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What are the health costs of environmental pollution?

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Page 14: Figure 3: Typical data inputs used to assess health impacts of air pollution. Source: Anenberg et al. (2016)
Page 15: Inserting ear plugs © getty images / anetlanda
Page 16: Figure 4: Types of health impact cost. Based on OECD, 2016 (p23)
Page 19: Figure 5: Example of comparison of health states using QALYs © Science Communication Unit, UWE Bristol
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Page 42: Figure 9: Global estimated deaths (millions) by pollution risk factor, 2005–15. Source: The Lancet (2017).
Page 43: Figure 10: Global estimated deaths by major risk factor and cause, 2015. Source: The Lancet (2017)
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Contents

1.	**Introduction: the health costs of environmental pollution**	**5**
Glossary		**6**
1.1	Air pollution	7
1.2	Noise pollution	8
1.3	Toxic chemical exposure	8
2.	**Evidence for health impacts: the dose-response relationship and thresholds**	**9**
2.1	Causation	9
2.2	The dose-response or concentration-response relationship and thresholds	9
2.3	Air quality guidelines, thresholds and exposure limits	10
2.4	Noise guidelines, thresholds and exposure limits	15
2.5	Chemical exposure and probability of health effects	15
3.	**How to put a value on life, health and illness**	**16**
3.1	Health risk as an externality: market and non-market values	16
3.2	Quality-Adjusted Life Year (non-market valuation; often derived from stated preference)	19
3.3	The Disability-Adjusted Life Year	21
3.4	Cost of illness (market-based valuation)	24
3.5	Revealed preference methods (non-market-based valuation)	25
3.6	Environmentally attributable fraction (non-market-based valuation)	26
3.7	Willingness-to-pay (non-market valuation; stated preference)	26
3.8.	Willingness to accept (non-market valuation; stated preference)	27
3.9	Value of a statistical life (VSL) (non-market valuation; often derived from stated preference)	28
3.10	Value of a life year (VOLY) (non-market valuation; often derived from VSL or stated preference)	29
4.	**Examples: measuring benefits to health**	**30**
4.1	Quality-Adjusted Life Years	30
4.2	Disability-Adjusted Life Years	30
4.3	Revealed preference	33
4.4	Environmentally attributable fraction	33
4.5	Willingness to pay	33
4.6	Value of a Statistical Life	36
4.7	Value Of a Life Year	38
5.	**Applications**	**40**
5.1	Assessing policies through monetised benefits	40
5.2	Co-benefits	43
5.3	Internalising costs	45
6.	**Conclusion: harmonising the valuation of health impacts**	**46**
References		**49**

1.Introduction: the health costs of environmental pollution

From sleepless nights caused by traffic noise to death, quantifying the toll of environmental pollution on human health has been the subject of much research. Hospital visits and incidences of illness can be counted and linked to certain types of pollution through statistical analysis; and although life and health are of intrinsic value, ascertaining a monetary equivalent that reflects public preference for the allocation of scarce resources offers a practical metric for use in policymaking so that we can better take account of them. Such a metric needs to account for the full costs of health impacts, including burdens on healthcare services, reduced economic productivity and, most importantly, lost utility associated with premature death, pain and suffering. Calculating these costs has been the subject of a number of studies over the last 30 years, with the resulting figures informing both media headlines and cost-benefit analysis in the field of environmental policymaking.

This Future Brief outlines some of the methodologies that have been used to account for health costs, both in Europe and other parts of the world. The strengths and weaknesses of each methodology are considered, and their potential applications explored. Finally, the future directions of research in this field are investigated.

Health costs related to three key categories of pollution — air pollution, noise pollution and exposure to toxic chemicals — are touched upon with an introduction given to each. However, environmental pollution is not limited to these categories (it can also be linked to water pollution, indoor air pollution, biological contamination, ionising or UV radiation and more) but it is beyond the scope of this brief to cover each in detail.

Glossary: acronyms

CEA: Cost-effectiveness analysis
CBA: Cost-benefit analysis
COI: Cost of illness
DALY: Disability-adjusted life year
GBD: Global burden of disease
PM: Particulate matter
QALY: Quality-adjusted life-year
VSL: Value of a statistical life
VOLY: Value of a life-year
YLD: Years of healthy life lost due to disability
YoLL: Years of life lost
WTA: Willingness to accept
WTP: Willingness to pay

BOX 1.

The 'chrysohedonistic illusion'

A misconception that the wealth of the world subsists in gold or other forms of money; that undue focus is given to that which can be quantified in monetary terms. The attempt to monetise health impacts can create the (misleading) perception that actual benefits to intervention are modest, because it has only been possible to assign a quantitative or monetary value to some of those benefits. It is therefore essential that decisionmakers do not use the monetised numbers unquestioningly, do recognise the uncertainties associated with each, and do understand the assumptions behind them (OECD, 2014; European Commission, 2017a).

1.1 Air pollution

"Unless we clean up the air, by the middle of the century one person will die prematurely every 5 seconds from outdoor air pollution."
- OECD, 2016, p3

This prediction is based on a scenario in which levels of fine particulate matter ($PM_{2.5}$) and ozone continue to rise, leading to 6-9 million premature deaths annually by 2060. In some large cities, air pollution is already above recommended levels on a daily basis, and for several days a year may reach extremely dangerous levels (cf. Gao *et al.*, 2015). According to the World Health Organisation (WHO), 56% of cities in high-income countries do not meet their air quality guidelines.

The WHO is clear that particle pollution has health impacts even at very low concentrations — i.e. no threshold has been identified below which there is no damage to health. Therefore, the WHO recommend aiming for the lowest concentrations of particulate matter possible.

The map in **Figure 1** shows annual mean concentrations of fine particulate matter in urban areas across the world, illustrating how the problem

> BOX 2.
> **WHO air quality guideline values:**
>
> $PM_{2.5}$: 10 µg/m³ annual mean and 25 µg/m³ 24-hour mean
>
> PM_{10}: 20 µ/m³ annual mean and 50 µg/m³ 24-hour mean

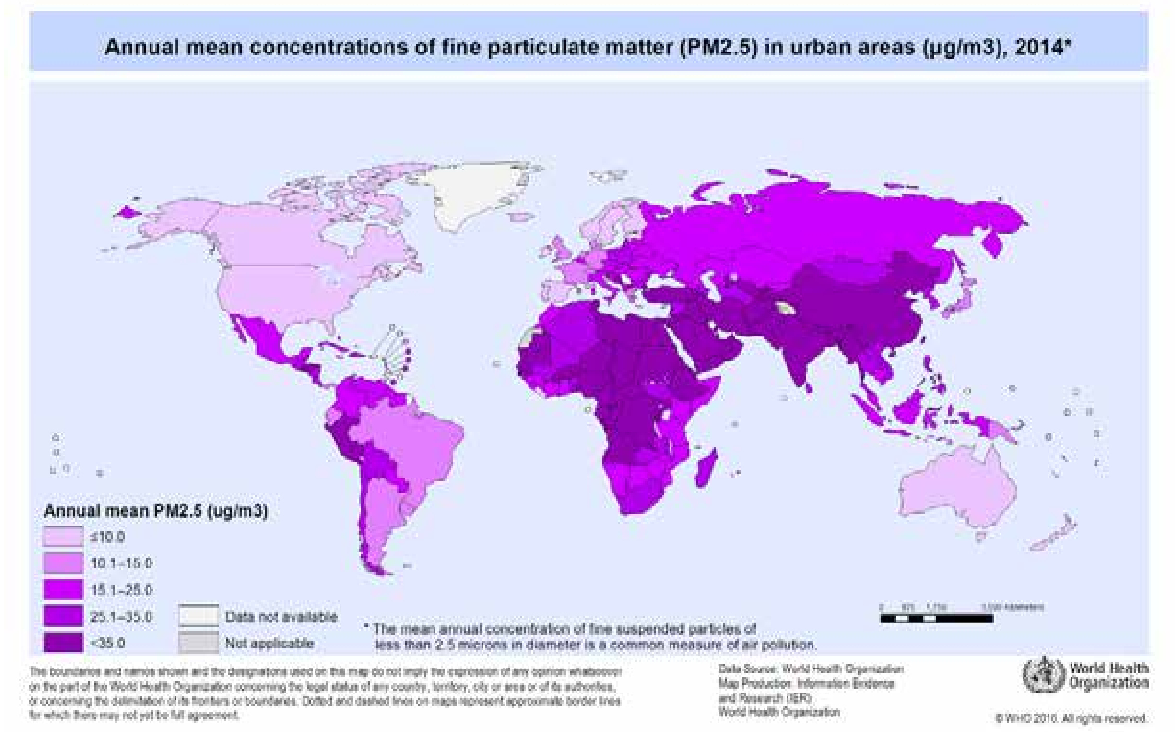

Figure 1: Annual mean concentrations of $PM_{2.5}$ in urban areas in 2014 (WHO, 2016).
Air pollution is linked to lung cancer, cardiovascular diseases (ischaemic heart disease and stroke), respiratory diseases (chronic bronchitis and asthma) and chronic obstructive pulmonary disease. Air pollution can also affect fertility and neonatal health, with consequences throughout the life course (RCP, 2016)[1].

1. This report from the Royal College of Physicians also raises possible links to diabetes, dementia and obesity.

is even worse in lower-income countries. For example, annual mean concentrations of $PM_{2.5}$ in urban areas in India in 2014 were 66 micrograms per cubic metre (µg/m^3) and in Uganda, 80 µg/m^3 (WHO, 2016), far exceeding guideline levels.

1.2 Noise pollution

Noise pollution has been linked to hypertension, heart attacks, stroke and dementia (Harding, 2013). It causes health problems by stimulating the nervous and endocrine systems, changing heart rate and blood pressure, and leading to the release of stress hormones including cortisol, with negative effects on wellbeing. It can also lead to cognitive impairment in children and tinnitus, and may even be a risk factor for diabetes (Dzambov, 2015).

More than 100 million EU citizens are estimated to be affected by exposure to high levels of noise from road traffic. Railways, airports and industry are also significant sources of noise pollution. Around 6 million have highly disturbed sleep as a consequence, while 69 000 hospital admissions and 15 900 cases of premature mortality are attributed annually to environmental noise in the EU (European Commission, 2017b).

1.3 Toxic chemical exposure

Both naturally occurring toxins (e.g. dioxins, arsenic) and manufactured chemicals (e.g. pesticides) can be poisonous to human health. There are tens of thousands of chemicals on the EU market, and while not all are likely to create health impacts, some (an unknown number) may be adding to the burden of disease (European Commission, 2017a). Exposure to several chemicals at once — even at low doses — may exacerbate or alter impacts (the 'cocktail effect') (Kortenkamp *et al.*, 2009). Exposure can occur via many pathways, including through inhalation of contaminated air and dust, ingestion of contaminated water and food, exposure through skin contact with products or foetal exposure during pregnancy (Prüss-Ustün *et al.*, 2011). Health effects of exposure can affect many different bodily systems, causing congenital disabilities, respiratory problems, neurodegenerative disease, skin disease, endocrine disorders or cancer, for example. The effects of exposure in childhood can be particularly profound.

The health effects of exposure to endocrine disruptors have drawn much attention in recent years (e.g. Bergman *et al.,* 2012). An endocrine disrupting chemical is an "exogenous substance or mixture that alters function(s) of the hormonal system and consequently causes adverse health effects in an intact organism, or its progeny, or (sub) populations" (European Commission, 2007)[2].

As there are so many different types of chemical exposure to consider, individual studies normally link individual and groups of chemicals to health effects. However, in recent years, there have been more studies that attempt to produce an overall estimate of the related burden of disease or of monetised benefits to health and environment from reduction of disease (e.g. European Commission, 2017a).

There is evidence that EU chemicals regulation has resulted in cumulative health and environmental benefits so far (European Commission, 2017a; European Commission, 2016), but there is also a need for better identification, testing and screening, and prioritisation of chemicals of concern. Human biomonitoring and approaches that anticipate toxicity may be useful to address these gaps (*ibid*).

2. More detailed, scientific criteria have recently been published in the 2017 Annex (C(2017) 5467 final) to the Commission Delegated Regulation EU/528/2012.

2. Evidence for health impacts: the dose-response relationship and thresholds

Before placing a value on health impacts, we need to know what those health impacts are. How many people are affected and in what way? Doing so requires an understanding of the impact pathway (how the impacts occur). In many cases, the impact is determined by looking at the intrinsic hazard (for example, of a chemical) and then at the exposure to it, to understand the risk.

2.1 Causation

Experimental research in the laboratory provides firm evidence that various pollutants can cause a specific effect (such as carcinogenicity). Laboratory experiments can control all variables, for example, the genetic composition of the organisms being tested and temperature and other environmental conditions.

However, there are limits to what can be tested in the laboratory, particularly in relation to the more important impacts on human health and 'cocktail effects' from simultaneous exposure to several chemicals under realistic conditions. In such cases, it is often necessary to rely on epidemiological research that assesses statistical evidence for 'association' between an agent (such as an air pollutant) and an effect (such as development of a specific cancer). However, such association can arise by chance, or because the effect and the pollutant under investigation are linked to a third, unidentified variable. Following from work on occupational diseases and drawing on similar experiences in relation to lung cancer from smoking, Sir Austin Bradford Hill (1965) developed a set of criteria for ascertaining whether epidemiological observations could be inferred as demonstrating a causal link between an agent and an impact, and these underpin the quantification of many types of impact, especially for air pollution.

Epidemiological studies have determined some of the strongest causal links between detrimental health effects and exposure to air pollution (e.g. Dockery & Pope, 1994; Lipfert, 2017; Perera, 2017; Seaton *et al.*, 1995), noise pollution (e.g. Babisch, 2006; Niemann *et al.*, 2006), asbestos (Boffetta, 2006; Świątkowska and Szeszenia-Dąbrowska, 2017), radon (Boffetta, 2006); and pesticides (Lai, 2017; Goldman *et al.*, 2017; Askari *et al.*, 2017; Androutsopoulos *et al.*, 2013).

The effects of single pollutants can be difficult to disentangle in epidemiological studies, as they are often part of complex mixtures from the same source. Road traffic, for example, generates pollution in the form of fine particles, nitrogen dioxide (NO_2), carbon monoxide and various other air pollutants, and noise. The situation becomes more complex as other determinants of health including socioeconomic status may also contribute to the same health effects. However, techniques are available for accounting, to at least some degree, for confounding factors.

2.2 The dose-response or concentration-response relationship and thresholds

The 'concentration-response function' or 'dose-response relationship' describes the size of the effect of a burden (e.g. a pollutant) on an individual or population after exposure to a certain concentration or dose (respectively). Furthermore, a 'hazard ratio' (risk rating) can be calculated by linking exposure to pollutants and specific health effects.

Research may suggest threshold levels of concentration or exposure, below which no or few harmful effects are likely, for humans and also other organisms. At the level of the individual, thresholds may depend on age — foetuses and children who are still developing may be more susceptible than adults.

At the societal level, accounting for the sensitivity of the population as a whole, there is often no evidence for thresholds above the level of background exposure. Levels of pollution considered 'acceptable', such as indicated by the air quality limit values, are subject to change, based on new evidence, and are debated. It is accepted that there is a safe threshold for most non-carcinogenic substances, but some researchers suggest that for some substances, there is no safe level[3]. For instance, several studies have concluded that there is no safe level of exposure to lead (Grandjean & Landrigan, 2014; WHO, 2017).

There is no known safe blood lead concentration. But it is known that, as lead exposure increases, the range and severity of symptoms and effects also increases. Even blood lead concentrations as low as 5 µg/dL, once thought to be a 'safe level', may be associated with decreased intelligence in children, behavioural difficulties, and learning problems.
- WHO Fact Sheet, 2017

The following sections highlight the role of guideline levels and dose-response functions in relation to air, noise and chemical pollution issues.

2.3 Air quality guidelines, thresholds and exposure limits

The 2005 WHO Air Quality Guidelines provide guideline values for key air pollutants that pose health risks, based on expert evaluation (WHO, 2006). These values provide a basic level of health protection across the population but for the most part do not reflect thresholds. The Guidelines are intended for worldwide use but have been developed to support actions for healthy air quality in different contexts, acknowledging the need of each country to set up its own air quality standards to protect the public health of their citizens based on local circumstances[4]. The EU Ambient Air Quality Directives (Directives 2008/50/EC and 2004/107/EC)[5] lay down the principal air quality standards (i.e. maximum pollutant concentration levels) for targeted pollutants that have a significant bearing on human health and ecosystem services. The respective standards may relate to differing measurement periods and threshold levels depending on whether impacts on human health and/or vegetation are induced by short and/or longer-term exposure to levels of air quality beyond which significant impacts have been observed. Different types of standards also come with differing (legal) connotations depending on the receptors and severity of the impacts (**Table 1**).

According to recent figures, 7-8% of the urban population in the EU-28 were exposed to levels of $PM_{2.5}$ exceeding limit values in 2013-2015 (EEA, 2017). **Figure 2** shows the location of $PM_{2.5}$ reporting stations across Europe for the purposes of assessing compliance with the Air Quality Directive, and variation in concentrations between them. Most show concentrations that comply with the current EU standard, but only a small number meet the WHO Guideline.

3. For some substances, there also exists the theory of 'hormesis', which is a biological phenomenon whereby exposure to low doses of certain chemicals or stressors may result in a beneficial effect (e.g. improved health, stress tolerance or longevity), even when high doses are otherwise toxic or even lethal (Mattson, 2009).
4. http://www.who.int/phe/air_quality_q&a.pdf?ua=1
5. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0050

Pollutant	Air Quality Guidelines (WHO, 2005)	EU Air Quality Standards (Directive 2008/50/EC)
$PM_{2.5}$	10 µg/m³ (micrograms per cubic metre) annual mean 25 µg/m³ 24-hour mean	25 µg/m³ annual mean (labelled as target value as of 2010, and as limit value as of 2015)
PM_{10}	20 µg/m³ annual mean 50 µg/m³ 24-hour mean	50 µg/m³ 24-hour mean (not to be exceeded more than 35 times a calendar year) 40 µg/m³ annual mean
O_3	100 µg/m³ 8-hour mean	240 µg/m³ 1-hour mean
NO_2	40 µg/m³ annual mean 200 µg/m³ 1-hour mean	40 µg/m³ annual mean 200 µg/m³ 1-hour mean (not to be exceeded more than 18 times a calendar year)
SO_2	20 µg/m³ 24-hour mean 500 µg/m³ 10-minute mean	125 µg/m³ 24-hour mean (not to be exceeded more than 3 times a calendar year) 350 µg/m³ 1-hour mean (not to be exceeded more than 24 times a calendar year)

Table 1: WHO Air Quality Guidelines (WHO, 2006); EU Directive 2008/50/EC.
http://apps.who.int/iris/bitstream/10665/69477/1/WHO_SDE_PHE_OEH_06.02_eng.pdf

With regards to health effects, figures for pollution-related mortality and morbidity (illness) differ depending on the response function used. For example, the European Study of Cohorts for Air Pollution Effects (ESCAPE) — the largest ever investigation in Europe into the adverse health effects of air pollution — showed that health risks (e.g. lung cancer) occur at concentrations well below the limit values shown in **Table 1** (Beelen *et al.*, 2014; Raaschou-Nielsen *et al.,* 2013).

The Global Burden of Disease (GBD) studies attributed 4.2 million deaths globally to fine particle ($PM_{2.5}$) pollution in 2015, and a further 254 000 to ozone (Cohen *et al.*, 2017; see **Section 3.3**). The function used in the GBD for health risk from different levels of $PM_{2.5}$ exposure was developed by Burnett *et al.* (2014), who note that they were restricted by a lack of long-term cohort studies in highly-polluted parts of Asia and the Middle East. Such studies are underway, therefore; as results from these become available in future, dose-response functions can be updated with new data.

The concentration-response functions recommended by the WHO project 'Health risks of air pollution in Europe — HRAPIE' are expressed as a relative risk (or risk ratio) per 10 µg/m³ average increase in the level of a pollutant over a specific time period (see **Table 2**) (Héroux *et al.*, 2015). The objective of HRAPIE was to provide evidence-based concentration-response functions for PM, O_3 and NO_2, which could support cost-benefit analysis (Holland, 2014a) and the 2013 EU Clean Air package[6].

6. http://ec.europa.eu/environment/air/clean_air/index.htm

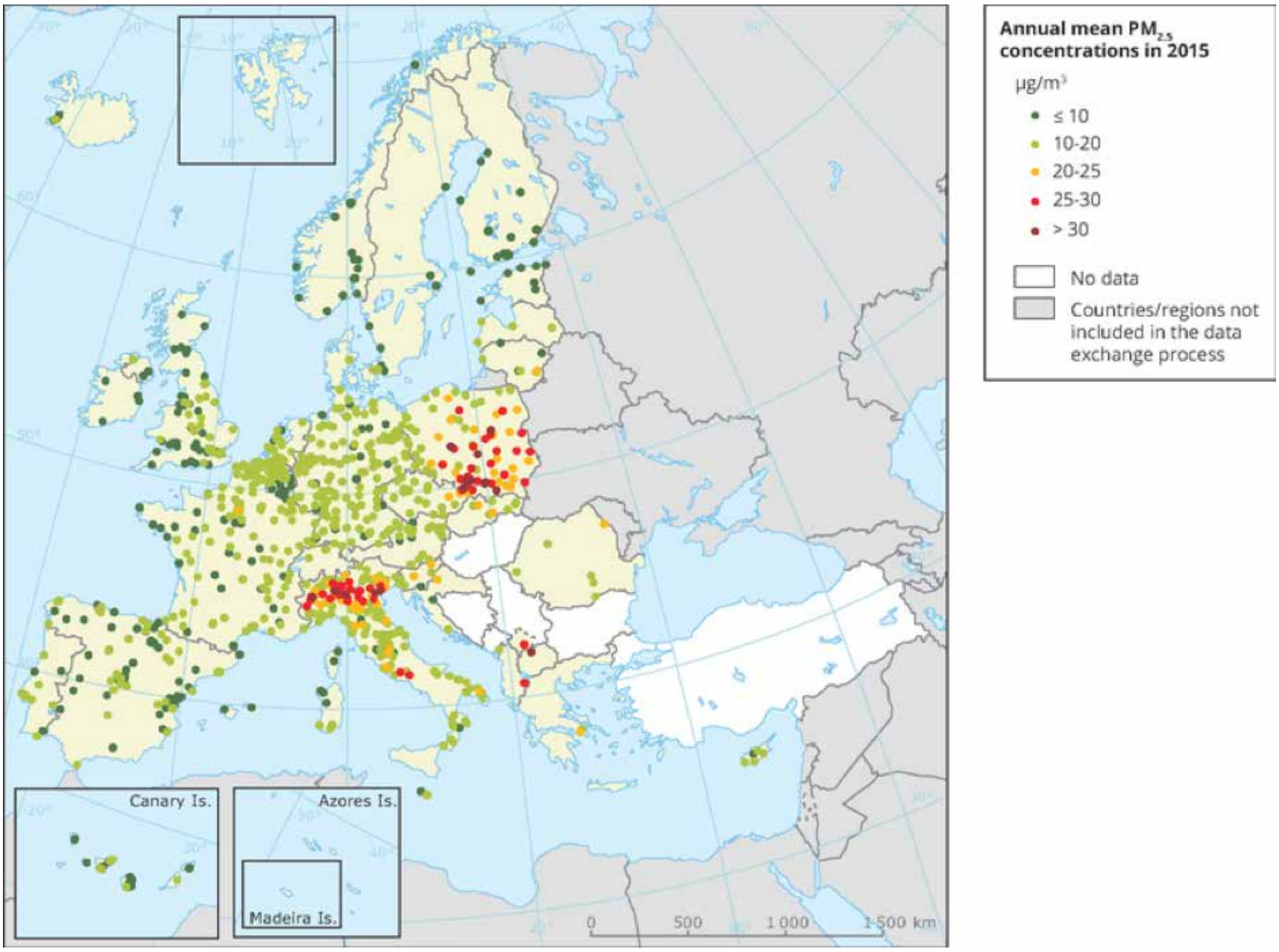

Figure 2: Annual mean $PM_{2.5}$ concentrations in 2015. Source: EEA (2017). https://www.eea.europa.eu/data-and-maps/figures/annual-mean-pm2-5-concentrations-2

Pollutant	Time period	Health outcome	Risk ratio
$PM_{2.5}$	Annual	Mortality (all-cause, age 30+)	1.062
PM_{10}	Annual	Prevalence of bronchitis in children, age 6–12 (or 6–18) years	1.08
PM_{10}	Annual	Incidence of chronic bronchitis in adults (age 18+ years)	1.117
O_3	8 hours	Mortality, all (natural) causes, all ages	1.0029
NO_2 (over 20 µg/m³)	Annual	Mortality, all (natural) causes, age 30+ years	1.055
NO_2	24 hours	Hospital admissions, respiratory diseases, all ages	1.018

Table 2: HRAPIE risk ratios (abbreviated) (after Héroux *et al.*, 2015) developed for application in analysis to support policy development in the European Union.

BOX 3.

Relative risk or risk ratio

In epidemiology, relative risk, or risk ratio, refers to the ratio of:
the probability of suffering the health effect (e.g. developing disease) in an exposed group, compared to:
the probability of suffering the health effect in a non-exposed group.

A relative risk of 1 suggests no difference between the groups tested; a risk ratio of 1.058 means the risk is 5.8% higher in the exposed group[7].

There have been a few critiques of air quality projection approaches. For example, regarding the source of the health data; the decision to use all-cause mortality instead of cause-specific functions; or the non-linear response function of PM mortality. The latter means that, as PM concentrations rise, the rate of increase in mortality declines, so projections of mortality following this model will be lower than if a linear relationship was assumed (Forouzanfar *et al.*, 2015). In linking health effects and mortality to pollution, studies also assume a time lag between the increase in pollution exposure and the impact.

Numerous studies quantifying health impacts associated with air pollution have been carried out, and there are now tools that automate air pollution health impact assessment (e.g. the BenMAP-CE and the DIDEM model (Ravina *et al.*, 2018)). However, Anenberg *et al.* (2016) describe the need for awareness of the assumptions built into such tools, which work on the basis of response functions, concentrations of pollutants and population exposure levels (see **Figure 3**). For example, Nedellec and Rabl (2016) evaluated the costs of health damage from atmospheric emissions of arsenic, cadmium, mercury and lead. Due to evidence from new epidemiological studies, the researchers found damage costs from these toxic metals to be much higher than previously thought. They therefore advise that new cost-benefit studies on abatement measures should incorporate these findings, highlighting the need for regular reappraisal of the evidence on which cost-benefit analysis is based.

7. Risk ratios: >1 suggests increased risk of outcome in exposed group; <1 suggests reduced risk of outcome in exposed group.

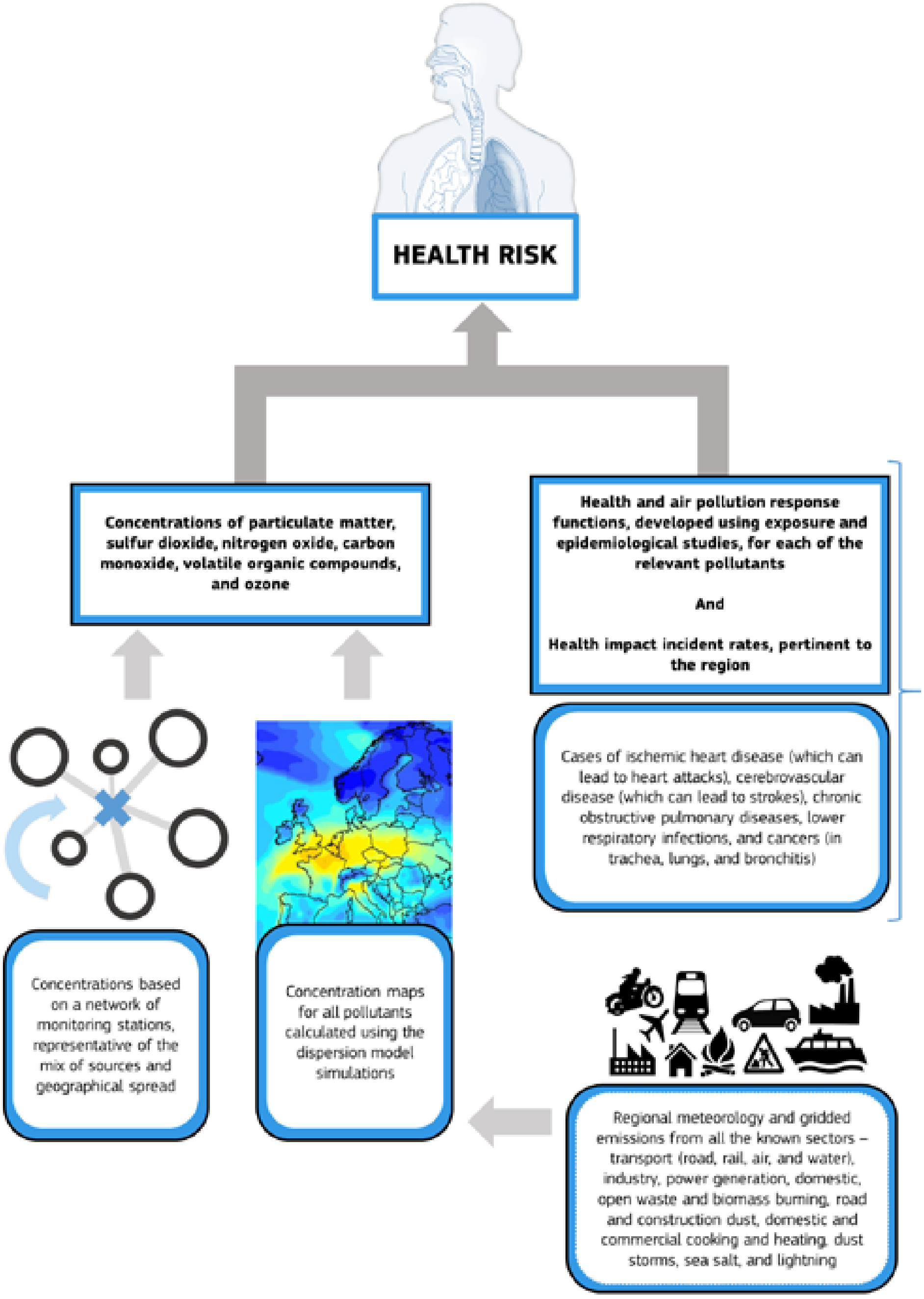

Figure 3: Typical data inputs used to assess health impacts of air pollution. Source: Anenberg *et al.* (2016).

2.4 Noise guidelines, thresholds and exposure limits

A 2002 European Commission position paper proposed dose-response curves that could be used to estimate the number of people annoyed by transport noise in their homes, and in turn inform the EU Directive on Environmental Noise (2002/49/EC). Since then, the WHO has published Night Noise Guidelines for Europe, recommending threshold values above which adverse health effects occur. The WHO working group found sufficient evidence that noise was related to self-reported sleep disturbance, self-reported health problems and insomnia, and that there was limited evidence for other effects as well (hypertension, myocardial infarctions, depression and others). Revised guidelines for noise in Europe are due to be published soon. In 2009, the evidence indicated that 40dB should be the target for night-time noise levels to protect the public, including the most vulnerable groups (WHO, 2009).

2.5 Chemical exposure and probability of health effects

In contrast to most air pollution health impact studies, the link between particular chemicals and health impacts has relied more often on **toxicology studies** carried out in the laboratory (e.g. Clarkson & Magos, 2008; Darnerud, 2003). Such studies use a control group against which to compare results, but are usually based on animal studies, lending some uncertainty to the transposition to human threshold doses (Bond & Dietrich, 2017). In **epidemiological studies**, effects are observed directly in humans — for example, looking at the risks of real-life exposure to air pollution (e.g. Thiering & Heinrich, 2015). The results of such real-life studies are subject to more uncertainty than a randomised controlled trial (as used in toxicology), however. It can be useful to use observations from both epidemiological and toxicological studies in conjunction, for example, to see if dose-response functions identified in animals are relevant to humans (Slama *et al.*, 2017).

When assessing the probability of the health effects due to chemical exposure, the quality and parameters of the data inputs is crucial; however, the evidence required for scientific consensus can take many years to gather.

3. How to put a value on life, health and illness

3.1 Health risk as an externality: market and non-market values

An external cost (externality) occurs as a consequence of an activity that affects a third party, who did not choose to incur that cost. For example, road vehicles, trains and aircraft are useful forms of transport, but also contribute to noise and air pollution. These impact on health — therefore health risk is an externality of transport, in economic terms.

Broadly speaking, there are two types of health-related costs due to pollution:

- **Market costs** can include losses in productivity due to illness (opportunity costs) and healthcare costs, e.g. the administrative cost of each hospital admission, use of technologies or pharmaceuticals etc (resource costs).
- **Non-market** or **welfare costs** can mean premature death and disutility (e.g. pain and stress) due to illness, or caring for others. These are increasingly being recognised as having significant costs to the economy, and can be given a monetary value through methods that evaluate individual preference among the public for allocation of resource (**see 3.2, 3.3, 3.5-3.10**). In assessment of pollution damage these values are, justifiably, usually much higher than market costs (Bickel *et al.*, 2006).

Both types of cost, in monetary terms, can be useful for policymakers to understand the benefit of interventions to reduce exposure to pollution. However, they can also be used in different ways in different contexts.

Healthcare economics and environmental economics vary somewhat in their aims. In healthcare, evaluation of the costs and impacts of interventions is necessary in order to prioritise

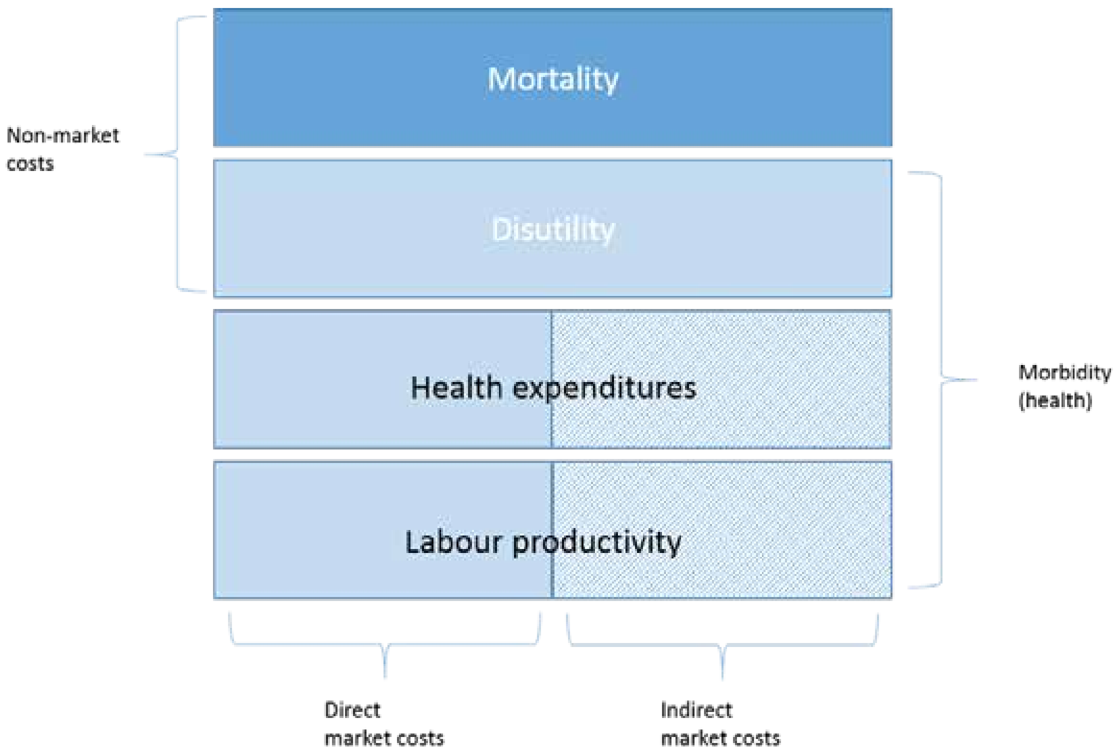

Figure 4: The broad types of costs associated with ill-health (morbidity) and premature death (mortality). Based on OECD, 2016 (p23).

BOX 4.
Glossary: key terms

Term	In this context:
Mortality	Death, at an earlier age than would have occurred in the absence of the health impact; premature death
Morbidity	Ill-health; the disease rate in a population
Disutility	The loss of the utility of good health, with respect to life expectancy, pain, suffering, distress or lost opportunity
Non-market cost	A cost to health that has no fixed monetary value, e.g. reduced lung capacity
Market cost	A health-related cost with a direct monetary value, e.g. time off work due to illness
Direct market costs	Costs directly related to the health impact on the individual, e.g. fees for healthcare and medicine; lost income
Indirect market costs	Costs caused by health impacts on the individual or population, which affect others or society, e.g. absence from work on the part of a carer; cost to an employer of replacing or covering the role of an absent worker
Health expenditures	Costs of medical care, e.g. ambulance call-out

resources. However, studies may use different approaches to demonstrate the effects of a treatment; for example, survival rates may be reported in one study and pain-free days in another. Survival rates alone give no information on the impact on quality of life; neither do data on pain and disutility alone indicate the full impacts of a health condition. Health economics is largely concerned with defining the cost-effective use of healthcare budgets, i.e. getting the 'most' good health from the defined amount of money available. The quality-adjusted life-year (QALY: see **Section 3.2**) and the disability-adjusted life-year (DALY: see **3.2**) have been routinely used as a summary measure in healthcare (Whitehead and Ali, 2010), using cost-effectiveness analysis (CEA) to place weights or valuations on different states of health. CEA places no monetary value on health outcomes, but rather provides a guide to maximising QALYs or DALYs with the available resources (Gray and Wilkinson, 2016).

Environmental economics is more often used when there is no defined budget. Environmental economics tends to concentrate on estimating the non-market costs (often assessed through the expressed preferences of the public) as a better indicator of how much society would value intervention, as opposed to only indicating the market cost savings that could be made. Combining market and non-market costs in an additive way needs to be done carefully to avoid double counting; however, some studies do include both (e.g. Chanel *et al.*, 2016). Crucially, these values are then used in a cost-benefit analysis (CBA) context, and not a cost-effectiveness context. In CBA, it is best to have both costs and benefits expressed in the same terms (e.g. in euros) to allow for direct comparison. While CBA is widely used in environmental (and transport) economics, CEA is the more usual way to value human health (Gray and Wilkinson, 2016).

Ascribing a monetary value to health outcomes (e.g. to death or disutility) caused by pollution helps to frame market costs in the same way as non-market factors, facilitating cost-effectiveness or cost-benefit analysis. In this way, the relative worth of different actions and policies can be evaluated, as can the trade-offs between the value of an economic activity and its associated health risks. The **cost of doing nothing** (the cost of inaction) can also be estimated, for future scenarios.

Monetising health impacts can inform measures aimed at **internalising costs**. As one example, imposing higher taxes on more polluting car engines internalises some of the health and environmental costs associated with them, and influences the extent of their use. Use of 'command-and-control' abatement measures, such as air quality standards, fuel efficiency standards and urban planning, may also draw on health cost figures as part of the internalised cost.

Finally, by monitoring actual levels of pollution and population health, the efficacy of abatement policies can be gauged in terms of **savings achieved**. A summary of the reasons for monetising health costs from environmental pollution is given in Box 5.

BOX 5.

Reasons for monetising health costs from pollution

- To communicate the burden of disease due to pollution.
- To measure the value of an activity and its associated health risks in a comparable way, allowing for better consideration of realistic trade-offs (comparing apples with apples and not with pears).
- To inform measures aimed at internalising costs (e.g. taxes and charges on polluting activities) and standards, or command and control measures.
- To permit calculation of health cost savings in alternative scenarios, or due to the implementation of policy that addresses pollution.

3.2 Quality-Adjusted Life Year (non-market valuation; often derived from stated preference)

A measure that captures both mortality and impacts on health (morbidity) has been developed: the quality-adjusted life year (QALY). This measure is used in economic evaluation in some countries, for example, the Netherlands, Sweden and the UK (Torbica *et al.*, 2015).

The QALY draws on surveys that ask people to rate or compare different health states. One such survey instrument is the EQ-5D questionnaire, developed by Euroqol[8], which is available in many languages (see Payakachat, 2015). This asks patients affected by a condition to rate different aspects of their health, for example their mobility, ability to carry out usual activities, pain and anxiety levels. The ratings can be converted into an index, or utility score, reflecting how conditions compare to each other in terms of preference. This scale usually runs from 0 (death) to 1 (perfect health).

A QALY is defined as a year of life spent in perfect health. QALYs experienced by patients with a condition are calculated by multiplying the utility score for a condition (as measured by a tool such as the EQ-5D) by the duration of time spent in a health state. For example, 10 years spent in perfect health (utility score 1) gives 10 QALYs. An individual receiving treatment for a condition, meanwhile, might be in a 0.7 health state for 10 years, giving 7 QALYs over the same period. Without treatment, they may have one year at 0.7 and 3 at 0.3, then 6 at 0.1, giving 2.2 QALYs over 10 years and demonstrating the value of treatment — see **Figure 5**. QALYs can also be discounted, taking into account that people prefer to receive health benefits now than in the future.

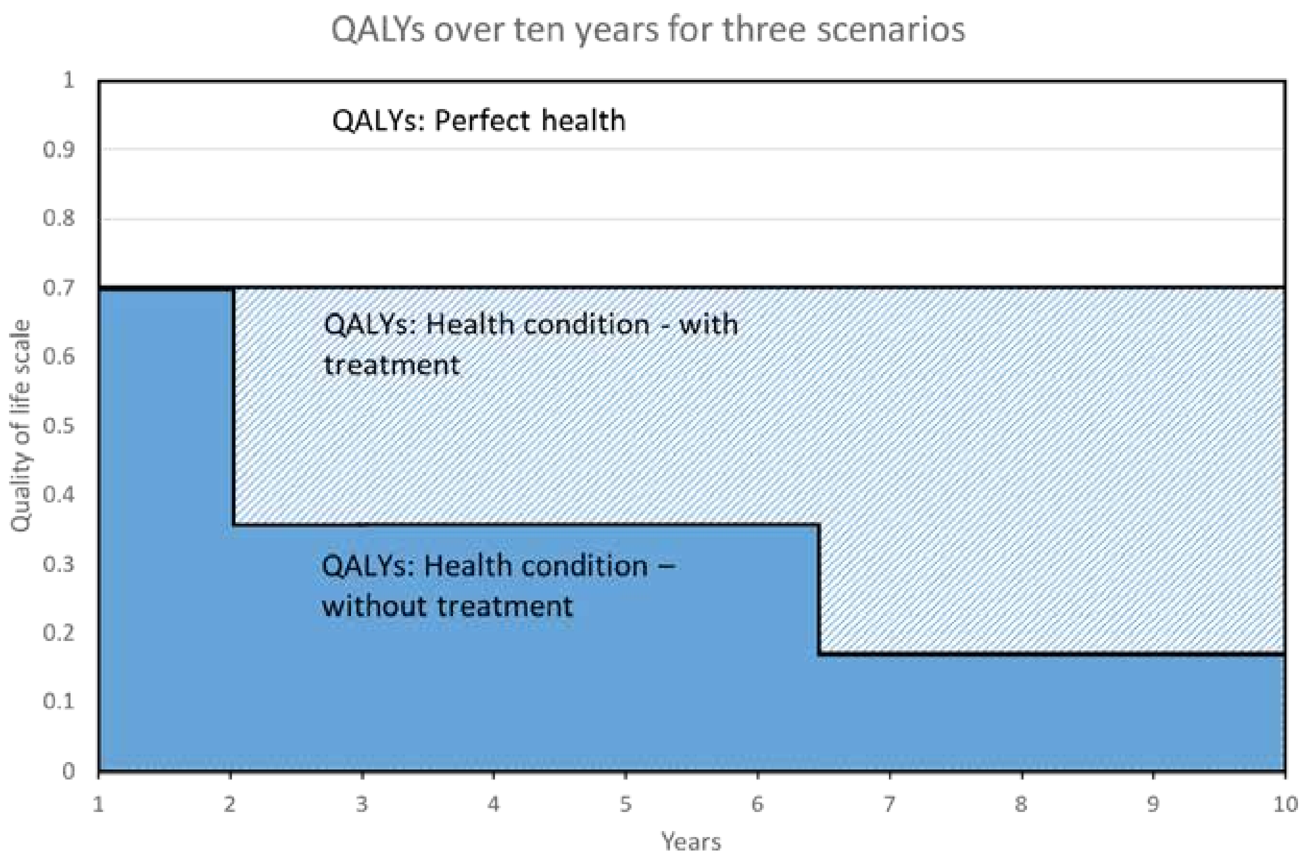

Figure 5: Example of comparison of health states using QALYs.

8. https://euroqol.org/

In this way, analysts can see the QALYs that can be gained from an intervention, as opposed to no intervention, or compare different types of intervention (or 'health technology'). In turn, a healthcare system can use QALYs in decisions about resource allocation. For example, an intervention could be seen as worthwhile if the cost per QALY gained is less than how much a QALY is valued by society.

Some health agencies (e.g. England's National Institute for Health and Care Excellence (NICE) and Sweden's health technology assessment agency, Tandvårds och läkemedelsförmånsverket (TLV)) use monetised QALYs to permit cost-benefit analysis (Mason *et al.*, 2009). In particular, cost-effectiveness of new drugs can be assessed by reference to QALYs, as performed by Australia's Pharmaceutical Benefits Advisory Committee, for example (Paris & Belloni, 2014). It is also possible to apply the QALY value to analysis of policy interventions (Schmitt, 2016).

To give an example, if the cost-per-QALY threshold is €100 000, then a surgical procedure that provides one QALY justifies a cost of €100 000; a procedure that offers five QALYs justifies an expense of €500 000. Medicine that provides a gain of 0.2 QALYs would justify an annual cost of €20 000 (€100 000 x 0.2).

Research has shown that people increase their valuation of a QALY depending on severity of health state (Shiroiwa *et al.*, 2013). Accordingly, in some countries — including Norway, Sweden and the Netherlands — the threshold QALY value considered (where an intervention is considered worthwhile) increases with the severity of the disease (Nord, 2017).

There are some ethical limitations to the QALY measure; for example, if QALYs are treated the same, no matter who accrues them. In the case that more people gain QALYs than people lose QALYS, then there are net QALY gains in the population overall — but this is not necessarily an ethical or moral assessment (Brazier *et al.,* 2017; McKie *et al.*, 2016). In addition, someone who stands to gain five QALYs probably does not perceive this as precisely half as valuable as a gain of 10 QALYs by somebody else.

Whether there is a moral or ethical justification for allocating more resources to the person who would gain more is still being debated by health economists; some have argued that QALYs should be weighted, for example, so that health gains for children are valued more (Donaldson *et al.*, 2011). In Norway, priority for new healthcare technologies is given to those who stand to lose more QALYs from a disease (Norwegian Ministry of Health, 2016), implicitly valuing younger lives more than those of older people. For the sake of societal equity, it may also be that some health gains need to be sacrificed to achieve greater distributional equity of health, for example, giving preference to lower socioeconomic groups.

The European Consortium in Healthcare Outcomes and Cost Benefit Research (ECHOUTCOME) carried out a study designed to test the robustness of monetised QALYs as used by the UK's NICE and officially does not endorse this method for evaluating new drugs (ECHOUTCOME, 2015). Based on a study of nearly 1 400 subjects in Belgium, France, Italy and the UK, the Consortium found that QALY measurement is inconsistent and does not reflect real behaviour patterns, therefore does not provide a scientific basis for decision-making. Instead, the Consortium propose five recommendations for conducting cost-effectiveness studies:

(i) clear distinctions between cost-benefit, cost-effectiveness and cost-utility analyses should be established;
(ii) QALY assessment for health decision-making should be abandoned;
(iii) cost-effectiveness analyses should be expressed as costs per relevant clinical outcome;
(iv) cost-effectiveness analyses should be validated by an interdisciplinary research team;
(v) cost-effectiveness analyses should use a tool box of various robust modelling techniques, to be selected on a case-by-case basis. (ECHOUTCOME, 2015)

The UK's National Institute for Health and Care Excellence (NICE) has made counterarguments that the QALY measure is useful and is being improved, but acknowledges that it need not be used in isolation.

3.3 The Disability-Adjusted Life Year

While QALYs are generally measured in 'gains' to express the positive effect of interventions, a similar measure has been developed that expresses the negative burden of disease: the disability-adjusted life year (DALY) (Murray, 1994; Murray and Lopez, 1994). The DALY combines the number of years lived with a disability and healthy years lost due to premature death, indicating the relative impact of illnesses and injuries on loss of healthy life years.

The basic calculation for a DALY associated with a disease or condition is:

DALY = YoLL + YLD

Where YoLL is 'years of life lost' from premature death (mortality) and YLD is 'years of healthy life lost due to disability' (morbidity).

Originally co-developed by the World Bank and WHO, the concept was adopted in the Global Burden of Disease studies (Mathers, 2017; see **Box 6**). The total loss of DALYs, combining years lost due to premature death and disability in the world population, is referred to as the 'global burden of disease'.

As explained by the WHO:

"One DALY can be thought of as one lost year of 'healthy' life. The sum of these DALYs across the population, or the burden of disease, can be thought of as a measurement of the gap between current health status and an ideal health situation where the entire population lives to an advanced age, free of disease and disability."
- WHO, http://www.who.int/healthinfo/global_burden_disease/metrics_daly/en/

BOX 6.

The Global Burden of Disease studies

Commissioned by the World Bank, the first Global Burden of Disease (GBD) study featured in the *World Development Report* 1993: *Investing in Health* (World Bank, 1993). Generating estimates for mortality and morbidity linked to 107 diseases and other health impacts, the publication had profound implications for health policy. For example, it recommended that competition in health services should be encouraged to improve quality and drive down costs of drugs, supplies and equipment.

The WHO subsequently produced a series of GBD estimates from 2000 onwards. Comprehensive updates, including new DALY estimates, were published in 2010 and 2015, this time co-ordinated by the Institute for Health Metrics and Evaluation at the University of Washington, Seattle.

The Global Burden of Diseases, Injuries and Risk Factors Study 2015 estimated the burden of disease due to 315 diseases and 79 risk factors, and analysed 249 causes of death in 195 countries from 1990 to 2015. A huge collaborative undertaking, the study involved 1870 experts in 127 countries. Along with smoking, high cholesterol and childhood undernutrition, **air pollution** was identified as a leading cause of global disease burden, particularly in low and middle-income countries (GBD 2015 Risk Factors Collaborators, 2016).

The DALY has become frequently used in low- and middle-income countries to assess health priorities and programmes, and the effectiveness of particular interventions, while the QALY is more often referred to in higher-income countries (Chen *et al.*, 2015; Rios-Diaz *et al.* 2016). A recent study investigated whether cost-effectiveness analysis for two vaccinations (against Human Papilloma Virus and pneumococcal infection) would produce different results depending on if it was based on the QALY or DALY measure (Augustovski *et al.,* 2017). They found that estimated QALY gains in HPV vaccination scenarios (compared to no vaccination) were greater than avoided DALYs, meaning that analysis based on QALYs would seem more cost-effective. There was no significant difference in QALY/DALY benefits for the other vaccine, however, DALYs avoided were generally more than QALYs gained through the vaccination. The study looked at scenarios in Argentina, Chile and the UK, which have different thresholds for cost-effectiveness. Differences between the QALY and DALY were largest in the UK for HPV, but there was no pattern for the other vaccine. It could be interesting to see this type of study extended to policy interventions relevant to environment and health.

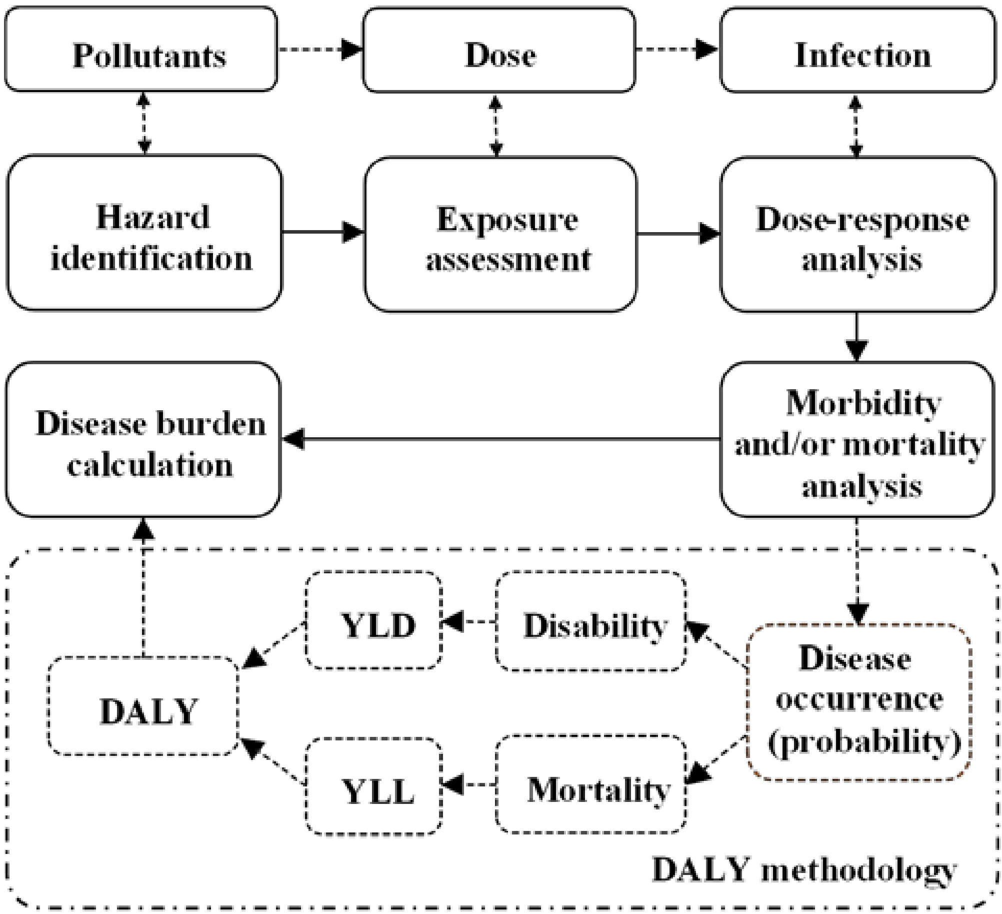

Figure 6: Framework of environmental burden of disease study with usage of DALY. Source: Gao *et al.* (2015b).

Some studies have used DALYs to quantify disease burden due to environmental pollution, based on the procedure shown in **Figure 6**. For example, Cohen *et al.* (2017) used remote sensing, ground-based measurements and chemical transport models to calculate global mean concentrations of $PM_{2.5}$ and ozone. They linked these concentrations to risk of mortality from four cardiovascular and respiratory system diseases, and lung cancer. Globally, they found that exposure to $PM_{2.5}$ was the fifth-ranking mortality risk factor in 2015, and caused 103.1 million DALYs. Ozone, meanwhile, was linked to the loss of 4.1 million DALYs from chronic obstructive pulmonary disease. **Figure 7** illustrates the burden of disease due to air pollution, by country, in terms of DALYs.

The DALY can also be used to compare impacts among different population groups — for instance, in a case study looking at reduction in $PM_{2.5}$ air pollution, Martenies *et al.* (2015) found that the 60-64 year-old age groups would benefit most. Gao *et al.* (2015b) note that standardised methodologies are needed to transform pollution data into disease data, using the DALY framework as a tool for quantitative assessment of environmental pollution. Indeed, since the equations and life expectancies used to estimate DALYs in the GBD have changed since its inception, it is difficult to compare the results of past studies and more recent updates (Chen *et al.*, 2015).

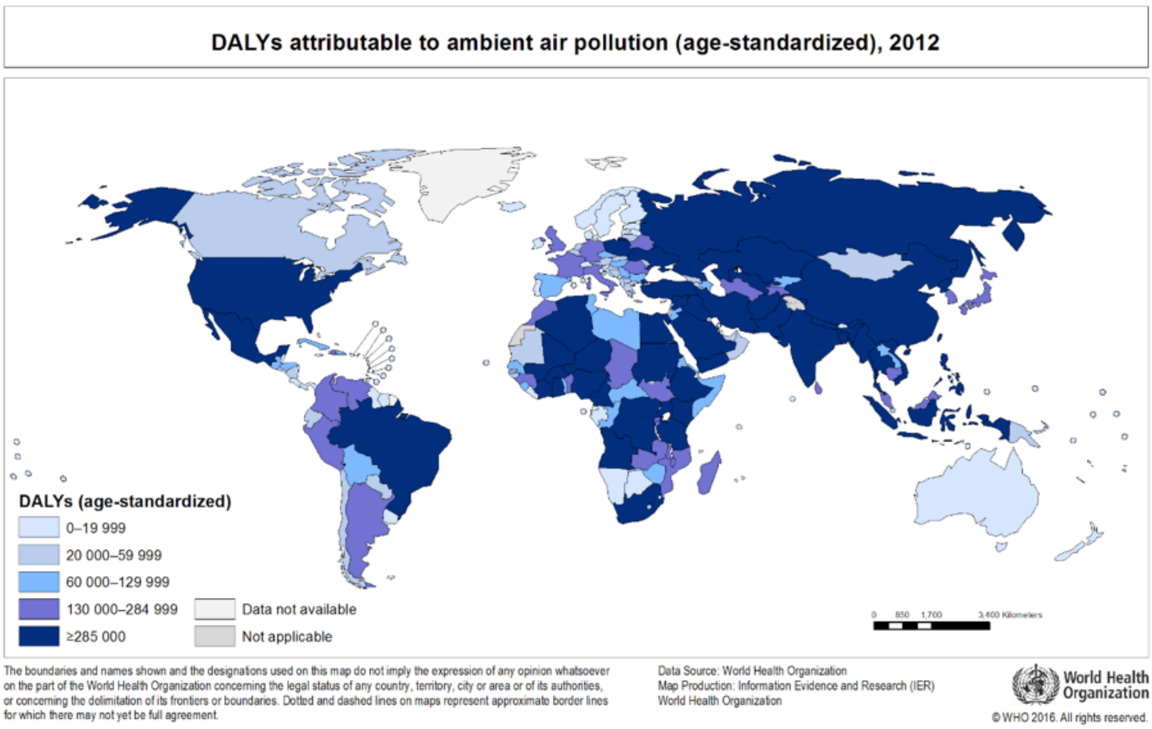

Figure 7: DALYs attributable to ambient air pollution (WHO, 2016).
http://gamapserver.who.int/mapLibrary/Files/Maps/Global_aap_dalys_age_standardized_2012.png

In the DALY framework, different life expectancies can be used for different age groups or regions. The 1990 GBD used life expectancies of 82.5 years (women) and 80 (men), based on rates in Japan, while the 2010 GBD studies used a global standard of 86 years. Under the latter assumption, a person who dies at 80 contributes 6 DALYs; however, a person who lives to 90 does not detract from the sum of DALYs. This means that the value of any intervention that contributed to the longer lifespan is not considered in the metric.

The calculation of the YLD also depends on the disability weighting attributed to particular conditions. For GBD 2010, surveys were conducted around the world asking participants to rate 'health loss' related to different conditions, allowing disability weights to be developed. Many criticisms have been levelled against this approach, for example, questioning whether health status can be rated independently of social context or vulnerability. Sight impairment for someone in a high-income, urban area with plentiful assistive technologies is differently experienced to someone in a low-income rural community, for instance, raising doubts over whether these ratings can be universalised. According to Chen *et al.* (2015), studies of disability rankings from different countries have shown consistency, but in DALY methodology, either regional values or those from the GBD may be used for YLD, highlighting the non-standard nature of DALY calculation. Age weighting — where higher DALYs are attributed to younger people affected by disease to reflect the higher loss in productivity compared to older people — was used in the earlier GBD studies but omitted in the 2010 GBD. This was described by the authors as a simplified method, but in fact addressed the difficulty in justifying unequal valuation based on age.

Despite this simplification, the DALY method is complex, requiring inputs about population age structure, life expectancy and cases of disease. It is important to note that, since the methodology used by the GBD has changed over time, and the methodology is not standardised, any reports using the DALY must transparently state the variables used (e.g. life expectancy and age weighting).

3.4 Cost of illness (market-based valuation)

Calculating the cost of illness (COI) brings together all the actual monetary costs related to an illness, including direct costs of healthcare and indirect costs, for example, income lost through time spent off work (also known as opportunity costs). It does not usually take into account non-market, disutility costs such as pain and suffering (Jo, 2014)[9]; the main cost related to a premature death, therefore, would be lost productivity. Stress and tiredness caused by noise would also lead to lower performance and productivity at work.

Lost productivity may be calculated by reference to average wages or Gross National Product (GNP) per capita, with future earnings discounted to give a present value (the human capital approach). This approach can be adapted so that the lives of children and retired people are not valued at zero. Another approach — the friction cost method — also takes into account the time and cost to employers of replacing sick staff (the 'friction period'). If a family member has to take time off work, this indirect cost can also be considered.

Using the COI approach, a partial estimate of the benefits of interventions can be expressed in terms of the money saved by the number of cases of illness

9. COI studies sometimes encompass willingness-to-pay methodology to calculate indirect costs, however this will be considered in the next chapter.

(and associated costs) avoided, either in a fixed period or over a lifetime. This must be based on a benchmark, for example, based on emissions levels and concentration-response functions. Although subject to a degree of error in calculation, COI can be seen as an objective metric, and is widely used in health economics to indicate how much society is explicitly spending on a disease. However, such studies do not indicate how much it would cost to prevent the illness, nor how much society would be willing to pay to do so. Neither is there an ethical justification that more costly diseases/health impacts should be allocated more resources.

One of the main shortcomings of pure COI approaches is that they fail to incorporate the entire scope of economic costs associated with illness (for example, psychological or other intangible or indirect or difficult-to-assess costs). Trasande finds, therefore, that COI estimates (of EDCs) must be considered underestimates (2015).

3.5 Revealed preference methods (non-market-based valuation)

Revealed preference methods look at how individuals actually behave in the market: for example, how much is spent on indoor air filters gives an indication of how much individuals value avoiding indoor air pollution.

'Hedonic pricing' (where both intrinsic and contextual factors are used to estimate values), is a commonly used type of revealed preference methodology. A type of hedonic pricing that looks at property prices near sources of noise such as airports has often been used to indicate the value of environmental noise reduction (e.g. He *et al.*, 2014), resulting in the Noise Sensitivity Depreciation Index (NSDI) — a measure of the percentage change in price due to unit change in noise level. However, this does not necessarily capture any of the health costs associated with noise exposure — only how much money people would accept in order to live with the annoyance or would pay in order to avoid the annoyance. Hedonic pricing has been broadly accepted as the standard method for measuring noise annoyance; however, meta-analyses have found wide and unexplained variation in NSDI values (Bristow *et al.*, 2015).

3.6 Environmentally attributable fraction (non-market-based valuation)

This fraction is defined by Smith *et al.*, (1999) as "the percentage of a particular disease category that would be eliminated if environmental risk factors were reduced to their lowest feasible levels." The environmentally attributable fraction (EAF) is a composite value and is the product of the incidence of a risk factor multiplied by the relative risk of disease associated with that risk factor. Its calculation has mainly been used as a tool in developing strategies for resource allocation and prioritisation in public health, but has also been used to assess the costs of environmental and occupational disease (Landrigan *et al.*, 2002).

Landrigan *et al.* (2002) use the calculation:

Costs = Disease rate x EAF x Population size x Cost per case

Here, 'cost per case' refers to discounted lifetime expenditures attributable to a particular disease, including direct costs of health care, costs of rehabilitation, and lost productivity — hence this is a market-derived measure.

3.7 Willingness-to-pay (non-market valuation; stated preference)

Where no market value exists for health effects (e.g. pain or stress), stated preference methods can be used to assign a monetary value that can be used in CBA.

Willingness–to-pay and willingness-to-accept surveys — known as 'contingent valuation' — are classed as stated preference methodologies as they explicitly elicit statements of preference. In the context of valuing health risks from environmental pollution, **willingness-to-pay** (WTP) approaches are usually based on questionnaires to find out how much individuals value health and/or longevity (although they can be based on hedonic pricing measures as well). The values do not suggest that people would be willing to trade their health or life for the stated sum, only that certain actions related to increasing/decreasing health risk are preferable to others.

For example, participants might be asked how much they would pay to avoid an increase in risk of dying or falling ill. It has been found that willingness to pay for environmental public goods can depend substantially on variables such as income inequalities (Baumgärtner *et al.*, 2017), or 'sense of place' (i.e. an individual's attitude towards a geographical setting; Nielsen-Pincus *et al.*, 2017), and there has also developed a broad literature on the avoidance of bias in contingent valuation studies (Loomis, 2014). Combes *et al.* (2018) conducted a study investigating the effect of contextual and individual factors on the likelihood of individuals' willingness to pay (any amount) to prevent environmental pollution, using data from the World Values Survey 2005-2008. It is assumed in the study that aspects that may positively influence individuals' willingness to pay may include higher: income, awareness of environmental pollution and its negative impacts on health, population density, public spending, and higher standards and more solid institutional structures. They found that a substantial proportion of country variation can be explained by individual characteristics (80% in developed countries; 90% in developing countries) — and that higher levels of education, income, 'post-materialist values', religion and membership of environmental organisation were consistent determinants in WTP to prevent environmental pollution. They also found evidence that democracy and government stability are negatively correlated with the intention to pay to reduce environmental pollution — although these findings mainly apply to developing countries.

3.8. Willingness to accept (non-market valuation; stated preference)

Willingness-to-accept (WTA) is an alternative method that elicits the amount of money an individual would accept to tolerate an increase in health risk from pollution. In principle, WTA may be a better approximation of the worth of pollution abatement, as it assumes the public should not be expected to pay to prevent the externalities caused by industrial pollution (Breffle *et al.*, 2015). However, because WTA is not bound by income, WTA surveys can result in large values (Kahneman and Taversky, 1979; Whittington *et al.*, 2017), though the discrepancies between WTP and WTA studies cannot always be explained by

income effects. Whittington *et al.* (2017) offer recommendations on where WTA surveys are appropriate for informing policy decisions, but in practice, WTP is preferred, even though it may underestimate the value of health to society.

3.9 Value of a statistical life (VSL) (non-market valuation; often derived from stated preference)

The value of a statistical life (VSL) is a commonly used economic method of valuing risk to life. VSL is derived from the trade-offs people are willing to make between fatality risk and wealth — it might be alternately phrased 'the value of preventing a fatality'. VSLs are based on the fairly robust theory of compensating differentials—the idea that workers must be paid more to take on tasks that are unpleasant or hazardous. Reservations remain over the use of these figures, because the methods may reflect ability to pay, and hence be discriminatory against poorer people. However, willingness-to-pay methods can be made sensitive to income distribution by using appropriate income-sensitive distributional weight (Laxminarayan *et al.*, 2014).

It is usually derived from aggregating expressed WTP to marginally reduce the risk of premature death. It does not directly represent the value of a particular life, but rather the sum of WTP values for a reduction in the risk of premature death. For example, the average WTP for a 1 in 100 000 reduction in annual risk of dying from air pollution might be €50. Therefore, one death per 100 000 would be prevented by 100 000 people paying €50. The sum of 100 000 WTP values in this case is €5 million — which is the VSL.

The value of VSL is not constant; it is related to income and rises over time with inflation (Viscusi, 2010). In addition, some studies have found that WTP for risk reduction peaks in mid-age, while it has also been suggested that children's VSL should be weighted higher than adults' (Aldy and Viscusi, 2007; Lindheim *et al.*, 2011). However, the suggestion of officially discounting the VSL of older people met with public outcry in the USA (Washington Post, 2003) and the US EPA does not support the use of different values for mortality risk reduction based on age (NCEE, 2010). As the metric is based on WTP, it does not incorporate economic costs such as lost productivity due to premature death, nor does it offer any value for morbidity.

VSL estimates are widely used to monetise fatality risk in CBA, for example, in road safety evaluation and, in the USA, in a CBA relating to air pollution, where the recommended VSL was US$7.4 million (€6.3 million) (IEc, 2010). The OECD has suggested a VSL for OECD countries of US$3 million (€2.6 million) and US$3.6 million (€3 million, 2005 prices) for EU countries (OECD, 2016); this value is referenced in the Commission's Better Regulation Guidelines as a good estimate.

Some studies on the health impacts of environmental pollution have not focused on VSL, which only values mortality, but rather on changes in average life expectancy due to air pollution. Desaigues *et al.* (2011), for example, argue that the monetary 'value of a life year' is a more useful measure; however, the OECD use the VSL and not the VOLY.

3.10 Value of a life year (VOLY) (non-market valuation; often derived from VSL or stated preference)

While the VSL is useful for accounting for lost lives, it is perhaps most appropriate when applied to lives lost in youth or middle age. For example, the average victim of a traffic accident stands to lose 35–40 years of life expectancy, yet fatalities due to air pollution tend to be older people in their 70s or 80s, with existing lung or heart problems. Accounting for life years lost (VOLY), as opposed to looking at willingness to pay for risk reduction (the VSL), is therefore an alternative measure that takes into account the number of life years at risk. The value of a life year (VOLY) has been used to cost health impacts from pollution in Europe (Desaigues *et al.*, 2011; Alberini *et al.*, 2006; Chilton, 2004; Holland *et al.*, 1998).

The cost of pollution in life years lost can be calculated by multiplying the remaining life expectancy of a person by the value of a life year (VOLY).

The VOLY can be proportional to the VSL, or can be calculated independently. For example, Desaigues *et al.* (2011) used contingent valuation to determine a VOLY for Europe, through surveying 1463 people in nine countries: France, Spain, UK, Denmark, Germany, Switzerland, Czech Republic, Hungary, and Poland. Part of the NEEDS project, the survey asked how much people would be willing to pay for reduction in air pollution which would lengthen their life by three or six months. A European VOLY of €40 000 was recommended by one of the researchers (Desaigues *et al.*, 2011). VOLY estimations also varied somewhat by country, for example, being higher in Switzerland and Denmark, and lowest in Poland and Hungary, perhaps reflecting the difference in average incomes. Life expectancy can also change the findings from using VOLY.

4. Examples: measuring benefits to health

To follow are some examples where environmental pollution has been monetised, using some of the non-market methodologies in **Section 3**.

4.1 Quality-Adjusted Life Years

The monetary value of a QALY may be based on willingness-to-pay surveys (see **3.7**) or other methods (Tilling *et al.*, 2016). The values derived are country-specific and not fixed (Vemer & Rutten-van Mölken, 2011). For example, the value per QALY recommended in the UK, based on interviews with 3 400 participants, is £60 000 (€66 500) in 2009 prices (Glover and Henderson, 2010) or £65 000 (€73 000) in 2013 prices (Schmitt, 2016). In the US, $150 000 (€125 000) is suggested (Neumann *et al.*, 2014). A recent Swedish study valued a QALY at €300 000, based on interviews with 800 participants (Olofsson *et al.*, 2016), and researchers in the Netherlands obtained values of €80 000 to €110 000 with a sample of just over 1 000 participants (Bobinac *et al.*, 2014).

Some health agencies (e.g. England's National Institute for Health and Clinical Excellence and Sweden's health technology assessment agency, Tandvårds och läkemedelsförmånsverket, TLV) use monetised QALYs to permit cost-benefit analysis (Mason *et al.*, 2009). In particular, cost-effectiveness of new drugs can be assessed by reference to QALYs, as performed by Australia's Pharmaceutical Benefits Advisory Committee, for example (Paris & Belloni, 2014). It is also possible to apply the QALY value to analysis of policy interventions (Schmitt, 2016; see **Box** 7).

To give an example, if the cost-per-QALY threshold is €100 000, then a surgical procedure that provides one QALY justifies a cost of €100 000; a procedure that offers five QALYs justifies an expense of €500 000. Medicine that provides a gain of 0.2 QALYs would justify an annual cost of €20 000 (€100 000 x 0.2).

Research has shown that people increase their valuation of a QALY depending on severity of health state (Shiroiwa *et al.*, 2013). Accordingly, in some countries — including Norway, Sweden and the Netherlands — the threshold QALY value considered (where an intervention is considered worthwhile) increases with the severity of the disease (Nord, 2017).

4.2 Disability-Adjusted Life Years

DALYs indicate the relative impact of illness and injury on loss of healthy life years. Some studies use the disease burden due to pollution quantified in DALYs, applying a monetised value to the DALY to value the health impact. For example, Meisner *et al.* (2015) valued DALYs according to a combination of the human capital approach, COI and VSL to show that air pollution in Macedonia resulted in costs of €253 million annually, with premature deaths accounting for 90% of this amount. Similarly, in Mumbai and Delhi, India, DALYs for air pollution impacts, taken from previous studies, were monetised by reference to the VSL for India ($94 721) and European WTP values for morbidity, adjusted by reference to Indian per capita income and purchasing power (Maji *et al.*, 2017). The annual cost of air pollution in those cities was thus estimated at US$4 269.60 million (€3 610.3 million) and US$6 394.74 million (€5 407.33 million), respectively, for the year 2015.

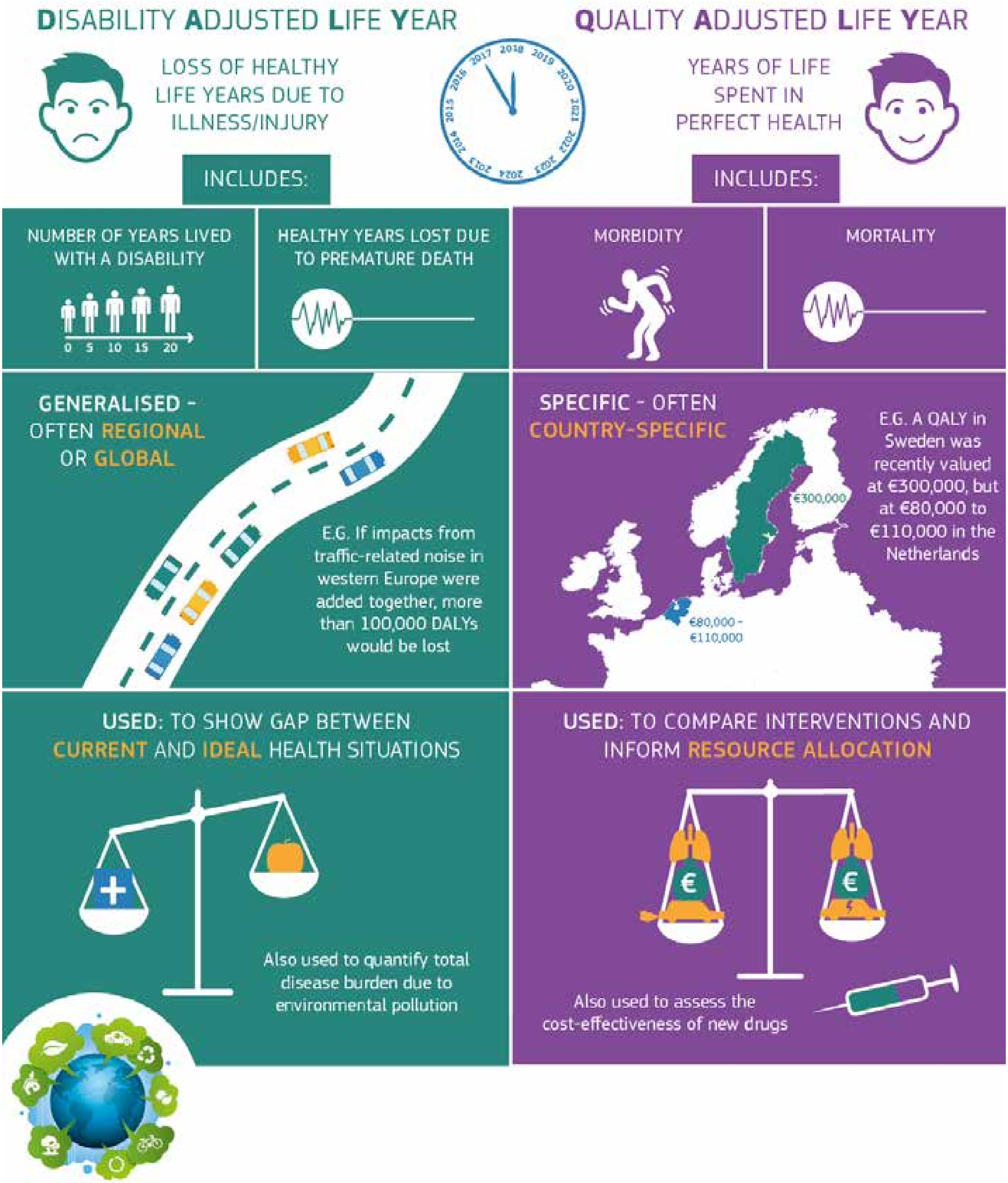

Figure 8: Infographic, Ways to cost environmental pollution: DALYs and QALYs explained.

BOX 7.

Savings from reducing air pollution in England and Wales, based on QALYS

A UK-based study looking at the benefits of reducing air pollution in London, as well as in England and Wales, used monetised QALYs to express the savings possible (Schmitt, 2016). The study considered the health states of individuals with three diseases related to or exacerbated by air pollution: chronic obstructive pulmonary disease, coronary heart disease and lung cancer. The study modelled the progress of the individuals' health from age 40-90, in the context of a 1 μg/m^3 reduction in $PM_{2.5}$ concentration — equivalent to a 7% reduction from current levels in London and a 9% reduction in England and Wales.

The benefits of the reduction in air pollution, over a 60-year period, were calculated at £4 billion (€4.5 billion) in London and £34 billion (€38.4 billion) in England and Wales. These figures were arrived at by using a QALY value of £65 000 (€73 400), multiplied by gains of 63 000 and 540 000 QALYs respectively, minus increased health costs associated with extending the lives of individuals with chronic cardiac or respiratory conditions. A healthcare service cost of £13 000 (€14 700) per QALY gain was used in this calculation.

A report published in the UK in 2014, Environmental Noise: Valuing impacts on: sleep disturbance, annoyance, hypertension, productivity and quiet, provided an example of the health impact of sleep disturbance in DALYs — and recommended moving to DALYs from the previous hedonic approach (Defra, 2014). For example, the value of sleep disturbance is calculated as:

Population exposed x proportion sleep disturbed x disability weight x health value (QALY)

In this case, the WHO Night Noise Guidelines for Europe recommend a disability weighting of 0.07, while the UK Department of Health sets the value of a QALY at £60 000 (€66 500). In this way, it was calculated that a household's increase in average dB exposure from 50.2 dB to 51.1 dB of night time noise would cost £39.66 (€44.52) per annum, or £62.92 (€70.62) for an increase from 59.2 to 60.1 dB. Separate values can also be calculated depending on the source of the noise (road, rail or aircraft).

The WHO puts noise pollution second only to air pollution in terms of health damage, based on WTP derived from a meta-analysis with at least one million healthy life years lost each year due to traffic-related noise in western Europe (WHO, 2011). DALYs lost from environmental noise in the EU countries were also estimated: 60 000 years for ischaemic heart disease, 45 000 years for cognitive impairment of children, 903 000 years for sleep disturbance, 21 000 years for tinnitus and 654 000 years for annoyance. Sleep disturbance and annoyance mostly related to road traffic noise comprise the main burdens of environmental noise in western Europe. If all of these impacts were considered together, the researchers estimate the total would be 1.0–1.6 million DALYs. Even with the most conservative assumptions that avoid any possible duplication, the total burden of health effects from environmental noise would be greater than one million years in western Europe (WHO, 2011).

4.3 Revealed preference

Revealed preference methods have not been widely applied to pollution-related health impacts beyond those that are noise-related. This hampers the shared use of data from noise and other studies (Istamto *et al.*, 2014), hence the emphasis for valuing noise health impacts is shifting to stated preference methods (Bristow *et al.*, 2015).

4.4 Environmentally attributable fraction

In the US, Landrigan *et al.* (2002) estimated the costs of lead poisoning, childhood cancer, developmental disabilities, and asthma at $54.9 billion (€46 billion) — a cost attributable to 'toxins', i.e. exposure to lead, methylmercury, pesticides and air pollution — using an 'environmentally attributable fraction' model. A 2015 report put the health cost in the EU of endocrine-disrupting chemicals at €157 billion, using the an approach based on environmentally attributable fraction, even while only counting those endocrine-disrupting chemicals[10] with greatest probability of causation; the authors note that a broader analysis probably would have produced greater burden of disease and cost estimates (Trasande *et al.*, 2015).

They use a 'weight of evidence' approach, adapted from the model used by the Intergovernmental Panel on Climate Change (IPCC, 2005), to attribute cases of illness to exposure. To evaluate the strength of the dose-response function for endocrine-disrupting chemicals[11], the study asked a panel of experts to participate in a modified Delphi approach — a type of forecasting methodology that has been used successfully in science, technology, health and education since the 1950s (Rescher, 1997). The experts agreed that endocrine-disrupting chemicals were 'at least probably' responsible for intellectual disability, autism, attention deficit hyperactivity disorder, childhood and adult obesity, adult diabetes, cryptorchidism, male infertility, and mortality associated with reduced thyroid function. They also estimated a strong probability that 13 million IQ points are lost each year in the EU due to exposure of foetuses to organophosphates, at a cost of €146 billion to the EU economy. They found a substantial probability of very high disease costs across the life span associated with endocrine-disrupting chemical exposure in the EU, although the study also attracted some critics [12].

4.5 Willingness to pay

Chanel *et al.* (2016) drew on WTP studies to calculate the costs of chronic disease caused by living near to road traffic pollution. For example, the cost of childhood asthma was valued at €1630 per year, based on the findings of two previous studies that asked how much parents would be willing to pay, annually, to control their child's symptoms. This value was added to market-valued costs to give a total annual cost of €3052 per asthma patient. The study estimated that 33 200 children across ten European cities could have developed asthma due to living close to busy roads, therefore this cause of chronic illness represents a 'hidden economic burden' of traffic

10. Several actions are being taken with regard to endocrine-disrupting chemicals in the EU, which are summarised here: http://ec.europa.eu/environment/chemicals/endocrine/index_en.htm

11. The study did not generally consider endocrine-disruting chemicals that are already banned in Europe under the Stockholm Convention, unless their global use could still contribute to health effects in Europe. For example, a chemical used in malaria control, DDE, persists in the environment and is transported long distances, therefore is still a risk factor though not used in Europe.

12. From e.g. Middelbeek & Veuger (2015) and Bond & Dietrich (2017).

pollution, valued at €101.33 million per year (2005 values).

Globally, the cost of aviation-related noise has been valued at $23 billion (€20 billion) (He *et al.*, 2014), based on WTP derived from a meta-analysis of hedonic pricing studies. Another study estimated the perceived values of traffic-related air pollution and noise health risks in five European countries through WTP (Istamto *et al.*, 2014), as part of the EU-funded INTARESE (Integrated Assessment of Health Risks from Environmental Stressors in Europe) project. Participants were first offered information on general health risks, specific health risks and combined health risks linked to a pollutant, before being asked to complete a survey on their WTP to reduce or avoid certain health risks. Averaged results from the study, conducted in Germany, Finland, the Netherlands, Spain and the UK, with over 10 000 participants, are shown in Table 3. Information about the participants was also collected, to allow the researchers to analyse links between responses and demographic characteristics, individual health, risk perception and attitudes, for example.

WTP question	Mean value	Median value
Related to road-traffic air pollution		
To reduce general health risk	€130 per person per year (pp/y)	€40 pp/y
To avoid shortening life expectancy by half a year	€80 pp/y	€10 pp/y
To decrease road-traffic air pollution by 50%	€330 pp/y	€50 pp/y
Related to road-traffic noise effects		
To reduce general health risks	€90 pp/y	€20 pp/y
To avoid a 13% increase in severe annoyance	€100 pp/y	€20 pp/y
To avoid a combined-risk scenario related to an increase of a noise level from 50 dB to 65 dB	€320 pp/y	€50 pp/y

Table 3: Findings of WTP study by Istamto *et al.* (2014).

Predicted increases in health-related costs due to air pollution, globally, under BAU scenario (based on OECD, 2016)		
	2015	**2060**
Healthcare costs	US$ 21 billion (€18 billion)	US$ 176 billion (€150 billion)
Working days lost	1.2 billion	3.7 billion
Welfare costs (based on willingness-to-pay to reduce risk of premature death)	US$ 3 trillion (€2.6 trillion)	US$ 2.2 trillion (€1.9 trillion)

Table 4: Projected health impacts due to air pollution

Based on their findings, the researchers urge caution when transferring results between countries, as risk perceptions and attitudes varied widely, affecting WTP estimates. For example, Dutch respondents were the least concerned about pollution levels, while Spanish respondents were five to seven times more concerned about road-traffic related pollution than others. Finnish people, meanwhile, expressed the highest WTP to reduce noise pollution. The researchers found that respondents who were concerned about the environment, or sensitive to air and noise pollution, were willing to pay more to reduce their health risks.

Interestingly, the average WTP for a 6-month gain in life expectancy found in this study (€80) was about four times lower than that found in the NEEDS project (Desaigues *et al.*, 2011), which was conducted across nine European countries. The researchers attribute this partly to differing methodology. The INTARESE study was open-ended, allowing respondents to choose an amount; the NEEDS project used the payment card method, where answers are based on a multiple choice of amounts. Open-ended methods tend to elicit lower values. There may also be a discrepancy due to different countries being involved in each study.

Conversely, the HEATCO study (Bickel *et al.* 2006; Navrud *et al.*, 2006), which also used the payment card approach, found lower mean and median WTP to reduce noise annoyance, compared to INTARESE. The two studies are difficult to directly compare, however; the HEATCO study chiefly surveyed people already affected by ambient noise.

In 2016, the OECD published a stark warning of the costs of air pollution under a business-as-usual scenario continuing until 2060 (**Table 4**). The Economic Consequences of Outdoor Air Pollution addresses the impacts of PM and ground level ozone on mortality and health, as well as crop yields. (Health impacts of NO_2 are not addressed due to insufficient reliable data at the global scale). In quantifying the costs of inaction, the report provides a benchmark for evaluation of policy action that leads to change. The report measures welfare costs, also known as non-market health impacts, in monetary terms, based on WTP studies.

A list of reference WTP values for nine health outcomes has also been developed by the European Chemicals Agency (ECHA), to guide EU Member States in preparing restrictions on the use of chemicals (ECHA, 2017). For example, a minor birth defect linked to chemical exposure is valued at €4 300 to €41 800 per case; mild, acute dermatitis at €227 per two-week case; and cancer (type unspecified) at €410 000 per case. These values were based on research conducted in four Member States[13] in 2012. It is noted that further valuation studies would be needed to assess the validity of some of the values found.

4.6 Value of a Statistical Life

Based on the overall mean value of all stated preference studies in the OECD's metaanalysis, **Table 5** shows a VSL estimate for OECD countries of about US$ 3 million, which means that 50% of the mean estimates from OECD countries are below 3 million, and 50% are above 3 million. For the EU-27, the corresponding VSL estimate is 3.6 million.

	Full sample	Trimmed sample	Quality-screen sample	OECD countries (screened)	EU-27 countries (screened)
Mean VSL (standard error)	6 064 679 (490 985)	4 959 587 (315 688)	2 792 963 (169 443)	4 007 900 (229 931)	4 704 038 (329 474)
Weighted mean VSL (standard error)	7 415 484 (885 235)	6 314 696 (301 182)	2 123 538 (255 835)	3 981 851 (289 793)	4 893 216 (439 370)
Median	2 377 592	2 377 592	1 680 571	3 012 558	3 614 506
Observations	856	814	405	261	163

Table 5: Summary of the estimates of value of statistical life (VSL) 2005-USD.
Source: OECD (2012) Risk Valuation in Environment, Health and Transport Policies Chapter: Recommended value of a statistical life numbers for policy.
https://read.oecd-ilibrary.org/environment/mortality-risk-valuation-in-environment-health-and-transport-policies/recommended-value-of-a-statistical-life-numbers-for-policy-analysis_9789264130807-9-en#page5 https://www.oecd-ilibrary.org/environment/mortality-risk-valuation-in-environment-health-and-transport-policies_9789264130807-en

13. Czech Republic, Italy, Netherlands, UK

BOX 8.

Health costs of a severe haze event in Beijing

Gao *et al.* (2015a) estimated the health cost of a severe haze event in Beijing at US$ 253.8 million (€215 million) (**Table 7**) — which included 690 premature deaths, 45 350 cases of acute bronchitis, and 23 720 cases of asthma in the Beijing area. The study simulated $PM_{2.5}$ pollution levels during the haze using a model that considers the interactions between meteorology and chemistry (other pollutants were not considered). Human exposure and health impacts were then estimated, based on WHO guideline values for $PM_{2.5}$. Disutility costs were based on WTP studies, and the economic cost of mortality was estimated using the VSL method, which indicates how much people would be willing to pay for a reduction in risk of death. Calculations of health costs based on COI were also used.

Valuation Method	Endpoint	Cost per case (US$)	Cost per case (EU€)	Total cost (million US$)	Total cost (million EU€)
VSL	Mortality	273 513	233 006	189	161
WTP	Acute bronchitis	407	347	19	16
	Asthma	300	256	7	6
COI	Clinic visit	84	72	8	7
	Hospitalisation	2761	2352	31	27
Total				**254**	**216**

Table 6: Predicted health costs of severe haze event in Beijing.

The OECD calculates, using the reference value above (i.e. VSL quantified via WTP), that air pollution-related healthcare costs could rise from US$21 billion (€17.5 billion)/year in 2015, globally, to US$176 billion (€147 billion)/year in 2060; working days lost could rise from US$1.2 billion (€1 billion)/year to US$3.7 billion (€3.1 billion)/year, if no improvement is made (OECD, 2016).

4.7 Value Of a Life Year

The life expectancy on which the VOLY is based is key. Studies, for example, basing life expectancy on that of Japanese people — the longest-lived in the world and using this age to calculate years of life lost due to premature death in the majority of countries with lower life expectancies will give unrealistic figures. Desaigues *et al.* (2011) used country-specific life expectancies for Switzerland, Czech Republic, Germany, Denmark, Spain, France, Hungary, Poland, UK, and came up with the following VOLY estimates:

EU15 (represented by Denmark, Spain, France, Germany and the UK) + Switzerland: €41 000

New Member Countries (represented by Czech Republic, Hungary, Poland): €33 000

They argue that the VOLY for the whole of the EU is at least €25 000, and at most €100 000.

The way that costs are calculated has a significant bearing on results. For example, despite both using OECD and WHO data, air pollution costs related to fossil fuels in the EU are found to be about three times lower than in the US (Andersen, 2017). This is due to the fact that cost-benefit analysis in the US considers the number of statistical fatalities avoided, valued by reference to the VSL (presently $7.4 million). In contrast, the focus in Europe has been on changes in average life expectancy resulting from air pollution, or life years lost, monetised with the VOLY measure.

BOX 9.
HEATCO: The contribution of Years of Life Lost

In the HEATCO (Developing Harmonised European Approaches for Transport Costing and Project Assessment) project, the health cost of emissions from ground-level transport (e.g. road vehicles and diesel trains) were calculated based on exposure-response functions established in the EU ExternE project (Bickel *et al.*, 2006).

According to the study, in urban areas of Austria and the UK, each tonne of $PM_{2.5}$ emissions caused annual health costs of €450 000, and in France and Germany, €430 000 (2002 prices), for example.

Reduction in life expectancy, in terms of Years of Life Lost (YoLL), was the major contributor to health costs, though other health costs were also included, such as hospital admissions due to respiratory problems. For example, for each 1 000 tonnes of $PM_{2.5}$ emitted by ground transport, 5 800 years of life were estimated to be lost in the population of Austria and 6 000 in France.

This is based on the assumption that premature deaths linked to air pollution chiefly occur among the elderly and infirm. Indeed, Andersen (2017) found that the average age of air pollution victims is 77 years for men and 81 years for women, however, their remaining life expectancies were still about 10 further years, suggesting that the loss is not insignificant. In addition, if life expectancies are higher than those assumed, the loss is underestimated. Added to this uncertainty, the different VSL used in the EU and US means that costs used in the EU are significantly lower than those used in the US.

Andersen (2017) found that the reduction in average life expectancy related to air pollution was 10.7 years in the EU. With a VOLY of approximately €44 000, this gives an average cost of €470 800 per premature death due to air pollution.

Adjusting values in line with the VSL, gives a VOLY of $106 406 (€90 500) for the OECD, gives a cost of $1.14 million (€1.19 million), while, for the US, the cost would be over $2.3 million (€1.97 million).

It has also been noted that cost-benefit analysis generally finds the benefits of reduced mortality to be far higher than those of reduced morbidity (Chanel *et al.*, 2016). This is explained in that the WTP to reduce risk of death tends to be higher than the WTP to reduce risk of illness, or the market costs of illness. Epidemiology has also focused on mortality because it is easier to measure.

5. Applications

5.1 Assessing policies through monetised benefits

There is an increasing recognition that environmental and health impacts often require valuation in economic terms in order to receive adequate consideration in policy. Measuring the impact of policies using monetised health impacts has been taking place in the EU for several years, to allow a better comparison of trade-offs.

The Better Regulation Guidelines and associated 'Toolbox'[14] (2017) aim to ensure that the European Commission is equipped with relevant and timely information on which to base its decisions. The Better Regulation Guidelines for Impact Assessment[15] state that the most relevant impacts should be assessed both quantitatively and qualitatively — and should include monetised impacts — whenever possible. This approach complements and supports other approaches that may take into account health impacts but stop short of monetisation; the Guidelines are clear that non-quantifiable impacts should still be taken into account. Although there is no harmonised methodology for estimating economic costs and benefits for the health impacts associated with environmental risks, in 2008 the WHO and UNEP's Health and Environment Linkages Initiative highlighted the importance of integrating approaches and disciplines (WHO/UNEP, 2008)[16]. There is an increasingly widespread use of valuation techniques in impact assessments — especially regarding air pollution and chemicals (see **Box 10**).

BOX 10.

Costs and benefits of REACH restrictions

The main cost category assessed in the restriction cases is substitution costs, i.e. investment and recurring costs to switch to alternative substance. The total costs assessed for all the restrictions in the EU having gone through the REACH procedure is estimated at €290 million per year, and the cost per restriction case vary between €0 and €100 million.

The human health and environmental impacts of restrictions are more challenging to estimate — but for a few cases the monetised benefits to human and environmental health have been estimated. The relevant restrictions introduce benefits by avoided adverse health effects and negative impacts on environment as follows:

- Health benefits equivalent to over €700 million per year
- Reduction of around 190 tonnes of releases of substances of concern per year
- Positive health impacts or removed risk for at least 81,000 consumers and workers per year.

Source: ECHA, 2016.

14. https://ec.europa.eu/info/law/law-making-process/planning-and-proposing-law/better-regulation-why-and-how/better-regulation-guidelines-and-toolbox_en
15. https://ec.europa.eu/info/better-regulation-toolbox_en
16. https://ec.europa.eu/info/law/law-making-process/planning-and-proposing-law/better-regulation-why-and-how/better-regulation-guidelines-and-toolbox_en

One way to assess the effect of a policy, by indicating a monetised cost of a health impact due to pollution, is by calculating the number of premature deaths due to pollution multiplied by the VSL, as follows.

Premature deaths due to pollution x VSL = cost of health impact of pollution.

Conversely, if a mitigating action saves lives, then the economic benefit can be expressed as the VSL multiplied by the number of lives saved:

Number of lives saved by policy x VSL = value of policy.

BOX 11.

Health impacts: using avoided costs to evaluate policies

A review of the cumulative health and environmental benefits of chemical legislation, carried out for the European Commission, has estimated the reduction in the statistical number of cancer cases, and the costs avoided through the general European population's exposure reduction to benzene over 1999-2008. The authors link this explicitly to the implementation of the Fuel Quality Directive, in the following way.

In the case of benzene, exposure has reduced due to a range of factors, many not related to chemicals legislation. However, a decline in emissions and exposure from petrol (where the Fuel Quality Directive limited benzene concentrations around the year 2000) is estimated to have led to a cumulative reduction of 175 in the statistical number of cancer cases (i.e. incidences) caused over the period 1999-2008.

Based on VSL data (including WTP), this reduction (i.e. costs avoided) has been valued at €680-875 million in total, around €60 million per year.

Values	
Value of a statistical life for cancer, euros	5 000 000
Benefits (annual snapshot, 2014 vs 1999	
Number of deaths avoided in 2014 in comparison to 1999	15
Benefit (2014 vs 1999), euros	74 002 120
Benefits, cumulative	
Number of cases avoided (cumulative) (2000-2014)	175

Table 7: Benzene case study calculations. Source: European Commission, 2017a

Estimating savings and benefits in terms of avoided health costs is one way of evaluating the performance of pollution abatement policies (e.g. Holland, 2014b).

This type of valuation also enables a comparison of the costs and benefits. For example, the cost of implementing recent restrictions on the use of lead, chromium VI and methanol under the REACH regulation, in four Member States, has been estimated at €173.1 million per year (ECHA, 2016). At over €700 million per year, however, the estimated health benefits associated with these restrictions far outweigh this cost. Specifically, restricting lead and lead compounds in jewellery (in France) and consumer articles (in Sweden) contributes to avoiding IQ loss; restricting chromium VI in leather articles (in Denmark) reduces allergic symptoms; and restricting methanol in windshield washing fluids (in Poland) avoids fatalities (through its abuse as a substitute for alcohol). Monetised impacts in this assessment were based on WTP to avoid symptoms, cost of illness, productivity loss and to avoid loss of net benefits to consumers of products.

Another study looks at progress on reducing gaseous and particulate emissions in Europe between 1970 and 2010 (Crippa *et al.*, 2016). The researchers used the EDGAR v.4.3.1 (Emissions Database for Global Atmospheric Research) global anthropogenic emissions inventory to model levels of emissions in 2010, as if there had been no improvement in technologies or efforts to reduce emissions, and compared this with the actual scenario in 2010. In the 'no improvement' scenario, they found that emissions of sulphur dioxide would have been 129% higher in 2010, and $PM_{2.5}$ emissions 69% higher, for example. However, they also found that stagnation of energy consumption at 1970 levels would have lowered emissions.

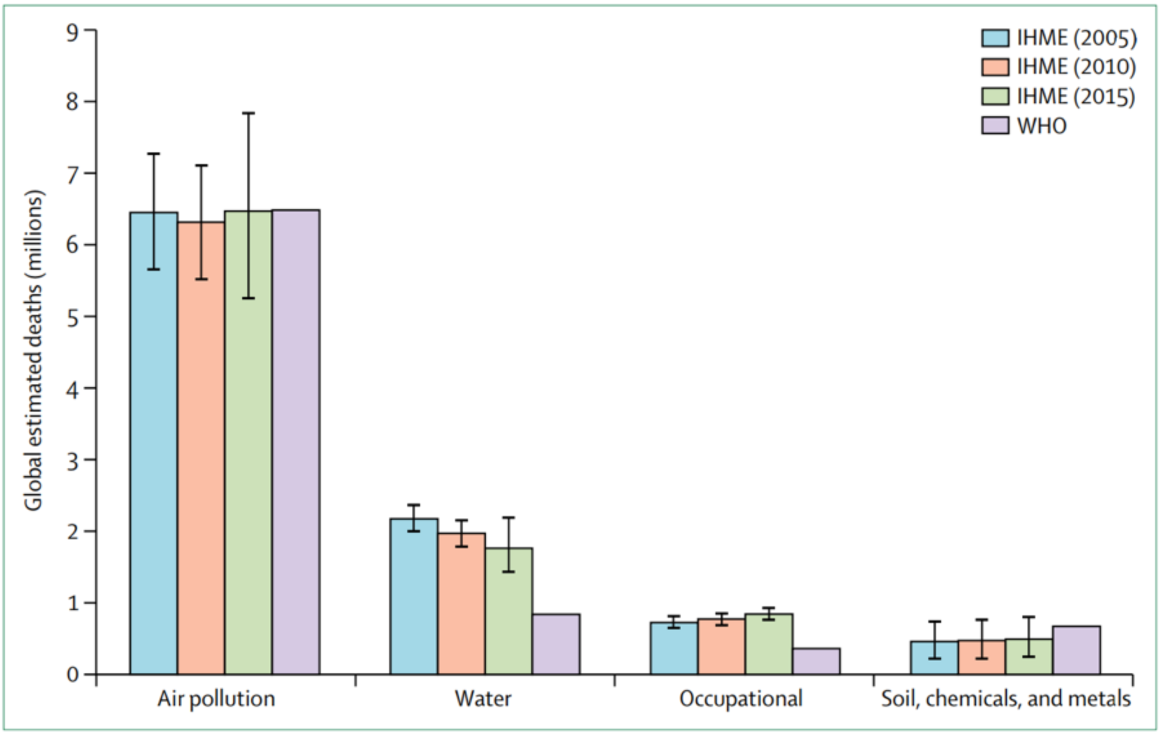

Figure 9: Global estimated deaths (millions) by pollution risk factor, 2005–15 Using data from the GBD study and the WHO. IHME=Institute for Health Metrics and Evaluation. Source: The Lancet 2017.

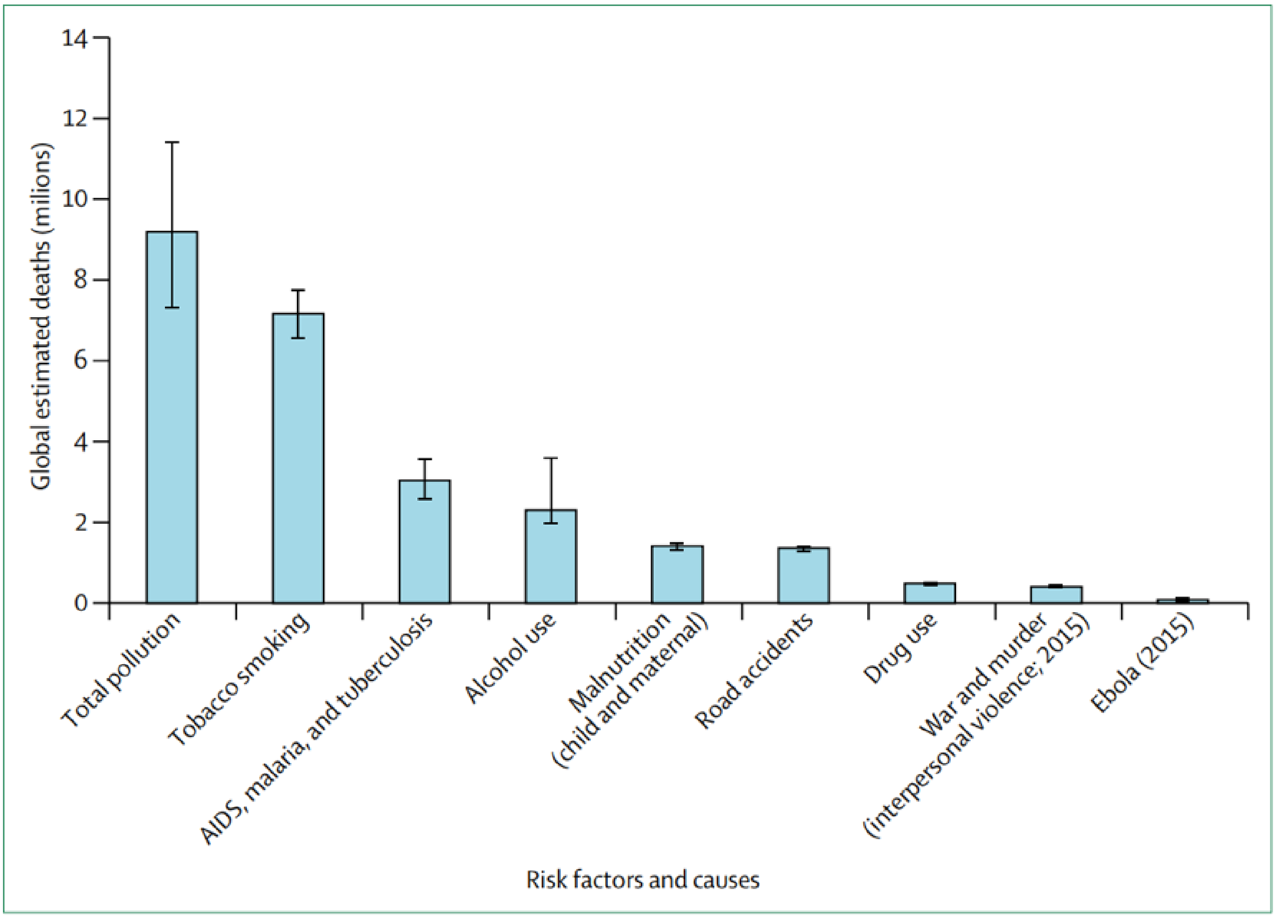

Figure 10: Global estimated deaths by major risk factor and cause, 2015 Using data from GBD, 2016.
Source: The Lancet, 2017. https://www.theThe Lancet.com/pdfs/journals/The Lancet/PIIS0140-6736(17)32345-0.pdf

Monitoring changes in pollution levels as well as health impacts will be necessary if future research is to evaluate the success of abatement strategies on health. Currently, the modelled scenarios of future impacts seem to address air pollution more than other types of pollution, reflecting the fact that air pollution is the first source of premature deaths linked to pollution in general (see **Figure 9**) and the availability of data. Many of these data are available due to the monitoring requirements of existing legislation, which is less the case for other types of environmental pollution.

5.2 Co-benefits

Reducing environmental pollution does not only have impacts on human health; there are broader implications as well. Having monetised the effects of climate change mitigation in terms of ambient air pollution, using VSL based on the OECD 2005 figure, Markandya *et al.* (2018) showed that, globally, health co-benefits are greater than the mitigation costs of achieving the Paris Agreement targets (2°C and 1.5°C). At the regional level, the costs of reducing greenhouse gas (GHG) emissions could be compensated with the monetised health co-benefits alone for China and India, whereas the co-benefits would make a valuable contribution towards covering the mitigation costs, from 7% to 84% in the EU-27 countries and from 10% to 41% in the USA. The researchers note that attaining a 1.5°C or even 2°C target will have great climate change impact benefits for all regions, including for health. Overall, the additional cost of going from a 2°C target to a 1·5°C target was calculated at around 20%.

A recent systematic review studied the human health and environmental co-benefits of domestic and global GHG mitigation (in 2050; Zhang *et al.*, 2017). Europe was the continent that was the subject of most research on co-benefits[17].

Another study (West *et al.*, 2013) found that, in 2050, global average co-benefits exceed the estimated carbon price. Co-benefits of avoided air pollution mortality were monetised using high and low values of a statistical life (VSLs) and were compared with the marginal costs of GHG reductions (the global carbon price) from 13 models (from Clarke *et al.*, 2009). In 2030, the monetised mortality co-benefits exceed the median carbon price in all regions but Australia. By 2100, GHG reductions and costs increased markedly, as more expensive reduction measures are implemented, and co-benefits are within the low range of the carbon price. Monetised co-benefit estimates are \$50–380 (ton CO_2)−1 for

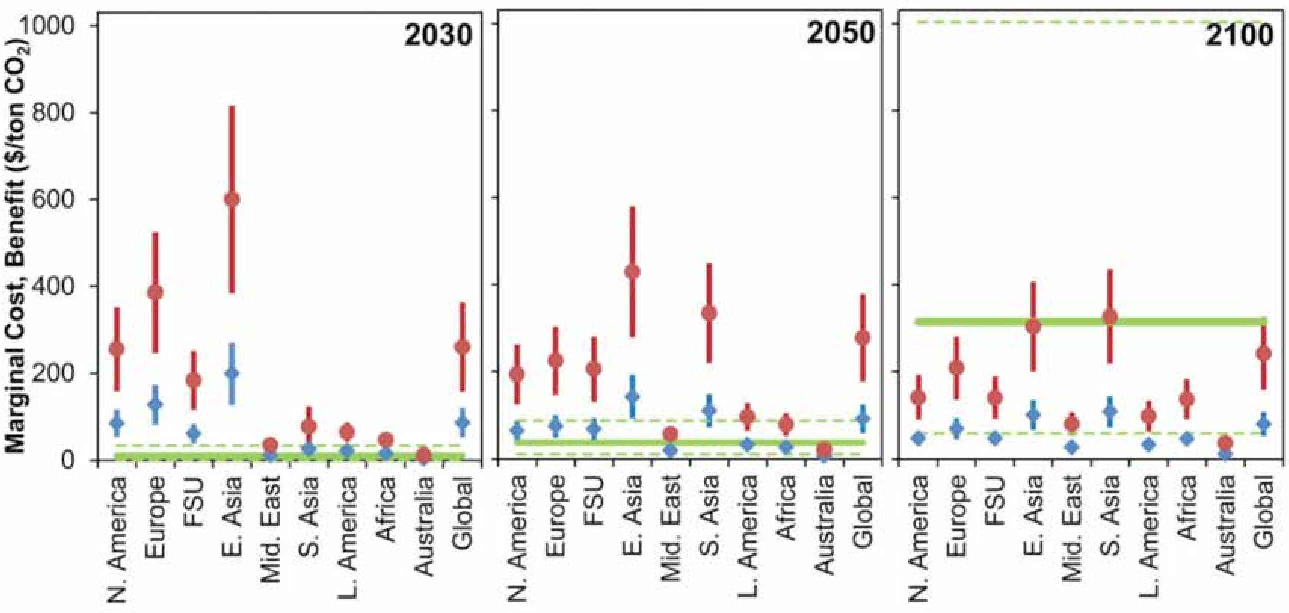

Figure 11: Regional marginal co-benefits of avoided mortality under high (red) and low (blue) VSLs, and global marginal abatement costs (the carbon price), as the median (solid green line) and range (dashed green lines) of 13 models. Marginal benefits are the total benefits (sum of ozone respiratory, $PM_{2.5}$ CPD, and $PM_{2.5}$ lung cancer mortality) divided by the total CO_2 reduction, in each year under RCP4.5 relative to REF. Uncertainty in benefits reflects 95% confidence intervals on the CRFs. Source: West *et al.* (2013). The researchers also acknowledge that the co-benefits depend on the future scenario and climate policies applied, which are uncertain (e.g. Rao *et al.*, 2017).

17. For Europe, studies on the co-benefits of ecosystem impacts (n.=71 papers), economic imacts (n.=54), health impacts (n.=44) and resource efficiency benefits (n.=45) were also found, out of a total of 1554 papers.

the worldwide average, \$30–600 for the US and Western Europe, \$70–840 for China, and \$20–400 for India (range includes differences over three years, high and low VSLs, and uncertainty in the concentration-response functions (CRFs)).

The EU-funded ClimateCost study looking at the costs and benefits of the adverse economic, health and environmental impacts of air pollution calculates that the annual air quality co-benefit in the EU-27 in 2050 under a 2°C (mitigation) scenario falls in the range of €44 to €95 billion, and that 480 000 years of life expectancy could be gained annually in the EU-27 by 2050, due to the improvement in air quality if we keep within 2°C. The researchers also calculated that, under this mitigation scenario, the EU could avoid significant air quality abatement costs of €36 billion/year in 2050. The ClimateCost study used both VSL and VOLY for valuation, to help mitigate the weaknesses of both methods (Holland *et al.*, 2011).

5.3 Internalising costs

Policies that look to set a price on emissions can look to internalise costs, based on estimates of health impacts. For example, a US study looked at how health damages might be incorporated into the costs of energy (Brown *et al.*, 2017). The study modelled a range of scenarios in which low (\$364), medium (\$1970) and high (\$4700) fees per ton of emissions were applied (2005 prices), based on associated health costs reported in literature.

The effects of these fees on the energy sector were predicted using the MARKAL (MARKet ALlocation) tool, which determines likely responses to fees and effects on emissions, compared to a baseline scenario under current policy. For example, additional fees might encourage the application of filters in power station chimneys. In all cases — including the baseline and despite increased energy demands — emissions were shown to decrease over time, compared to the situation in 2010. Compared to the baseline, low fees would lead to a decrease of a few per cent, mid-range fees 11–33%, and high fees would result in decreases of up to 82%, in 2045. These results show that shifting the cost of health externalities to source emitters could lead to reduced emissions. The researchers note that there is uncertainty related to the damage figures used and that there are limitations on using fees to internalise externalities but monetisation of impacts can help with setting fee levels (Jenkins, 2014).

Another recent study uses health costs in assessing the life cycle impacts of electric vehicles (EVs) to conclude that EVs are least damaging to health when not exported long distances to the retail market, and when charged using low-emission electricity (Romejko and Nakano, 2017).

6. Conclusion: harmonising the valuation of health impacts

A number of different approaches can be used to value health, life and illness due to environmental pollution (or the reduction thereof), with differences in approach depending on discipline and location. Each approach has strengths and weaknesses; **it is essential to understand the method and assumptions behind monetisation approaches** before using the valuation figures. To aid understanding, we include the following table.

Market-based approaches	Non-market based approaches		
Cost of illness	**Revealed preference**	**Willingness to pay**	**Willingness to accept**
Calculates healthcare costs and opportunity costs (lost productivity)	An indication of how much individuals value avoiding pollution	Used to monetise non-market or intangible health damages	Amount of money an individual would accept to tolerate an increase in health risk from pollution
Used to calculate how much society is spending on disease	Uses both intrinsic and contextual factors to estimate values	Willingness–to-pay questionnaires find average values related to health/illness	Assumes the public should not be expected to pay to prevent externalities caused by pollution
Does not capture full costs as neglects disutility costs	Focuses on market behaviour and pricing	Hypothetical and can depend substantially on variables (e.g. income, sense of place, level of education, democracy and government stability)	Not bound by income — so can result in large values
Only looks at market costs	Often used to study noise avoidance	Widely used in cost-benefit analysis	
	Less used than WTP in European studies	Can also be based on hedonic pricing	
	Does not necessarily capture any of the health costs associated with noise exposure	Can underestimate the value of health to society	

In calculating health damages from environmental pollution, the primary current use of monetised health costs is to incorporate them into policy assessment (including via cost-benefit analyses). Combining approaches in a single analysis needs to be done carefully. However, **combining several approaches** (e.g. WTP, COI, VSL and VOLY) **may mitigate some of the issues** created by using

Table 8: Summary table of strengths and weaknesses of monetisation approaches for the costs of environmental pollution.

Value of a Statistical Life	Value Of a Life Year	Quality-Adjusted Life Year	Disability-Adjusted Life Year
What people would pay to reduce the risk of premature death	Values premature death due to pollution (cost of pollution/remaining life expectancy = VOLY)	QALY = a year of life spent in 'perfect' health	Indicates the relative impact of illness and injury on loss of healthy life years; total loss of DALYs is the 'total burden of disease'
Derived from the trade-offs people are willing to make between risk of fatality and wealth	Can be used to account for the cost of pollution in terms of years of life lost	Can be used to compare the potential effects of intervention vs no intervention, or different types of intervention to inform resource allocation	Combines the number of years lived with a disability (morbidity) and healthy years lost to premature death
Often derived from stated preference, e.g. WTP	Can be calculated variously, e.g. as proportional to VSL or through WTP	Captures both mortality and morbidity	Can be expressed as the gap between current health status and an ideal health situation; requires inputs about population age structure, life expectancy and cases of disease
Only values mortality	Confers higher value on risks leading to a significant loss of life years per person affected	Values are country-specific and not fixed	Can be used to quantify disease burden due to environmental pollution between different countries or population groups
Can be dependent on variables in a similar way to WTP	Life expectancy used can create variation between studies	Some ethical limitations to using QALYs	Interpretation of results must factor in life expectancies and weightings assumed
Ethical limitations: includes GDP, so can place e.g. a higher value on human life in developed countries than developing ones		Some advise it should not be used in isolation	Does not account for social context or vulnerability

only one. It is crucial to ensure that impacts are not omitted, or that impacts, or causalities, are not misallocated. **Increasing the standardisation of methods of valuation between air, noise and chemical pollution will be a crucial step** to consider the costs and benefits of proposed solutions to some complex, contemporary, global problems. Studies on differences between environmental economic approaches and QALY and DALY analysis of environmental health policy interventions could warrant further study.

Significant differences in perspective are also evident between health economics, which works to prioritise resources within a budget, and environmental economics, which does not try to fit costs generated into a budget, but instead tries to create preliminary benchmark values for external costs not currently valued by the market. At times, these disciplines, and the participants who inform their results, diverge in the values they give to a given health outcome. However, with the increasing use of methods of analysis that account for disutility and pain and suffering costs, **there is potential for some alignment of perspectives, techniques or methods across health and environmental economics**, in using such non-market cost information to inform decisions. Indeed, there is more work to be done to bring together the work of health and environmental economists, and to explore and **hone our understanding of the values societies assign to concepts such as a healthy life, or morbidity**. As more evidence is generated on non-market valuation, defining and counting separate values may become easier.

As the research around valuation progresses, it is also essential to keep abreast of advances elsewhere; **one of the most important factors to take into consideration alongside the valuation of health is the dose-response or concentration-response relationship** on which estimations of damaging impacts are based. It is also worth remembering that this report has limited scope, addressing only air and noise pollution and toxic chemical exposure-related studies, and that other types of environmental impacts, such as biological contamination or ionising radiation, may be subject to different considerations in both market and non-market valuations.

While life and health are clearly invaluable, and monetised health impacts are not ready to be used as some sort of universal proxy, it is also clear that monetisation can help policymakers to compare options as part of a well-rounded analysis, one which already considers a diverse range of values. It brings another, useful aspect through valuation, which can permit calculation of the impacts of alternative options and comparison of ever-more-realistic scenarios. Monetisation of these types of health effects also has a **practical use as a communication tool, to measure successes and to evaluate and disseminate the results of policy implementation**; indeed monetised health impacts are already helping to measure the impact of policies in the EU.

Continuing to develop the linkages and coherence between the different types of health-monetisation analysis will play a key role in enabling better, more balanced global decisions, and in **ensuring that non-market costs are sufficiently represented in systems currently biased towards the market**.

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