Chapter 2

A framework for assessing the economic consequences of outdoor air pollution

This chapter presents the methodology used in this report to analyse the economic consequences of outdoor air pollution. The methodology is based on the impact pathway approach, which requires multiple steps, from creating projections of air pollutant emissions, to calculating concentrations of key pollutants, calculating the biophysical impacts on health and crop yields, and calculating the economic costs with the ENV-Linkages model for market impacts and with results of direct valuation studies for non-market impacts. For each step the modelling framework and economic techniques used are explained.

2.1. Overview of the assessment framework

The framework to assess the economic consequences of outdoor air pollution links projections of economic activity to changes in air quality and to the associated biophysical and economic consequences. Modelling and projecting these consequences is done using the impact pathway approach, which requires multiple steps and the use of different techniques and modelling frameworks. Figure 2.1 summarises the different steps employed in this analysis.

First, an economic modelling framework is needed to obtain projections of economic activity, as well as the emissions that they imply. A computable general equilibrium (CGE) model, such as the OECD's ENV-Linkages model, is the ideal framework as it also includes projections of sectoral and regional economic activities. As explained in Section 2.2, the projections of economic activity to 2060 at the sectoral and regional level rely on a range of important drivers and exogenous trends, including those for demographic developments and technological change.

Second, for each year, emissions of a range of air pollutants are linked to the different economic activities as projected in step 1. In some cases, emissions are directly linked to a specific element in the production process, such as the combustion of fossil fuels. In other cases, emissions are linked to the scale of activity, and thus to production volumes. Some emissions that are not directly linked to economic activity are projected using exogenous trends. Together, these establish projections for regional emission levels, as described in Section 2.3.

Third, emissions of air pollutants are used to calculate concentrations of $PM_{2.5}$ and ozone. This step relies on an atmospheric dispersion model and on downscaling national emissions to a spatial grid of local emission levels. It delivers a "gridded map" of concentrations for the period between 2010 and 2060, which forms the basis for the assessment of the health and environmental impacts. Section 2.4 explains this step and the modelling framework used in detail.

Figure 2.1. **Steps to study the economic consequences of outdoor air pollution**

1 PROJECTIONS OF SECTORAL ACTIVITIES

- The OECD's ENV-Linkages model is used to obtain projections of sectoral activities.

2 PROJECTIONS OF AIR POLLUTANT EMISSIONS

- Data on emissions of air pollutants are linked to sectoral activities within ENV-Linkages to obtain projections of future emissions levels that are consistent with the economic baseline.

3 CONCENTRATIONS OF AIR POLLUTANTS

- The European Commission Joint Research Centre (EC-JRC)'s TM5-FASST model is used calculate concentrations of particulate matter and ozone.

4 BIOPHYSICAL IMPACTS OF OUTDOOR AIR POLLUTION

- The concentrations of PM2.5 and ozone are used to calculate the biophysical impacts on health (e.g. number of lost working days) and agriculture (e.g. crop yield changes).

5 VALUATION OF HEALTH IMPACTS OF OUTDOOR AIR POLLUTION

- Unit values are established for each health endpoint and health market impacts are evaluated and converted to economic costs (e.g. health expenditures).

6 MARKET COSTS OF OUTDOOR AIR POLLUTION

- The indicators relative to the market impacts are used to calculate the macroeconomic costs of air pollution in the ENV-Linkages model, so as to obtain an overall effect on economic growth.

7 WELFARE COSTS OF OUTDOOR AIR POLLUTION

- Impacts that cannot be included in the model are evaluated using results from direct valuation studies. Market costs are also expressed as welfare costs.

Fourth, the biological and physical impacts caused by the high levels of population-weighted concentrations of $PM_{2.5}$ and ozone are calculated using data on population, exposure to the pollutants and results of studies calibrating concentration-response functions (see Section 2.5). This step aggregates the detailed spatial concentration information to the national level, covering 181 countries for $PM_{2.5}$, and 161 for ozone. A range of indicators is used to present the biophysical impacts, to allow differentiated effects on e.g. number of lost working days, hospital admissions and agricultural productivity impacts.

Fifth, the direct economic consequences of the health impacts are calculated at the country level. This step comprises the calculation of unit values for the evaluation of the health impacts for each endpoint. For example, hospital admissions are translated into health expenditures and a welfare cost is established for each premature death. This step is further discussed in Section 2.6.

Sixth, market costs are analysed using the ENV-Linkages general equilibrium model, which is also employed in steps 1 and 2. The direct impacts in terms of agricultural yield shocks, changes in health expenditures and labour productivity changes are aggregated to the regional aggregation level of the CGE model and used as an input to calculate the economic consequences of outdoor air pollution (see Section 2.7 for more details). This step reflects the feedback of outdoor air pollution impacts on the economy, and represents the core of the assessment of the economic consequences of outdoor air pollution.

Finally, in the last step, laid out in Section 2.8, the costs that are not directly linked to any economic variable are quantified. These non-market costs are evaluated in terms of welfare changes using results from direct valuation studies.

The impacts for which there was enough reliable data for quantification and which are included in the modelling framework are those related to change in healthcare expenditures, labour productivity changes linked to lost working days, and agricultural crop yield changes. It was not possible to include other impacts, such as those on forestry, biodiversity or cultural heritage, in the modelling framework because there are no robust studies that quantify the pollution-attributable costs at the global scale.

In line with the OECD's analysis of the economic consequences of climate change (OECD, 2015), these are introduced in the model following a production function approach (for a general framework see Sue Wing and Fisher-Vanden, 2013; for an overview of modelling applications to climate change see Sue Wing and Lanzi, 2014; Vrontisi et al., 2016, use the same approach for the assessment of the EU's Clean Air Policy Package). This means that each impact is linked to variables that are at the core of the production functions underlying the model structure.

The results are presented in the form of a stream of future costs of inaction on outdoor air pollution. For a cost-benefit analysis of specific policies, the net present value of both the costs and benefits of the policy action would need to be quantified. This additional step, which is not included in this report, crucially depends on the choice of a discount rate to evaluate intertemporal changes. By presenting the economic consequences in this report as they emerge over time, rather than converted to a present value, this controversial step is avoided.

Theoretically, one could expand the modelling framework with a utility function that includes health and other relevant factors. This approach has been experimented with in e.g. Mayeres and Van Regemorter (2008), but such an approach requires very bold assumptions on the substitutability of consumption and health impacts, and is limited to morbidity impacts. Further, it is virtually impossible to find robust estimates of the substitution elasticities between these various elements in the expanded utility function for all regions. Therefore, non-market impacts, such as the economic value of premature deaths or the disutility linked to illness, are assessed outside the general equilibrium modelling framework.

2.2. Socio-economic trends in a baseline projection

The OECD's multi-region, multi-sector dynamic CGE model ENV-Linkages (see Chateau et al., 2014 and Annex A for further details on the model) is used to create a socio-economic baseline projection of sectoral and regional economic activities until 2060. The baseline projection used in this report excludes new policies and feedbacks from air pollution and climate change impacts on the economy. It serves as a reference to calculate the future costs of air pollution. The baseline projection used in this report is identical to the no-damage baseline used for the assessment of the economic consequences of climate change (OECD, 2015).

Two different baseline projections are presented in this report. The "central" projection describes a baseline projection that considers the feedback effects of outdoor air pollution on the economy. It describes the main socioeconomic trends, emissions and concentrations of air pollutants and the resulting impacts on health and agriculture. It also contains the feedbacks of these impacts on the economy. This central projection is contrasted with a hypothetical socioeconomic projection which excludes economic feedbacks of air pollution. This "no-feedback" baseline projection describes hypothetical baseline developments in absence of feedback effects of air pollution on the economy, and is used as the starting point to calculate emissions and concentrations of air pollution, which are then used to assess the impacts and economic feedbacks of the central projection.[1]

The logic of this approach is not to deny that outdoor air pollution is already affecting the economy, but rather to measure the total economic consequences of such air pollution. The no-feedback projection describes the pressures that economic activity puts on the environment, by linking economic activity to emissions and concentrations. The central projection takes the corresponding air pollution impacts, describes how these feed back to the economy and projects the resulting changes in economic activity and specific indicators such as gross domestic product (GDP). The difference in GDP between the two projections reflects the full macroeconomic costs of inaction of outdoor air pollution.

A baseline projection is not a prediction of what will happen, but rather it describes a certain storyline on how key economic and demographic trends affect future economic development in the absence of unexpected shocks. The chosen baseline reflects a continuation of current socio-economic developments, including demographic trends, urbanisation and globalisation trends. The baseline also reflects a continuation of current policies for climate, energy and air pollution (see Box 2.1 for an overview of air pollution policies included in the baseline).

Demographic trends play a key role in determining economic growth. Population projections by age, together with projections of participation and unemployment rates, determine future employment levels. Human capital projections, based on education level projections by cohort, will drive labour productivity. Demographic projections, including effects of changes in fertility, death rates, life expectancy and international migration, are taken from the UN population prospects (2012). The labour force database (participation rates and employment rates by cohort and gender) is extracted from ILO (2011) active population prospects (up to 2020) and OECD Labour Force Statistics and Projections (2011).

Box 2.1. **Current air pollution policies included in the baseline**

Governments have already implemented a range of policy approaches to limit outdoor air pollution. Information on a large number of economic instruments and voluntary approaches for air pollution can be found in the OECD database on instruments used for environmental policy, at www.oecd.org/env/policies/database. In many countries, so-called "command-and-control" approaches using e.g. regulatory standards are complemented by various economic instruments such as taxes and tradable permit schemes. Voluntary programmes aimed at replacing ovens and heaters, replacing old with LPG and enhanced cook stoves, and retiring old highly-polluting vehicles have also been introduced in recent years in several countries.

In most OECD countries, air pollution policy interventions have become increasingly integrated over the past 10-15 years, helping to increase cost efficiency. Examples include the US Clean Air Act, the Canada-US Air Quality Agreement, the Clean Air Policy Package of the European Commission, and the National Environment Protection Measure for Ambient Air Quality (Australia), all of which have set standards for air quality, focusing on target-setting for a range of air pollutants from stationary sources. These overall frameworks include legislative programmes which target specific sectors, such as power generation, transport, and industrial and residential energy demand. In non-OECD economies, there are fewer examples of cohesive programmes for controlling air pollution. In recent years, much of the focus is on specific policies for controlling emissions from transport, both through standards and economic instruments.

The emission projections presented in this report reflect the effects of current legislations as depicted by International Institute for Applied Systems Analysis (IIASA) in the Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) model (see Section 2.3 for more details). In principle, all legislation for which information was available is included in the emissions projections, where relevant (e.g. for fuel taxes and congestion charges) through the associated projections of energy use. However, any policy that was not yet fully implemented by late 2012, or that still requires a policy effort to be reached (e.g. the Chinese 11th five-year plan), is excluded from the baseline. This approach provides a snapshot of the effect of policies on current and future emissions; it is a reference point for the assessments of the costs of inaction and the benefits of policy action, and does not reflect a view on the state of very recent and planned environmental policies.

The regional aggregation of ENV-Linkages is used to calculate economic activity, emissions of air pollutants and the feedbacks from pollution impacts on the economy (more detailed representations underlie the calculations of concentrations and biophysical impacts; see Sections 2.4 and 2.5). As shown in Table 2.1, ENV-Linkages distinguishes 12 major countries and 13 groups of countries (regions), based on a mixture of geographical and economic characteristics. For illustrative purposes, some graphs and tables in this report group the underlying 25 regions in 8 "macro-regions", but in all cases the analysis is done at the 25 region level.

Macroeconomic projections for OECD countries are aligned with OECD (2014c). Projections on the structure of the economy, and especially on future sectoral developments, are fundamental for the analysis in this report as they affect the projected emissions of air pollutants. The sectoral assumptions are particularly important as different emission sources are linked to different sectoral economic activities. For instance, final energy demand and power generation affect emissions of a range of pollutants from combustion processes, and in agriculture emissions, especially of NH_3, are linked to the production processes of agricultural goods.

Table 2.1. **Regional aggregation of ENV-Linkages**

Macro regions	ENV-Linkages countries and regions
OECD America	Canada Chile Mexico United States
OECD Europe	EU large 4 (France, Germany, Italy, United Kingdom) Other OECD EU (other OECD EU countries) Other OECD (Iceland, Norway, Switzerland, Turkey, Israel)
OECD Pacific	Oceania (Australia, New Zealand) Japan Korea
Rest of Europe and Asia	China Non-OECD EU (non-OECD EU countries) Russia Caspian region Other Europe (non-OECD, non-EU European countries)
Latin America	Brazil Other Lat.Am. (other Latin-American countries)
Middle East & North Africa	Middle-East North Africa
South and South-East Asia	India Indonesia ASEAN9 (other ASEAN countries) Other Asia (other developing Asian countries)
Sub-Saharan Africa	South Africa Other Africa (other African countries)

Projections of sectoral energy intensities until 2035 are in line with the IEA's World Energy Outlook "Current Policy Scenario" (CPS) (IEA, 2013). After 2035, the IEA trends are extrapolated to fit the macroeconomic baseline thereafter. In fast-growing economies such as the People's Republic of China (henceforth "China"), India and Indonesia, the IEA projects coal use to increase in the coming decades. In OECD regions, however, there will be a switch towards gas, not least in the USA, and this especially in the power generation sector. Further, in OECD economies, energy efficiency improvements are strong enough to imply a relative decoupling of energy use and economic growth, while for emerging economies the decoupling will only be effective in the coming decades. The increase in final energy demand is driven by electricity and by transport; in particular in emerging economies. In line with the trends of the IEA's CPS scenario, electrification of transport modes is assumed to be limited globally.

The projections on agricultural yield developments (physical production of crops per hectare) as well as main changes in demands for crops as represented in the ENV-Linkages baseline are derived from dedicated runs with the International Food Policy Research Institute (IFPRI)'s IMPACT model (Rosegrant et al., 2012) using the socioeconomic baseline projections from ENV-Linkages and excluding feedbacks from climate change on agricultural yields. The underlying crop model used for the IMPACT model's projections is the DSSAT model (Jones et al., 2003). As IMPACT only provides projections to 2050, the trends are linearly extrapolated to 2060. The detailed projections of agricultural production and consumption from IMPACT are then summarised and integrated in ENV-Linkages. According to the projections, while population will increase by 50% from 2010 to 2060, average per capita income is projected to more than double in the same time span. Agricultural production as measured in real value added generated in the agricultural sectors will also more than double by 2060, partially reflecting a shift in diets towards higher-value commodities. The large increase in agricultural production is characterised by a growing share of production in African countries. On the contrary, the market share of OECD countries is projected to decrease.

In principle, feedbacks from climate change on agricultural yields could threaten projected improvements in global food security. Such feedback effects are described extensively in OECD (2015), but excluded from the calculations in this report to allow full focus on the impacts of air pollution. An integrated analysis of both climate and pollution feedbacks is left for future research, but interactions between both themes are discussed in Section 4.2.

2.3. From economic activities to air pollutant emissions

Emissions of air pollutants have been included in the ENV-Linkages model linking them to production activities in different key sectors. The main emission sources are power generation and industrial energy use, due to the combustion of fossil fuels; agricultural production, due to the use of fertilisers; transport, especially due to fossil fuel use in road transport, and emissions from the residential and commercial sectors.

In this study, estimates for selected air pollutants were included: sulphur dioxide (SO_2), nitrogen oxides (NO_x), black carbon (BC), organic carbon (OC), carbon monoxide (CO), volatile organic compounds (VOCs) and ammonia (NH_3). Even if this list does not cover all air pollutants, it includes the main precursors of PM and ground level ozone, which are the main causes of impact on health and on crop yields.

The data on air pollutants used for this report is the output of the GAINS (Greenhouse Gas and Air Pollution Interactions and Synergies) model (Amann et al., 2011 and 2013; Wagner et al., 2007 and 2010; Wagner and Amann, 2009), developed at International Institute for Applied Systems Analysis (IIASA). The GAINS model estimates historic emissions of air pollutants using data from international energy and industrial statistics (not least the EDGAR database), emission factors originating from peer reviewed literature and measurement campaigns, and information about implementation of environmental legislation. Although global coverage and international comparability are most important, the results are compared with the national and international emission inventories that are either published in peer reviewed literature or supplied by countries to the international organisations within existing commitments, e.g. Convention on Long-range Transboundary Air Pollution (LRTAP), United Nations Framework Convention on Climate Change (UNFCCC) Protocol, and EU legislations. The GAINS model structure includes all key known emission sources distinguishing up to about 2000 sector-fuel-technology combinations for each of the 170 countries and regions covered in the model.

The emission projections of the GAINS model used for this project are those relative to the “Current Legislations” (CLE) scenario, which reflects the state of committed air pollution legislation assuming that the required standards can be achieved by existing technologies. These projections are based on activity levels and energy use that reflect those of the 2011 World Energy Outlook (IEA, 2011), but have been rescaled to the more recent energy demand projections of the ENV-Linkages baseline. The projections of the GAINS model used for this project are those that have been prepared for the EU FP7 LIMITS project (see e.g. Rao et al., 2016; Kriegler et al., 2013). The LIMITS project was a large model inter-comparison exercise on interactions between climate policies and other environmental issues, such as air pollution and energy security.[2]

The CLE scenario used in this analysis represents the status of air pollution policies by the end of 2010. Hence, some important developments of the past few years are not captured. The most prominent example is the 11th five year plan in China and the associated legislation; the targets were published already in 2010 but the specific laws and emission limits (more stringent SO_2 and NO_x legislation for the power sector and also for industrial boilers) that are needed for the multi-sectoral assessment of emission factors were introduced later and these could not be considered in the current version of the GAINS scenario.

Emission coefficients have been calculated using the GAINS model projections until 2050. The coefficients are sector- and region-specific to reflect the different implementation rates of respective technologies required to comply with the existing emission legislation in each sector and region. They also change over time to reflect technological improvements, the change in the age structure of the capital stock (more recent generations of equipment submitted to environmental policies replacing the older ones), and the influence of existing policies. Between 2050 and 2060, the emission coefficients (but not total emissions) are assumed to be constant.

The emission coefficients are linked to the projected activity levels to obtain emission projections that are coherent with the economic baseline. Coefficients related to emissions from combustion processes in industrial sectors, transport and residential and commercial energy demand are calculated and linked to the inputs of fossil fuels.[3] Other emissions are linked directly to output (e.g. agricultural goods, cement, metals or waste). Finally, some sources of emissions have been included exogenously in the model as it was not possible to link them to specific economic activities. These are for instance emissions from biofuels. Emissions from forest, agricultural and savannah burning could not be included as they cannot be easily projected to future years. Emissions from aviation and marine bunkers have not been included as they were not part of the GAINS database, although in some coastal regions the effects of marine bunkers on local concentration levels may be significant. This means that, while the main sources of emissions are considered, total emissions of air pollutants have likely been underestimated.

2.4. From emissions to concentrations of air pollutants

Emission projections of precursor gases are used to calculate the associated concentrations of $PM_{2.5}$ and ground level ozone (O_3). High concentrations of $PM_{2.5}$ and O_3 are the drivers of strong impacts on human health and the environment. As discussed in Section 1.3, health impacts caused by NO_2 could not be included in the analysis.

The concentrations of ozone and $PM_{2.5}$ have been calculated using the European Commission Joint Research Centre (EC-JRC)'s TM5-FASST (Fast Scenario Screening Tool) model, which has also been used in e.g. UNEP (2011), in the EU FP7 LIMITS project (Rao et al., 2016; Kriegler et al., 2013) and for the Global Burden of Disease studies (Forouzanfar et al., 2015, and Brauer et al., 2016). TM5-FASST is a reduced form version of TM5 CTM (Krol et al., 2005; Huijnen et al., 2010), a global nested 3-dimensional atmospheric-chemistry-transport model, which simulates ozone and aerosol components with a spatial resolution of 1°×1°.[4] TM5-FASST is based on a set of pre-calculated linear emission-concentration response functions for 56 emitting source regions (Leitao et al., 2015), linking the emissions of precursors SO_2, NO_x, CO, BC, OC, VOCs and NH_3 to the resulting concentrations of pollutants O_3 and $PM_{2.5}$. For further information on TM5-FASST, see Annex B.

While the concentrations are calculated using the ENV-Linkages emission projections as an input, TM5-FASST also includes a fixed natural component from wind-blown dust and sea salt, hence considering both natural and anthropogenic pollution sources. While dust and sea salt are particularly strong in areas with low or no population, they can be carried by winds so it is still important to take them into consideration. Furthermore, TM5-FASST also considers climatic projections in calculating the concentrations, as climatic conditions influence the chemical reactions between pollutants and hence the levels of concentrations. For this project, the RCP8.5 (Riahi et al., 2007) scenario is used. This scenario is the closest to the ENV-Linkages projection of greenhouse gas emissions and average temperature increase and it was previously used as a reference climate scenario for the analysis of the economic consequences of climate change (OECD, 2015).

As impacts are related to exposure, the concentrations are calculated as population-weighted mean concentrations, rather than average concentrations across areas with widely varying population densities. The calculation of the national means of population-weighted $PM_{2.5}$ concentrations is based on combining the spatial concentrations with population maps that approximately reproduce urban background (Rao et al., 2012). The TM5-FASST model also takes into consideration population projections and urbanisation. This is fundamental as the population-weighted concentrations also need to reflect the higher levels of exposure caused by urbanisation.

The TM5-FASST model takes as input the emission projections of the ENV-Linkages model for each of the precursors, regions and sectors considered in the model. The sectoral contributions for each primary pollutant are detailed as much as possible, distinguishing for example between emissions from transport, energy supply and demand, residential and commercial sectors, agriculture, industry and chemicals. This sectoral categorisation is used in the atmospheric model to associate the emissions to specific locations and to estimate the local urban increment from primary $PM_{2.5}$ emissions associated with transport and the residential sector.

A remapping process is used to translate the emission projections for the 25 aggregate regions of ENV-Linkages to the more detailed 56 source regions required for the TM5-FASST model. This is done using available information on emissions from individual countries from a reference gridded emission dataset, in this case RCP8.5 (Riahi et al., 2007), as a proxy for the baseline projection developed in the current study. In a first step, the relative contributions of all countries that are part of a given ENV-Linkages region to the emissions in the RCP8.5's region are used to break down the emissions from the ENV-Linkages' regions to individual countries. In a second step the countries' emissions are re-aggregated to the 56 TM5-FASST source regions.

Concentrations of $PM_{2.5}$ that are used for the calculations of the health impacts are quantified as population-weighted $PM_{2.5}$ values per country. TM5-FASST provides different metrics for ozone impacts. For the O_3 impact on human health, the maximal 6-months mean of daily maximal hourly ozone (M6M) is most appropriate. For damages to crops, an average is taken of the impacts as calculated using AOT40, which is the accumulated hourly ozone above 40 parts per billion (ppb) during a 3-monthly growing season; and using M12, which is the daytime (12 hours) mean ozone concentration during a 3-monthly growing season. These indicators for concentrations of $PM_{2.5}$ and ozone are the starting points to calculate impacts on health and on crop yields.

2.5. From concentrations to impacts on health and agriculture

The following *health impacts* of $PM_{2.5}$ and O_3 were assessed in this analysis: mortality, hospital admissions related to respiratory and cardiovascular diseases, cases of chronic bronchitis in adults and in children ($PM_{2.5}$ only), lost working days ($PM_{2.5}$ only), restricted activity days, and minor restricted activity days due to asthma symptoms ($PM_{2.5}$ only). This selection of impacts is based on the recommendations of the World Health Organization (WHO) under the "Health risks of air pollution in Europe" (HRAPIE) study (WHO, 2013). While this covers a large part of the recognised economic impacts of air pollution on health, there are other impacts that could not be calculated as there is not enough information available (see Chapter 1 for a discussion of other impacts).

The effects of air pollution on health are assessed with concentration-response functions, which link health impacts to the population-weighted mean concentrations of $PM_{2.5}$ and O_3. Concentration-response functions are typically estimated by gathering data on the occurrence of the health impacts, and running regressions that relate them to population-weighted concentrations of air pollutants, controlling for factors such as temperature, relative humidity, wind speed or season.

To obtain projections of the impacts of air pollution on health, it is also necessary to understand future levels of exposure. Information is needed on population projections, as well as the demographic structure of the population and its expected development over time. The calculation of health impacts has been done based on UN's demographic and population projections (2012), in line with the data used for the ENV-Linkages baseline and the OECD's long-term macroeconomic projections (OECD, 2014c).

For the base year, 2010, the impacts of $PM_{2.5}$ on mortality assessed in this study are based on the results of Forouzanfar et al. (2015) and Brauer et al. (2016).[5] Effects of ozone on mortality in 2010 are based on the earlier results of Lim et al. (2012) and Burnett et al. (2014). While updated results for the health impacts of ozone are available in Forouzanfar et al. (2015), the results in this report are based Lim et al. (2012). Given the dominance of the impacts of PM2.5 using older estimates for ozone only marginally affects the total results on the total costs of outdoor air pollution calculated in this report.

Forouzanfar et al. (2015) adopt a non-linear response function for PM mortality, with the rate of increase of mortality declining as PM concentrations rise (see Box 2.2 for an overview of these Global Burden of Disease studies). This assumption has been followed to generate lower projections of mortality. Upper projections are based on a linear relationship between mortality and concentrations. The use of a range recognises potentially significant uncertainty in the development of the non-linear relationship.

Quantification of morbidity effects requires different data, including the concentration-response relationship, the size of the population at risk, and the prevalence of morbidity. As this level of information was available for only a small number of countries, the quantification of morbidity effects is based on extrapolation of the results of studies performed for the Clean Air Policy Package of the European Commission (Holland, 2014a; European Commission, 2013) where the HRAPIE recommendations of WHO (2013b) were implemented as multipliers on the all-cause mortality from pollutant exposure. To ensure consistency, a correction was applied to account for differences between quantified all-cause deaths from Holland (2014a) and cause-specific mortality estimates from Forouzanfar et al. (2015). Ideally changes in behaviour (e.g. in diet, smoking habits, etc.), social changes (e.g. healthcare and employment) and medical changes (e.g. changes in healthcare systems and in treatment of diseases) over time and between world regions, should be factored into the analysis, but this is not possible owing to lack of data at global level. Further details on the methodology used to calculate health impacts are presented in Annex C.

Crop yield changes have been estimated following the methodology described in Van Dingenen et al. (2009). Crop losses for rice, wheat, maize and soybean are calculated in TM5-FASST based on concentrations of ozone during the growing season.[6] Gridded growing season and crop yield data are obtained from the Global Agro-Ecological Zones (GAEZ, version 3) (FAO/IIASA, 2012). For wheat and rice, growing season data are available for different varieties (spring wheat, winter wheat/dryland rice, wetland rice); however, yield data are provided for total wheat only. For maize and soybean, only one growing season dataset is available. Yield losses have been calculated assuming either that all wheat is spring wheat or that it is all winter wheat. The same assumptions have been taken for rice. The calculations in this report have been made with average values between the two assumptions on crops being all spring or all winter; a sensitivity analysis is presented in Chapter 4. It should be acknowledged that the projected crop yield changes are less robust than the projections of health impacts, owing to a much smaller underlying scientific literature. To ensure consistency with the crop yield projections in ENV-Linkages, the crop yield changes are expressed as a percentage change from the ENV-Linkages no-feedback projections.

Box 2.2. **The Global Burden of Disease studies**

The Global Burden of Diseases, Injuries, and Risk Factors Study (GBD) provides a methodology to quantify health loss from hundreds of diseases, injuries, and risk factors. GBD is the largest and most comprehensive effort to date to measure epidemiological levels and trends worldwide (www.healthdata.org/gbd).

The GBD initiative dates back to the early 1990s, when the World Bank commissioned the original GBD study (World Development Report 1993: Investing in Health). GBD work was institutionalised at the World Health Organization (WHO), and the organisation continued to update GBD findings.

The next comprehensive GBD update, the Global Burden of Diseases, Injuries, and Risk Factors Study 2010 (GBD 2010) published new estimates for the complete time series from 1990 to 2010 and an explanation of its methods in The Lancet in December 2012 (see Lim et al., 2012). While earlier work had been conducted mainly by researchers at Harvard and the WHO, GBD 2010 brought together a community of nearly 500 experts from around the world in epidemiology, statistics, and other disciplines.

With the Institute for Health Metrics and Evaluation (IHME) as the co-ordinating centre for an international network of GBD contributors, the entire time series of GBD estimates is being updated regularly to provide detailed information on population health (IHME, 2015). The first update, GBD 2013 (see e.g. Forouzanfar et al., 2015 and Brauer et al., 2016), expands the methodology, datasets, and tools used in GBD 2010 and presents estimates of all-cause mortality, deaths by cause, years of life lost, years lived with disability, and disability-adjusted life years by country, age and sex. GBD 2013 produced estimates for 323 diseases and injuries, 67 risk factors, and 1 500 sequelae for 188 countries. It reflects the work of more than 1 000 researchers in more than 100 countries.

Crop yield changes for those crops that are not covered by the calculations with TM5-FASST are projected using the information in Mills et al. (2007), following the methodology of e.g. Chuwah et al. (2015): yield changes for these crops are based on their relative sensitivity to ozone as compared to rice. For instance, Mills et al. find that sugar is roughly 1.5 times as sensitive as rice, and thus for each region the projected yield impacts in ENV-Linkages are also assumed to be 1.5 times those of rice. While necessarily very crude, this approach ensures that all crops are covered and avoids major distortions in the projections that might result from missing data.

2.6. Unit values for the analysis of health impacts

The valuation of the health impacts of outdoor air pollution includes both mortality and morbidity. Total health costs can be calculated by multiplying the impacts for each endpoint considered (e.g. number of hospital admissions, cases of illness, and premature deaths) by appropriate estimates of the unit value of each impact (e.g. the economic value of a hospital admission, a case of illness, and a premature death).

Different techniques are available to establish unit values. They can be estimated through a cost-of-illness approach and/or through direct monetary valuation techniques such as stated preference (SP) or revealed preference (RP) methods to assess the willingness-to-pay (WTP) to reduce environmental risks. Cost of illness and direct valuation techniques are often used in different contexts. The cost-of-illness approach is generally used in cost-effectiveness analysis (CEA) in order to provide an economic rationale for the rationing of health care resources in specific policy or programme proposals. In this instance, the benefits of investing in such resources are expressed in terms of the number of cases of illness avoided, or an index such as the number of "quality adjusted life years" (QALYs) gained. In contrast, WTP measures of these benefits are often used by economists in cost-benefit analysis (CBA), where the total costs and benefits of projects and policy proposals can be compared using a common money metric.

The cost-of-illness approach estimates the societal burden of disease by quantifying all costs related to illness that can be linked to market or financial transactions. These include "direct costs" (e.g. healthcare costs, expenditures in medicines and medical supplies) and "indirect costs" (e.g. the value of lost productivity because of reduced working time). The cost-of-illness approach does not take into consideration any of the costs that do not have a marketable or tradable value, such as the costs of pain and suffering. Using this approach disregards a potentially significant part of the loss to people related to mortality and morbidity. For example, using a cost-of-illness approach, a premature death would be evaluated with the future production potential of the deceased person, hence ignoring other aspects of premature death and the associated monetary values.

Stated and revealed preference techniques, by contrast, usually aim at estimating the welfare costs of illness or risk of premature death, often focusing on the non-market costs. SP methods (such as contingent valuation or choice modelling) rely on surveys to ask respondents for their WTP to reduce their mortality risk. RP methods use market behaviour to reveal individual preferences. In particular "hedonic pricing" methods are based on individuals' behaviour in markets where prices reflect differences in mortality risk (e.g. a labour market, where wages reflect differences in workplace mortality risks), and "averting costs" methods are based on markets for products that reduce mortality risks (e.g. buying motorcycle helmets to reduce mortality risks in traffic accidents).

Both SP and RP methods have their strengths and weaknesses, but there has been a growing emphasis on SP methods in recent years (OECD, 2012), especially in the context of environmental impacts. While these techniques are very useful in the evaluation of total economic costs of health or environmental impacts, they are generally not as accurate as cost-of-illness estimates. For example, stated preferences techniques – based on the responses to surveys – potentially introduce a number of biases and difficulties. One of the main difficulties to consider when using results from SP surveys is that respondents to surveys on the willingness to pay for a reduction in the risk of dying prematurely, may have different background or initial risks (i.e. the perceived risk of "dying anyway"). Providing informing factors can limit this issue. Perhaps the main potential bias is that responding to a survey does not involve a real commitment to pay what is stated in the survey – it is hypothetical only. For a comprehensive overview of the characteristics and shortcomings of the valuation literature see OECD (2006) and OECD (2012).

While the cost-of-illness and direct valuation approaches both aim to associate an economic cost to episodes of illness, the estimates of the two methodologies largely differ, as they measure two different aspects of the same issue. An example of the difference in measurement is provided by Chestnut et al. (2006), who estimate the economic benefits of reducing respiratory and cardiovascular hospitalisations based on both cost-of-illness and SP. The WTP estimates indicate that individuals value prevention of a five-day hospitalisation event at an average of approximately USD 2 400, while the average total cost-of-illness estimates per hospitalisation are USD 22 000-39 000.

Combining the two methodologies poses challenges in terms of double counting and comparability of estimates, but it can also help better assess the full societal costs of air pollution. Stieb et al. (2002), combine empirical data on the duration and severity of episodes of cardiorespiratory disease with cost-of-treatment, lost productivity, and WTP to avoid acute cardiorespiratory morbidity outcomes linked to air pollution.

Willingness-to-accept (WTA) is an alternative technique to WTP to attribute monetary values to mortality and to the disutility of illness. Using WTA generally provides larger estimates (Horowitz and McConnell, 2002), in part because the respondents to a WTA survey are not bounded by income. This means that, especially in the context of mortality risk valuation, respondents to surveys could provide unrealistically large values. Further, there is often a large share of "don't know" and protest responses when respondents are asked to accept an increase in mortality risk (OECD, 2012). OECD (2006) presents a detailed comparison of both concepts, and provides theoretical and practical reasons for using WTP.

Establishing unit values for mortality

The valuation of mortality impacts in this report relies solely on results from SP studies. In particular, it is based on estimates of the "value of a statistical life" (VSL) (see Box 2.3 for a discussion on valuing premature deaths due to air pollution). This is a long-established metric, which can be quantified by aggregating individuals' WTP to secure a marginal reduction in the risk of premature death over a given timespan (see OECD, 2012 and OECD, 2014a). Using solely direct monetary valuation means that certain indirect costs related to premature deaths are possibly not considered. Respondents to surveys on the risks of dying prematurely are unlikely to consider costs such as those related to the economic repercussions of lost productivity on the economy (for the working population). These are, however, likely to be a minor component of the value that can be associated to the premature death of an individual.

Box 2.3. **Valuing premature deaths with the value of a statistical life**

One of the most common procedures to value risks to life in standard economic theory is the value of a statistical life (VSL) (OECD, 2006). The VSL is derived from aggregating individuals' WTP to secure a marginal reduction in the risk of premature death over a given timespan.

The VSL is most commonly elicited through stated preference techniques, although revealed preferences techniques are also used. Alberini et al. (2016) provides an overview of the different methodologies used to elicit the VSL as well as their characteristics and shortcomings.

Box 2.3. **Valuing premature deaths with the value of a statistical life** *(continued)*

OECD (2012) describes the basic process for deriving a VSL from a state preference survey. Suppose the survey finds an average WTP of USD 30 for the reduction in annual risk of dying from air pollution from 3 in 100 000 to 2 in 100 000. This means that each individual is willing to pay USD 30 to have this 1 in 100 000 reduction in risk. In this example, for every 100 000 people, one death would be prevented with this risk reduction. Summing the individual WTP values of USD 30 over 100 000 people gives the VSL – USD 3 million in this case.

It is important to emphasise that the VSL is not the value of an identified person's life, but rather an aggregation of individual values for small changes in risk of death (OECD, 2012). As such, the total economic cost of the impact equals the VSL multiplied by the number of premature deaths; the economic benefit of a mitigating action becomes the same VSL multiplied by the number of lives saved (OECD, 2014a).

One large debate in the use of VSL is how the age of individuals matters in relation to different risk contexts. The same VSL is easily applicable in contexts in which the risk of premature deaths is reduced to the same extent for populations of all ages. In cost-benefit analysis exercises for policies that specifically focus on children's health, it is preferable to use specific values to evaluate the policy benefits for children (OECD, 2010). There are, however, difficulties in establishing child-specific VSL values since it is not possible to use surveys to elicit children's own preferences and biases, such as altruism, may arise when adults are asked to value risks for their children. In cases of evaluation of regulations targeted to reducing children's health risks, OECD (2012) and Lindhjem and Navrud (2008) suggest that VSL for children should be a factor of 1.5-2.0 higher than adult VSL. Air pollution is found to lead to premature deaths mostly of elderly people and, to a smaller extent, of children (WHO, 2014). Nevertheless, mortality risks, which are the ones considered in this report, mostly affect the elderly and the contribution from acute respiratory deaths in children (younger than 5 years of age) is very small. An adjustment is therefore not needed in the calculations of this report.

Age can also be taken into consideration by using the "value of a life year lost" (VOLYs), sometimes described as "value of a statistical life year" (VSLY). This technique calculates the number of "years of life lost" (YOLLs) owing to a specific risk and based on an estimated life expectancy, and then evaluates them by multiplying them by the VOLY. One issue with this technique is that the combination of counting YOLLs, rather than lives lost, means that the VOLY approach "explicitly places a lower value on reductions in mortality risk accruing to older populations with lower quality of life" (Hubbel, 2002). While there is a general agreement that children's health risks should ideally be valued differently, there is little support for the differentiation for adults of different ages. Further, VOLYs are rarely derived from surveys (Hunt, 2011). There are also major complications in the robust estimation of YOLLs, and the extent to which existing country-specific life expectancy values can and should be used. YOLLs can be calculated using country-specific life tables which are provided by the UN World Population Prospects (UN, 2015), although this requires elaborated calculations to obtain YOLLs for all world regions. The Global Burden of Disease studies define YOLLs as the difference between the age at death minus the global "longest possible life expectancy" (www.healthdata.org/gbd/faq) when calculating the numbers of years gained by avoiding a premature death. Using this assumption implies that especially in countries that currently have relatively low life expectancy, the total number of YOLLs will be greatly overestimated. In such cases it is possible that the valuation of premature deaths through a large number of VOLYs become significantly larger than when using VSL. Nevertheless, costs of premature deaths are usually higher when calculated using VSL. Given the limitations of the use of VOLYs, and following OECD (2012, 2014a), in this report the premature deaths are evaluated with the same VSL for all age groups.

Box 2.3. **Valuing premature deaths with the value of a statistical life** *(continued)*

One further issue when using VSL in the context of health impacts caused by air pollution is latency, namely the difference between time of exposure and the actual impact (premature death). The effect of latency on WTP is theoretically undetermined (OECD, 2012). Economic theory is usually based on the principle that people discount the future at a positive rate. Their utility will also vary with different periods of life in a way that can make WTP to reduce future mortality risks higher than their WTP to reduce immediate risks (see e.g. Hammitt and Liu, 2004). The meta-analysis in OECD (2012) was used to study whether VSL estimates systematically vary with different characteristics of the valuation methodology employed, characteristics of the change in mortality risk (e.g. type of risk, latency, cancer risk etc.), socio-economic characteristics of the respondents and other variables. Based on the literature review and the meta-analysis, OECD (2012) concludes that no adjustments should be made for latency in base VSL values.

OECD (2014a) provides country-specific VSL values for adults for OECD Member countries and some non-OECD economies, while OECD (2014b) does so for countries in the South and South East Asia region. As this report has global coverage, it was necessary to calculate VSL values for countries not covered by previous OECD studies. This was done using the benefit transfer methodology based on average national income, as outlined in OECD (2012) and detailed in Box 2.4. The key parameter in this methodology is the elasticity of income, which determines the extent to which the VSL changes according to different income levels. In this report, the income elasticity used for the calculations if 0.8 high income countries, 0.9 for middle-income countries and 1 for low-income countries. To analyse the sensitivity of the results to the chosen values of the income elasticity, alternative elasticity values are considered (see Section 5.1).

Box 2.4. **Benefit transfer for the value of a statistical life**

OECD (2012) provides a methodology to calculate country-specific VSL based on average national income through a benefit transfer methodology. In units of 2005 USD, the indicated range for OECD countries is USD 1.5-4.5 million, and the recommended base value is USD 3 million. Reference VSL values for OECD in 2005 are obtained from a rigorous meta-analysis of VSL studies (OECD, 2012). Starting with 1 095 values from 92 published studies, OECD-recommended VSL values were calculated for an average adult.

As argued in OECD (2006 and 2014a), WTP varies with income and income is one of the main indicators used in preference-based technique for measuring VSL. Country-specific VSL values are calculated starting from a reliable estimate for a specific region, in this case the OECD base value of USD 3 million, and then adjusting the VSL for other countries based on income levels. The use of a local VSL reflects the situation that the valuation is done in a specific country; this is appropriate as both the costs and benefits of air pollution and pollution control policies are largely within the same region (OECD, 2014a). This is in contrast to e.g. climate change, where the use of a different VSL for mortality in different countries is very controversial, as the beneficiaries of a policy are largely located in other countries (as greenhouse gases are uniformly mixing in the atmosphere).

Box 2.4. **Benefit transfer for the value of a statistical life** *(continued)*

Several studies attempt to evaluate the income elasticity of the WTP to reduce the risk of premature death. The meta-analysis in OECD (2012) finds that the income elasticity is in the range of 0.7-0.9 for OECD countries, with significantly higher income elasticities for countries in the bottom 40th percentile of income. Longitudinal studies provide additional evidence that WTP varies at different stages of economic development (Hammitt and Robinson, 2011). In particular, the range proposed in OECD (2012) has been judged to be too low for low income countries as using such values would imply unrealistically high WTP values for these countries. Given this evidence, this report uses an elasticity of 0.8 for high-income countries, 0.9 for middle-income countries and 1 for low-income countries (country groups are distinguished using the World Bank income thresholds).

This benefit transfer methodology is used to adapt VSL to other countries, but also to estimate its growth over time. As argued in OECD (2006), income should be used as the reference variables also to adapt WTP over time, so as to avoid situations in which for instance the WTP to save a statistical life rises faster over time than the rate of inflation. Existing studies, such as Costa and Kahn (2004) who calculate the VSL changes in the US for the period 1940-80, find that VSL rises over time as income rises.

Country- and year-specific VSL is calculated following this formula:

$$VSL_r^t = VSL_{OECD}^{2010} \cdot \left(\frac{Y_r^t}{Y_{OECD}^t}\right)^{\beta}$$

where:

Y is the average income (GDP per capita) of country r in year t expressed in 2010 USD PPP;

β is the income elasticity of VSL. It measures the percentage increase in VSL for a percentage increase in income.

This methodology is applied in this analysis to obtain VSL values for the all countries in the world, as well as for the projections to 2060. The extrapolations are based on the projected country-specific income values. The income projections used are the same as those used to calibrate the ENV-Linkages model: IMF Economic Outlook (2014) until 2017 and then on the economic projections of the ENV-Growth model (Dellink et al., 2016).

Establishing unit values for morbidity

The valuation of morbidity in this report combines a separate evaluation of cost-of-illness (healthcare and labour productivity costs) and welfare costs.[7] The literature on the costs associated with air pollution effects on the demand for healthcare is very sparse compared with that which seeks to provide estimates of the overall economic cost of air pollution on health. Discussion in the literature often misrepresents estimates of total cost as being the costs of healthcare or of healthcare and productivity losses, when these are only a few components of the total costs of outdoor air pollution. For the purpose of this work, the term “healthcare costs” is specific to the costs incurred in treating illnesses, while the costs of discomfort, pain and suffering related to illness are referred to as “disutility costs” or “welfare costs”.

As already discussed, healthcare costs can be evaluated using the cost-of-illness approach. While data availability is certainly an issue, a quantification of healthcare costs is at least theoretically straightforward, as they are linked to market transactions and thus have established, observable prices.[8] Nevertheless it is not easy to establish a reference unit value for healthcare costs, as they vary substantially across different countries, owing to the differences in healthcare systems, but also in the way people face illness. Even within the same continent there can be large variations. For instance, healthcare costs for chronic bronchitis have been estimated in a series of European studies using similar methods to be EUR 530/patient/year in France (Piperno et al., 2003) but EUR 3 238/patient/year in Spain (Izquerdo, 2003). Unit values for healthcare expenditures have been established for the OECD based on Holland (2014a). Country-specific unit values are then calculated based on the relationship between healthcare expenditure and GDP per capita, using the World Bank's 2015 total healthcare expenditure as a percentage of GDP at the national level (World Bank, 2015).

Welfare costs, which include the disutility costs of illness related for example to pain and suffering, are evaluated using available results on WTP from SP studies. In particular, this report uses the values calculated for the European Commission (Holland, 2014a) as a starting point to establish unit values for the welfare costs of the morbidity endpoints. Extension of morbidity welfare costs to specific countries uses benefit transfer based on income, as for mortality costs (see Box 2.4). There is a potential bias in transferring estimates of the disutility of morbidity from existing studies, mostly developed in Europe, to the global context. Preferences on health and the valuation of illness can greatly vary between different countries. For example, Ready et al. (2004) illustrate that using international transfer of unit values in the evaluation of the benefits of specific health impacts introduced a transfer error even between European countries. In the context of a global study, however, benefit transfer is the only available technique, as the availability of valuation studies on the impacts of air pollution are only focused on a few areas of the world.

Resulting unit values

The unit values used are presented in Table 2.2 for each health endpoint, including a breakdown to the different cost elements (welfare and healthcare costs).[9] The value used for mortality is USD 3 million, following OECD (2014a). The morbidity values are established based on (Holland, 2014a).

Table 2.2. **Unit values used in the analysis of the health impacts**

USD, 2005 PPP exchange rates

Effect	Cost element	Value
Mortality, premature deaths	Welfare cost	3 million
Chronic bronchitis in adults (new cases)	Welfare cost	61 610
	Healthcare cost	13 070
Bronchitis in children (cases)	Welfare cost	680
	Healthcare cost	57
Equivalent hospital admissions (respiratory and cardiovascular diseases)	Welfare cost	575
	Healthcare cost	3 430
Restricted activity days	Welfare cost	106
Minor restricted activity days (asthma symptom days)	Welfare cost	48

Note: Values are for the OECD. They are unit values and as such they refer to costs per statistical life, case of illness, hospital admission and day with restricted activity.

Source: Own evaluation based on Holland (2014a).

The methods adopted leave little potential for double counting of the different elements of valuation of morbidity as costs are fully attributed to the main cost component. Mortality is only associated with welfare costs and it is not included in the modelling analysis of market costs. Unit values for chronic bronchitis in adults, bronchitis in children and hospital admissions are established for both welfare and healthcare costs. While respondents to surveys that are used to derive welfare costs may also consider some of the market costs in their answers, the largest share of the costs are likely to be related to non-market costs. The SP studies used to establish the unit values for the disutility of illness have been conducted in countries with well-functioning public health care systems, which reduce the risk of respondents including components other than disutility when they state their WTP for avoiding clearly specified episodes of illness. The values used should therefore reflect different and complementary aspects of illness.

For lost working days, the assumption is that the main impact is reduced productivity, while for (minor) restricted activity days discomfort is assumed to dominate. Hence, lost working days impact labour productivity and are included in the calculations of market impacts with the ENV-Linkages model. Costs associated with (minor) restricted activity days on the contrary are assessed through their welfare costs. Annex C further discusses double counting issues.

Once the unit values are established, the overall healthcare costs, welfare costs of illness and of mortality can be calculated by multiplying the number of cases of illness and of premature deaths by the unit values (see Section 2.8). The overall costs are therefore an aggregate of average individual costs for the affected individuals. The overall healthcare costs are used as an input to the ENV-Linkages model to calculate the market costs of outdoor air pollution. The market costs then include direct costs related to the overall health expenditures as well as indirect costs originating from the repercussions on consumption, savings, production and other economic activities (see Section 2.7).

2.7. From impacts to consequences for economic growth

The market impacts are modelled directly in ENV-Linkages following a production function approach. This means that market impacts are not assumed to only affect macroeconomic variables such as GDP, but to directly affect specific elements in the economic system, such as labour productivity or land productivity. The impacts are thus modelled as changes in the most relevant parameters of the production function underlying the model structure. The resulting changes in the economy (both at sectoral and macroeconomic level) are expressed as percentage change with respect to the projection without feedbacks to the economy (cf. Section 2.2). They are calculated for each time period up to the time horizon (2060) and thus reflect the annual economic consequences that result from the stream of impacts over time. The scenario which includes the market impacts from air pollution is referred to as the *central projection*.

Three market impacts are included in the model: changes in health expenditures due to increased incidence of illnesses, changes in labour productivity due to increased incidence of illnesses, and changes in agricultural crop yields. Table 2.3 summarises the impacts modelled and the data sources.

Changes in health expenditures are implemented in the model as a change in demand for the aggregate non-commercial services sector. The amount of additional health expenditures introduced in the model is calculated multiplying the number of cases of illnesses and of hospital admissions by the unit values for healthcare specified in Section 2.6. It is assumed that the additional health expenditures affect both households and government expenditures on healthcare.[10] The extent to which households or governments are affected depends on regional characteristics of the health system in terms of their relative contribution to healthcare. The distinction between households and government expenditures has been done using World Bank data on the proportion of healthcare expenditures paid by households and by the government (World Bank, 2015). A close relationship is noted between healthcare expenditure and GDP per capita for all but a few countries (World Bank, 2015), facilitating extrapolation of data on specific health endpoints between countries.

Table 2.3. **Air pollution impacts included in ENV-Linkages**

Impact categories	Impacts modelled	Data sources
Health	Changes in health expenditures due to changes in incidences of bronchitis, respiratory and cardiovascular diseases, etc.	Calculations based on Holland (2014a) and on results from the Global Burden of Disease studies (Forouzanfar et al., 2015, and Brauer et al., 2016 for PM; Lim et al., 2012, and Burnett et al., 2014 for ozone).
	Changes in labour productivity due to lost working days caused by changes in incidences respiratory and cardiovascular diseases.	
Agriculture	Changes in crop yields	Calculations by the EC-JRC Ispra with the TM5-FASST model (Van Dingenen et al., 2009).

Changes in labour productivity are directly implemented in the model as percentage changes in the regional productivity of the labour force. Productivity losses are calculated from lost working days, following the methodology used in Vrontisi et al. (2016), using assumptions on the average number of work days per year in each region (World Bank, 2014). The approach to reduce labour productivity rather than labour supply is more appropriate when the dominant effect of the illness is to reduce average output per worker, rather than total labour costs borne by employers. This holds especially when employees are compensated for sick leave, or when workers show up to work while being ill (presenteeism).

Changes in crop yields are implemented in the model as a combination of changes in the productivity of the land resource in agricultural production, and changes in the total factor productivity of the agricultural sectors. This specification, which is in line with OECD (2015), mimics the idea that agricultural impacts affect not only purely biophysical crop growth rates but also other factors that affect output, such as the effectiveness of other production inputs. Air pollution affects crop yields heterogeneously in different world regions, depending on the concentrations of ground level ozone.

Once impacts on crop yields, health expenditures and labour productivity have been included in ENV-Linkages, the model is used to calculate the macroeconomic costs of air pollution in the central projection. These costs are the result of the direct market costs as well as the adjustment processes that take place in the model (indirect market costs). For instance, an increased demand for healthcare may result in a lower demand for other services, while changes in crop yields for certain crops may result in changes in production of other substitute crops and even other sectoral activities as well as changes in trade patterns.

2.8. From impacts to welfare costs

The last step in the analysis is the assessment of the welfare costs of outdoor air pollution. For non-market costs related to health impacts, these are calculated by multiplying the results related to the relevant health endpoints (step 4) with the appropriate unit values (step 5). More precisely, for mortality, the number of premature deaths is multiplied with the value of a statistical life (VSL). Similarly, for each of the morbidity endpoints, the results are multiplied with the corresponding unit value to calculate the welfare costs related to the disutility of illness.

The analysis of the economic consequences of the market impacts is done with a focus on the most common indicator of economic activity, GDP. This is also the reference point to investigate the consequences of air pollution on economic growth. The market costs are also expressed in terms of welfare to facilitate the comparison with other cost components. This is done using the equivalent variation of income, which is a common measure of welfare impacts of a shock in a general equilibrium framework. It measures the change in income that, at initial prices, would have the same welfare effect as the changes induced by the shock to the system (Hicks, 1939). Thus, the welfare costs of market impacts are represented as a change in income, in constant USD. The equivalent variation represents the maximum willingness to pay to avoid the deterioration in the welfare of consumers (this is known in the economics literature as Hicksian equivalence).

Finally, market and non-market welfare costs can be compared and aggregated to provide an assessment of total welfare costs. Having different methodologies to calculate market and non-market costs complicates the possibilities to aggregate numbers. However, market and non-market costs can be added when both are expressed in terms of aggregate income losses, and using the same metric, i.e. constant 2010 USD using PPP exchange rates.

One further issue comes with aggregating welfare costs across countries and regions. In principle, equity weights can be used to create a social welfare function that affect how a trade-off between welfare changes in different countries is measured. Such weights could be used in establishing VSL and morbidity values for developing countries, and in the welfare measures used in the general equilibrium model. The effect of welfare weights is that they provide a "fairer" measure of the global social welfare associated with the welfare costs presented in this report; they would also reflect that the marginal utility from an additional unit of income is larger in poorer countries than in rich countries. However, this report abstains from introducing welfare weights for two reasons. First, the aim of the report is not to find a socially optimal level of pollution; rather, it aims at highlighting the regional consequences of unmitigated outdoor air pollution. Although the regional results are sometimes aggregated to a global total, that is purely for illustrative purposes. A second reason not to adopt equity weights is that these reflect essentially a moral judgement and it is extremely difficult to find appropriate welfare weights that would be uncontroversial. Finally, equity weighting introduces a new level of complexity in the results that is avoided by focusing on the results expressed in terms of *income* changes.

Notes

1. In principle, the feedback effects will affect emission levels and thus one should iterate between the central projection and the no-damage projection until consistency is reached on the level of emissions. This iterative process is, however, very computationally expensive, and only relevant when the emission levels in the central projection are significantly different from those in the no-damage projection.

2. This dataset has also been used as a basis for the Energy Modelling Forum (EMF) 30 model comparison exercise, whose output has been used to check on the robustness of the implementation of the air pollutants in the ENV-Linkages model.

3. Ideally, for transport, it would be better to consider fuel use per kilometre or passenger, but the ENV-Linkages model does not include such details.

4. The reduced-form version TM5-FASST mimics the full set of chemical, physical and meteorological processes represented in TM5-CTM, for the meteorological year 2001. They represent the formation of secondary ammonium sulphate and nitrate from SO_2, NO_x and NH_3 emissions, the formation of O_3 from NO_x and VOC and the transport and wet and dry removal of all pollutants from the atmosphere.

5. By building on the GBD studies, the implicit weaknesses of those studies are included also here. For instance, there may be a risk that interactions between air pollution and tobacco smoking are not adequately addressed in attributing mortality to outdoor air pollution. Nonetheless, the GBD studies provide the most robust and comprehensive information available for assessing the impacts of air pollution on mortality at a global level.

6. Rice, wheat, maize and soybean represent more than half the total volume of global agricultural production, but less than half of the value.

7. It is also possible to distinguish the morbidity costs of the health impacts of air pollution into *(i)* resource costs, which are represented by the direct medical and non-medical costs associated with treatment for the adverse health impact of air pollution plus expenditures on averting behaviour; *(ii)* opportunity costs, which are associated with the indirect costs related to loss of productivity and/or leisure time due to the health impacts; and *(iii)* disutility costs, which refer to the pain, suffering, discomfort and anxiety linked to the illness. The analysis of this report covers each of these three types of impacts at least partially, as resource costs relate to health expenditures, opportunity costs to labour productivity changes and disutility costs are included in the welfare cost evaluation.

8. For regions where healthcare costs cannot be directly assessed, results for other regions have been extrapolated.

9. For consistency with original sources, the figures in this table are given in 2005 USD. These have then been converted to 2010 USD in the modelling framework. Results from the analysis are also presented in 2010 USD.

10. In reality, private sector business also plays a role in the supply of healthcare through employer-based insurance. These expenditures are not considered separately in the modelling framework. Further, an alternative assumption on governments and households, is that they could decide not to increase their health expenditures and accept a lower level of health care. Such a response will, however, likely result in larger welfare costs. The approach used here can therefore be seen as a lower bound for the health costs.

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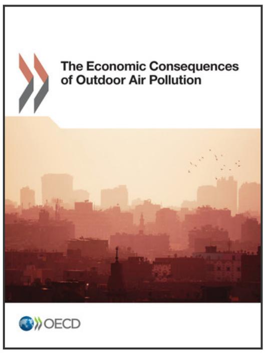

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