the basis of accurate discounting, is not "prudence" – rather the contrary. To say that we "cannot afford" to invest, *after taking into account the value of the sacrifice of present consumption*, is simply a disguised way of admitting that we "cannot afford" to consume as we do (Roy, 2008).

Somehow the decision-making process has implicitly placed a value on consumption greater than the value that consumers place on it. There is, therefore, a job of work to be done in persuading decision-makers to read and act on the signals that society has been attempting valiantly to communicate.

This in turn requires an examination of what it is that has prevented these signals from being read. Such a task lies beyond the limits of the present report. But it would be as well to add a few words here on the probable location of the problem: namely, the intersection of public finance and economics (See Roy (2008) for a fuller statement of the argument).

In principle, it is true that, even in the case of the most worthwhile investments, a government *may* find itself in a position where it cannot borrow from the world's capital markets in sufficient quantity and at an affordable interest rate. Even if it were able to do so, it *may* find itself in a position where the investment expenditure, whilst of high economic value in the long term, carries the risk of negative macro-economic impacts in the short term, for example, by way of creating labour shortages and inflation. Over the years, it is by invoking just such hypothetical possibilities that many macro-economic experts have counselled, and many Finance Ministries have concurred, that an economically optimal investment policy is unrealistic.

It would therefore be helpful to table the evidence from the real world against these hypothetical possibilities invoked in the name of "realism". For the evidence shows that the economic take-off in People's Republic of China alone has vastly expanded the world's pool of savings, that the cost of borrowing for most OECD governments has never been lower, that governments have in fact borrowed a-plenty but borrowed to consume rather than invest, and that much of the excess savings of the emerging economies has been transformed into excess consumption in the most high-consumption economies – in place of much-needed infrastructural and other environment investments both in the OECD countries and in the emerging economies.

In addition, and especially so following the correction that commenced in 2008, the OECD world today is not in fact afflicted by labour shortages and inflation. On the contrary, there is today a macro-economic argument to be made to support the already-strong micro-economic case for public investments in a range of infrastructural and other environmental initiatives that would deliver high benefits to these societies over the long term (on this last point, see in particular Drèze and Durre, 2013). So it may be that it is in this new period of sobriety following the "irrational exuberance" of the run-up to 2008 that the case for higher levels of high-benefit public investments will receive its due hearing.

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The Cost of Air Pollution

HEALTH IMPACTS OF ROAD TRANSPORT

Outdoor air pollution kills more than 3 million people across the world every year, and causes health problems from asthma to heart disease for many more. This is costing OECD societies plus China and India an estimated USD 3.5 trillion dollars a year in terms of the value of lives lost and ill health, and the trend is rising. But how much of the cost of those deaths and health problems is due to pollution from cars, trucks and motorcycles on our roads? Initial evidence suggests that in OECD countries, road transport is likely responsible for about half the USD 1.7 trillion total.

Based on extensive new epidemiological evidence since the 2010 Global Burden of Disease study, and OECD estimates of the Value of Statistical Life, this report provides evidence that the health impacts from air pollution are about four times greater than previously estimated and the economic costs significantly higher than previously thought.

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