Economy-Wide Modeling: Benefits of Air Quality Improvements White Paper

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1 Introduction

Economy-wide analyses of tax and trade policy typically consider the benefits and costs of those interventions simultaneously. This is, in part, because of the important interactions between the two that help define the overall net impact on the economy and society. In these cases, it may be more natural or straightforward to consider the economy-wide benefits of policy interventions due to their market nature. However, the typical non-market nature of benefits associated with improved air quality does not imply that they have no impact on the economy or do not interact with the costs of realizing those improvements. While it has become commonplace in applied benefit cost analysis to assume that benefits are additively separable in households' utility functions and therefore do not affect other household choices, this assumption is at odds with much of the applied environmental economics research on the benefits of improved environmental quality. Many estimates of the benefits associated with environmental policies leverage changes in market activity that complements household consumption of non-market goods and services as environmental quality varies. For example, households' willingness to pay for environmental improvements are commonly estimated based on the way spatial and/or temporal variation in environmental quality affects housing prices due to its influence on housing demand (Palmquist, 2005). Similarly, studies leveraging observed housing market behavior in response to variation in environmental amenities, have estimated households' willingness to pay for improvements in health (Gayer et al., 2002). Benefits from improvements in environmental quality are also routinely estimated from consumers' willingness to trade-off other consumption for improved environmental quality in their recreation decisions (Phaneuf and Smith, 2005). Furthermore, improvements in environmental quality have direct effects on economic markets through improved labor and agricultural productivity (Zivin and Neidell, 2012; US EPA, 2011), amongst other direct impacts. Therefore, the benefits of improved air quality may have important economy-wide impacts and may interact with the cost of achieving those improvements, such that including both benefits and costs in an economy-wide analysis of some air regulations may be important for assessing the net welfare effects. To this end, the chapter on economy-wide computable general equilibrium (CGE) models in the Handbook of Environmental Economics states "if the [CGE] model is to be used for evaluation of policies, it should be capable of quantifying both the costs and the benefits of the policies in question" (Bergman, 2005).

Introducing the role of the environment in production and household behavior at an economy-wide scale is an area of applied analysis that is relatively new. However, the suite of studies that have partially incorporated the impacts of environmental quality in economy-wide CGE have found substantial economy-wide impacts of improvements in environmental quality (Matus et al., 2008; Carbone and Smith, 2013). In its analysis of the benefits and costs of the Clean Air Act Amendments (CAAA) from 1990 to 2020, EPA found including only part of the human health benefits associated with improved air quality, along with the compliance costs, in a CGE model dramatically changed the estimated economic and welfare impact of the regulations (US EPA, 2011). In its review of that analysis, the Advisory Council on Clean Air Compliance Analysis stated that inclusion of benefits in the economy-wide model specifically adapted for use in that study "represent[ed] a significant step forward in benefit-cost analysis." However, serious technical challenges remain when attempting to evaluate the benefits of a specific air regulation within an economy-wide model.

This paper aims to describe the main approaches used by EPA and the academic literature to estimate the economy-wide effects of improved air quality, highlighting potential areas of improvement and technical challenges that exist. The methodological approaches discussed could have implications for both adequately estimating the net economy-wide impacts and the benefits of air regulations. Before considering the incorporation of environmental quality into applied economy-wide modeling, this paper begins by providing background on the quantified and unquantified benefits of improved air quality. This is followed by a discussion of the methods EPA typically uses to monetize these changes in benefits of air regulations as a point of comparison for the economy-wide approach. The paper then reviews the studies conducted by EPA and outside researchers that have incorporated, at least partially, environmental quality into computable general equilibrium (CGE) models to estimate economic and welfare impacts. This work is then examined in the context of the types of benefits typically associated with improvements in air quality and the theoretical and empirical literature on benefits to assess key challenges and potential areas of improvement. The paper concludes with a discussion of the importance of capturing spatial heterogeneity when estimating the benefits of changes in air quality and how that may affect the incorporation of benefits into economy-wide assessments.

2 Air Quality Benefits

To provide context for the later discussions in this paper regarding the incorporation of environmental quality into economy-wide modeling, this section provides a basic overview of the beneficial impacts of improved air quality and the methods that EPA typically uses to estimate the social benefits of improvements in air quality for a regulatory analysis. It begins by summarizing the general categories of benefits associated with improving air quality, including effects that are typically quantified in contemporary benefit cost analyses (BCAs) and those that are not quantified but potentially could be if improved data and methods are developed in the future. The section then provides a general overview of the typical valuation framework that EPA employs, the methods it uses in performing benefits analysis, and an example illustrating how results are often presented, including the characterization of uncertainties.

2.1 Air Quality Benefits Traditionally Included in Benefit-Cost Analyses

Improvements in air quality may affect a wide variety of endpoints, including those related to human health and others such as visibility, materials damage, and agricultural productivity. In recent EPA regulatory analyses (US EPA, 2012) and its analysis of the benefits and costs of the Clean Air Act from 1990 - 2020 (US EPA, 2011), EPA typically quantifies and monetizes a number of health effects, listed in Table 1. Table 1 also lists health effects that, while potential benefits of air regulations, are not yet quantified in EPA regulatory analyses. Table 2 provides an analogous table of quantified/monetized and unquantified non-health benefits.

Some categories of human health benefits, such as changes in the incidence of adult or infant mortality or of heart attacks, are more easily quantified and monetized due to the availability of scientifically defensible concentration-response functions relating health outcomes to ambient pollution concentrations and estimates of the monetized value for marginal changes in health outcomes. The effects of other health outcomes, for instance changes in labor productivity due to a change in health status, are more difficult to quantify. Depending on the endpoint, limited assessments may be available for some subsets of affected populations (e.g., outdoor fruit pickers).

The monetized benefits of non-health impacts are typically more difficult to quantify due to several factors including: 1) lack of available concentration- or deposition-response functions; 2) complexity of ecosystem responses to pollutants; 3) heterogeneity in responses and dependence of responses to numerous geographically varying factors; 4) lack of functions to translate changes in ecological endpoints into changes in ecosystem services; and 5) lack of willingness-to-pay or other economic valuation estimates for many types of ecosystem services. In past regulatory analyses, EPA has estimated the willingness to pay for changes in visibility impairment and materials damages from reductions in $PM_{2.5}$ and foliar damage related to NO_x emissions. In some cases, such as acidification of streams, EPA has been able to quantify changes in stream chemistry and certain measures of fish health, but has not been able to monetize those changes.

2.2 General Framework for Benefits Valuation

EPA typically follows a "damage-function" or "effect-by-effect" approach to estimate total benefits of modeled changes in environmental quality when analyzing the effects of an air regulation (see for example Levy et al., 2009; Fann et al., 2012; and Tagaris et al., 2009).[1] In this approach, changes in individual endpoints (specific effects that can be associated with changes in air quality) are estimated and dollar values reflective of households' willingness to pay (WTP) are assigned to those changes. Total benefits are then calculated as the sum of the benefit estimates for all non-overlapping endpoints.

In some cases, the changes in environmental quality that are anticipated to result from an air regulation are associated with endpoints that are valued directly by individuals, such as visibility in scenic national parks. For such changes, it is relatively straightforward, at least conceptually, to link the pollution externality to human welfare.

In other cases, additional analysis must first be conducted to map changes in air quality to changes in endpoints more directly associated with human welfare. For example, EPA estimates how changes in ozone and particulate matter (PM) concentrations affect health conditions that people value directly, with health impact functions based on effect estimates in the epidemiology literature. Specifically, a health impact function measures the change in a particular health endpoint (e.g., hospital admissions, mortality rates) for a given change in air quality. A standard health impact function has four components: 1) an effect estimate or risk coefficient from a particular epidemiological study; 2) a baseline incidence rate for the health effect (obtained from either the epidemiology study or a source of public health statistics such as the Centers for Disease Control); 3) the affected population; and 4) the estimated change in the relevant air quality metric (e.g. annual mean $PM_{2.5}$).

[1] The damage function approach may provide a more bottom-up method of estimating total benefits in some cases than other methods such as a hedonic price approach applied to housing prices, which would require strong assumptions, such as perfect information, to fully characterize willingness to pay estimates.

Table 1: Human Health Effects of Air Pollutants Regulated by EPA

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized
Reduced incidence of premature mortality and morbidity from exposure to **$PM_{2.5}$**	Adult premature mortality based on cohort study estimates and expert elicitation estimates (age >25 or age >30)	✓	✓
	Infant mortality (age <1)	✓	✓
	Non-fatal heart attacks (age > 18)	✓	✓
	Hospital admissions—respiratory (all ages)	✓	✓
	Hospital admissions—cardiovascular (age >20)	✓	✓
	Emergency department visits for asthma (all ages)	✓	✓
	Acute bronchitis (age 8–12)	✓	✓
	Lower respiratory symptoms (age 7–14)	✓	✓
	Upper respiratory symptoms (asthmatics age 9–11)	✓	✓
	Asthma exacerbation (asthmatics age 6–18)	✓	✓
	Lost work days (age 18–65)	✓	✓
	Minor restricted-activity days (age 18–65)	✓	✓
	Chronic Bronchitis (age >26)	—	—
	Emergency department visits for cardiovascular effects (all ages)	—	—
	Strokes and cerebrovascular disease (age 50–79)	—	—
	Other cardiovascular effects (e.g., other ages)	—	—
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc.)	—	—
	Cancer, mutagenicity, and genotoxicity effects	—	—

Table 1: Human Health Effects of Air Pollutants [continued]

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized
Reduced incidence of mortality and morbidity from exposure to **ozone**	Premature mortality based on short-term study estimates (all ages)	✓	✓
	Premature mortality based on long-term study estimates (age 30–99)	✓	✓
	Hospital admissions—respiratory causes (age > 65)	✓	✓
	Hospital admissions—respiratory causes (age <2)	✓	✓
	Emergency department visits for asthma (all ages)	✓	✓
	Minor restricted-activity days (age 18–65)	✓	✓
	School absence days (age 5–17)	✓	✓
	Decreased outdoor worker productivity (age 18–65)	—	—
	Other respiratory effects (e.g., premature aging of lungs)	—	—
	Cardiovascular and nervous system effects	—	—
	Reproductive and developmental effects	—	—
Reduced incidence of mortality and morbidity from exposure to NO_2	Asthma hospital admissions (all ages)	—	—
	Chronic lung disease hospital admissions (age > 65)	—	—
	Respiratory emergency department visits (all ages)	—	—
	Asthma exacerbation (asthmatics age 4–18)	—	—
	Acute respiratory symptoms (age 7–14)	—	—
	Premature mortality	—	—
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—

Table 1: Human Health Effects of Air Pollutants [continued]

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized
Reduced incidence of mortality and morbidity from exposure to SO_2	Respiratory hospital admissions (age > 65)	✓	✓
	Asthma emergency department visits (all ages)	✓	✓
	Asthma exacerbation (asthmatics age 4–12)	✓	✓
	Acute respiratory symptoms (age 7–14)	✓	✓
	Premature mortality	—	—
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—
Reduced incidence of morbidity from exposure to methylmercury	Neurologic effects—IQ loss	—	—
	Other neurologic effects (e.g., developmental delays, memory, behavior)	—	—
	Cardiovascular effects	—	—
	Genotoxic, immunologic, and other toxic effects	—	—
Reduced incidence of morbidity from exposure to lead	Neurologic effects—IQ loss	✓	✓
	Increased diastolic blood pressure and sequelae	—	—
	Other neurobehavioral and physiological effects	—	—
	Delinquent and anti-social behavior	—	—
	IQ loss effects on compensatory education	—	—
	Hypertension	—	—
	Non-fatal coronary heart disease	—	—
	Non-fatal strokes	—	—
	Premature mortality	—	—
	Other cardiovascular diseases	—	—
	Neurobehavioral function	—	—
	Renal effects	—	—
	Reproductive effects	—	—

Table 2: Non-Health Effects of Air Pollutants

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized
PM2.5 atmospheric effects	Visibility impairment (includes effects from NOx and SOx on nitrate and sulfate PM)	✓	✓
PM2.5 atmospheric and deposition effects	Materials damage	✓	✓
	Climate	—	—
	Ecosystem effects (organics and metals)	—	—
NOx atmospheric effects	Vegetation injury (through ozone)	✓	✓
NOx atmospheric and deposition effects	Materials damage	—	—
	Climate	—	—
NOx deposition effects (includes effects of deposition of particulate nitrate)	Acidification (terrestrial and aquatic) plus associated ecosystem effects	✓	—
	Nitrogen enrichment (terrestrial and aquatic) plus associated ecosystem effects	—	—
	Secondary health effects (e.g. blue baby syndrome)	—	—
SOx atmospheric effects	Vegetation injury	—	—
SOx atmospheric and deposition effects	Materials damage	—	—
	Climate	—	—
SOx deposition effects (includes effects of deposition of particulate sulfate)	Acidification (terrestrial and aquatic) plus associated ecosystem effects	✓	—
	Mercury methylation	—	—
Mercury atmospheric and deposition effects	Ecosystem effects	—	—
	Mercury methylation and associated effects	✓	✓

A typical health impact function has the form:

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

where Δy is the change in the health endpoint, y_0 is the baseline incidence, equal to the baseline incidence rate times the potentially affected population, β is the effect estimate, and Δx is the estimated change in the air quality metric. Other functional forms are sometimes applied, but the basic elements remain the same. Upon mapping changes in emissions and the resulting air quality changes into endpoints relevant for human welfare, economic valuation methods are used to estimate people's WTP for such changes.

Reductions in ambient concentrations of air pollution generally lower the risk of future adverse health effects by a small amount for a large population. Epidemiological studies generally provide estimates of the relative risks of a particular health effect for a given increment of air pollution (e.g., often per 10 µg/m3 for $PM_{2.5}$). These relative risks can be used to develop health impact functions that relate a unit reduction in air emissions to changes in the incidence of a health effect given the baseline prevalence. To value these changes in incidence, WTP for changes in risk need to be converted into WTP per statistical incidence. This measure is calculated by dividing individual WTP for a risk reduction by the related observed change in risk. For example, suppose a measure is able to reduce the risk of premature mortality from 2 in 10,000 to 1 in 10,000 (a reduction of 1 in 10,000). If individual WTP for this risk reduction is $100, then the WTP for an avoided statistical premature mortality amounts to $1 million ($100/0.0001 change in risk). Using this approach, the size of the affected population is automatically taken into account by the number of incidences forecast by the health impact functions, which are applying results from epidemiological studies to the relevant population. The same type of calculation can also produce values for statistical incidences of other health endpoints.

For some health effects, such as hospital admissions, WTP estimates are generally not available. In these cases, EPA has used the cost of treating or mitigating the effect as a proxy. For example, when assessing the benefits of reduced hospital admissions EPA has used the avoided medical costs to monetize changes. This approach, sometimes referred to as cost-of-illness (COI) estimation, generally (though not necessarily in every case) understates the true value of reductions in risk of a health effect (Harrington and Portney, 1987; Berger, 1987). This is because it reflects the direct expenditures related to treatment ex post but not the value of avoided pain and suffering associated with the health endpoint, and therefore, not the full willingness to pay to avoid the illness ex ante.

In most cases, EPA relies on a benefit transfer approach to monetize health and non-health endpoints, instead of conducting an original revealed or stated preference study of its own. Benefits transfer takes primary research from the published literature (i.e. values or functional forms) that is deemed relevant and similar enough to the regulatory context being evaluated and adapts it to the policy being analyzed. For instance, when necessary, adjustments are made to account for the level of environmental quality change, the socio-demographic and economic characteristics of the affected population, and other factors to improve the accuracy and robustness of benefit transfer estimates. See EPA's *Guidelines for Preparing Economic Analyses* for more information on the particulars of benefits transfer.

2.2.1 Temporal Dynamics of Health Impacts

In assessing the benefits of air pollution regulations, reductions in premature mortality associated with chronic $PM_{2.5}$ exposure present unique challenges due to the importance of the temporal dynamics of exposure and health impacts. For premature mortality, there is a "cessation" lag between reduced $PM_{2.5}$ exposures and the total realization of changes in mortality risk. Although the structure of the lag is uncertain, EPA follows the advice of the Health Effects Subcommittee of the Advisory Council on Clean Air Compliance Analysis and assumes a segmented lag structure characterized by 30% of mortality risk reductions in the first year, 50% over years 2 to 5, and 20% over the years 6 to 20 after the reduction in $PM_{2.5}$ (US EPA, 2004a). To take this into account in the valuation of premature mortality reductions, the value in future years is discounted using rates of 3% and 7% based on OMB guidance (OMB, 2003). Discounting is also applied in the valuation stage for endpoints such as nonfatal heart attacks, where a portion of the costs associated with the endpoint may occur in years following the event.

2.2.2 Income adjustments

Economic theory suggests that the WTP for most goods will increase with real income (i.e., they are normal goods). There is substantial empirical evidence that the income elasticity of the WTP for marginal health risk reductions is positive, so as real income increases the WTP for environmental improvements that lead to a reduced negative health outcome also increases. However, there is uncertainty about the exact value of the income elasticity estimate. Although many analyses assume that the income elasticity of WTP is unit elastic (i.e., a 10% higher real income level implies a 10% higher WTP to reduce risk changes), some empirical evidence suggests that income elasticity may be less than one (i.e., inelastic) or greater than one (i.e. elastic).[2] If it is inelastic (elastic), this implies that, as real income rises the WTP value also rises but at a slower (faster) rate than real income.

In addition, there may be differences in income elasticities across the WTP for different health endpoints, for example, reflecting differences between acute and chronic conditions. Reported income elasticities show that WTP is more elastic for severe health conditions and mortality risk, and more inelastic for minor conditions (Kleckner and Neumann, 1999). As such, EPA uses different elasticity estimates to adjust the WTP for minor health effects, severe and chronic health effects, and premature mortality. In recent regulatory analyses, EPA has used elasticities ranging from 0.14 to 0.45 depending on the health endpoint.

2.3 Conducting and Presenting the Analysis

In a typical benefits analysis of an air quality regulation, health impacts are calculated at a relatively fine spatial scale (e.g. at a $12km^2$ grid resolution) and then aggregated to state or national totals for

[2] EPA's Environmental Economics Advisory Committee of EPA's Science Advisory Board noted that estimates of the income elasticity of the VSL range from 0.1 to 1.0 in cross-section analysis of stated preference survey results and between 1.3 and 3.0 in longitudinal studies of hedonic wage results (US EPA, 2011 p 21).

Table 3: Monetized $PM_{2.5}$ Mortality Risk Reduction Benefits for the Revised and Alternative Annual Primary $PM_{2.5}$ Standards (Incremental to Analytical Baseline) (Millions of 2006$, 3% discount rate)

	Revised and Annual Standards (95th percentile confidence interval)		
Health Effect	**13 µg/m³**	**12 µg/m³**	**11 µg/m³**
Krewski et al. (2009) (adult mortality age 30+)	$1,300 ($120--$3,500)	$4,000 ($370--$11,000)	$13,000 ($1,200--$35,000)
Lepeule et al. (2012) (adult mortality age 25+)	$2,900 ($250--$8,100)	$9,000 ($800--$26,000)	$29,000 ($2,600--$82,000)
Woodruff et al. (1997) (infant mortality)	$3.4 ($0.29--$10)	$11 ($0.91--$32)	$35 ($3.0--$100)

Notes:

1. Reproduced from Table 5-19 of the Regulatory Impact Analysis for the 2012 PM NAAQS.

2. All estimates are rounded to two significant digits. Estimates do not include unquantified health benefits noted in Table 5-2 or Section 5.6.5 or welfare co-benefits noted in Chapter 6. These estimates reflect incremental emissions reductions from an analytical baseline that gives "an adjustment" to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

3. The reduction in premature fatalities each year accounts for over 98% of total monetized benefits in this analysis. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure.

presentation. Central estimates of health impacts are presented along with estimated 5th and 95th percentiles, based on standard errors around the effect estimates from the epidemiological literature. Central estimates of monetized values of health impacts are also presented along with confidence intervals that reflect uncertainties in both the health impacts and unit values (e.g., WTP per case of disease or hospital admissions). Table Table **3** provides an example of how economic benefits are presented based on the recent RIA for the 2012 PM NAAQS rule (U.S. EPA, 2012).

In addition to the limited uncertainty analysis focused on propagating standard errors from the epidemiological studies, benefits analyses typically include a number of sensitivity analyses for important analytical choices, such as alternative effect estimates, functional forms (e.g. threshold models), discount rates, lag structures, and inclusion of additional endpoints. While these sensitivity analyses are not formal uncertainty assessments, they provide insights into how sensitive overall benefit estimates are to different assumptions. In general, because of the large impact of $PM_{2.5}$ on premature mortality and the large magnitude of the VSL, total benefits are most sensitive to assumptions regarding the $PM_{2.5}$ health impact and valuation functions.

Because of the resource and time requirements required to generate air quality modeling results (e.g., months for a single year's baseline and policy scenario), typical EPA benefits analyses choose one or two analytical "snapshot" years, usually reflecting a year when full implementation of the regulation is expected to have occurred (or as close to full implementation as possible given future year air quality modeling constraints). Benefits are calculated only for these snapshot years, and thus are not representative of an annualized stream of benefits or a net present value of benefits. They are simply the annual benefits resulting from emissions reductions occurring during that snapshot year. In a few cases, such as the analysis of the non-road diesel regulations issued in 2004, EPA estimated benefits in multiple years and interpolated benefits to generate a stream of benefits over the full implementation period for the regulation (U.S. EPA, 2004b).

3 The Benefits of Air Quality Improvements in CGE Modeling

While the partial analysis described in the previous section remains the most common approach for applied benefit cost analysis, some studies have incorporated limited beneficial impacts of environmental quality into CGE models for economy-wide estimates of the impacts. This section provides an overview of those studies, including how benefits were included in the CGE modeling for the analysis of the benefits and costs of the Clean Air Act from 1990 - 2020 (EPA, 2011) and studies by other researchers that have incorporated benefits into CGE models.

3.1 EPA CGE Benefits Modeling – Benefits and Costs of the Clean Air Act from 1990 to 2020

In March 2011, EPA released a study evaluating the benefits and costs from 1990 to 2020 of programs implemented under the 1990 Clean Air Act Amendments (CAAA), relative to a hypothetical baseline in which control programs under the 1970 Clean Air Act and 1977 Amendments remained constant at their 1990 levels of scope and stringency. In addition to estimating partial equilibrium measures of the prospective (2010 to 2020) direct compliance costs and household benefits, the study also included a CGE analysis that incorporated both compliance costs and some benefits over this time period.

The CGE analysis performed for the study was the first analysis by EPA to include some categories of air quality related benefits in a CGE model (specifically, the dynamic version of the Economic Model for Policy Analysis Computable General Equilibrium or EMPAX-CGE model).[3] The intent was to acknowledge that while there have been general equilibrium costs associated with the implementation of the CAAA, there have been concomitant improvements in health that may also have feedback effects on the economy.

[3] EMPAX-CGE is a multi-industry, multi-region computable general equilibrium model of the U.S. economy. The version of EMPAX-CGE used for the 2011 EPA study models 35 sectors and 4 representative households for each of the 5 regions, and is resolved with 5-year time steps. It is assumed that households have perfect foresight of future changes in policy and maximize utility over the full time horizon of the model by adjusting their consumption patterns and their decisions about labor force participation. EMPAX-CGE is a full-employment model, so households choose between labor (within their region's single labor market) and leisure based on income and substitution effects alone. Detailed information about the structure and calibration of the EMPAX-CGE model is available at http://www.epa.gov/ttnecas1/EMPAXCGE.htm.

The benefits included in EMPAX-CGE represented 1) expected changes in medical expenditures associated with pollution-related illness and 2) the expected change in workers' time endowment stemming from changes in mortality risks and changes in workdays lost due to illness.

As previously noted, EPA benefits analyses using the effect-by-effect approach described above are typically conducted for one or two "snapshot" years once full implementation is expected to have occurred. This presents a challenge for the inclusion of such benefits estimates in a dynamic CGE model, since more than one or two years are modeled and the timing of impacts is important. In the case of the 2011 EPA study, multiple years of air quality modeling with and without the CAAA were available (2000, 2010, and 2020), and it was therefore possible to approximate the stream of benefits needed for inclusion in EMPAX-CGE. The following sections briefly describe how these benefits were calculated and incorporated into the model.[4]

3.1.1 Changes in Medical Expenditures

Estimates of the change in incidence or prevalence of several morbidity endpoints in 2010 and 2020 were estimated using the environmental Benefits Mapping and Analysis Program (BenMAP).[5] These estimates of changes in incidence and prevalence were then multiplied by COI estimates from the published literature to calculate total changes in medical expenditures associated with PM and ozone-related health endpoints. Per event values of these annual medical expenditure estimates represented in Table 4. The total change in medical expenditures were then incorporated into EMPAX-CGE as a change in the representative household's expenditure pattern. To interpolate the additional year 2015 needed for inclusion in the EMPAX-CGE model, an estimated emissions trajectory was combined with estimates of the benefit per ton of avoided emissions to create a benefits trajectory.[6] An index was created using this benefits trajectory, and this index was used to deflate the 2020 benefits estimates to their corresponding values in 2015.

3.1.2 Effects on the Labor Force

The effect of $PM_{2.5}$ related premature mortality and ozone and $PM_{2.5}$ related morbidity on the labor force is implemented in the EMPAX-CGE model as a reduction in the representative household's time endowment. The expected effect of the CAAA on population due to changes in $PM_{2.5}$ related premature mortality was modeled using a dynamic population simulation model (PopSim) that incorporates life table

[4] A more complete discussion can be found in Chapter 8 of EPA (2011).

[5] The Environmental Benefits Mapping and Analysis Program (BenMAP) is a software package containing a library of concentration–response relationships, population data, baseline incidence rates, and valuation functions. It quantifies and values the $PM_{2.5}$ and ozone health impacts from air quality scenarios. Documentation for the most current version of the software is available at http://www2.epa.gov/sites/production/files/2015-04/documents/benmap-ce_user_manual_march_2015.pdf.

[6] Benefit per ton ($/ton) estimates are derived by dividing the total monetized human health benefits associated with an air quality scenario by the emission reductions associated with that scenario.

Table 4: Annual Medical Expenditures per Case, By Morbidity Endpoint [2006$]

	2010	2020
PM[2]		
Acute Myocardial Infarction[3]	$17,600	$17,300
Chronic Bronchitis[4]	$715	$810
Emergency Room Visits, Respiratory[5]	$369	
Hospital Admissions, Cardiovascular[6]	$27,400	
Hospital Admissions, Respiratory[6]	$21,000	
Ozone[7]		
Emergency Room Visits, Respiratory[5]	$369	
Hospital Admissions, Respiratory[6]	$16,400	$17,100

Notes:

1. Except for chronic bronchitis and acute myocardial infarction, medical expenditures per case are applied to the change in annual incidence. For chronic bronchitis and acute myocardial infarction, medical expenditures per case are applied to the annual changes in the prevalence of each disease, to generate an annual rather than lifetime estimate of costs for these chronic diseases.

2. Medical expenditure estimates for the following PM morbidity endpoints were not readily available: acute bronchitis, acute respiratory symptoms, asthma exacerbation, lower respiratory symptoms, upper respiratory symptoms, and work loss days.

3. Derived from Wittels et al. (1990) and Russell et al. (1998), both as cited in Abt Associates (2008).

4. Cropper and Krupnick (1999).

5. We assume that each E.R. visit equals one day of lost work time per worker affected. The estimate of 0.2 days per case reflects the percentage of cases realized by the working-age population, the ratio of workdays to total days in a year (235/365), and the percent of the working-age population in the labor force.

6. Agency for Healthcare Research and Quality (2000), as cited in Abt Associates (2008).

7. Medical expenditure estimates for the following ozone morbidity endpoints were not readily available: minor restricted activity days, school loss days, and outdoor worker productivity.

Disease Control. For a given policy scenario, PopSim simulates the U.S. population by age group and gender for each year through 2050. This approach is different from the population projection methodology employed by BenMAP, which uses exogenous growth factors to project 2010 Census data to the appropriate analysis year. Instead, PopSim accounts for the effect that death rates in one year have on the population at risk in future years, essentially following the population through time while accounting for air-pollution related changes in population each year. As the EMPAX-CGE model is dynamic and tracks changes in the economy over time, the population estimates produced by PopSim are more appropriate for this exercise than the population estimates based upon static population growth factors generated by BenMAP. Population was simulated for the with-CAAA scenario and without-CAAA scenario, so the difference in any given year represents the change in population in that year due to the CAAA.

Because EMPAX-CGE only models working age individuals, the impact of changes in morbidity and premature mortality incorporated into the model is limited to the cases expected to affect that demographic. To this end, the simulated change in population was combined with age- and gender-specific labor force participation rates from the Bureau of Labor Statistics. The percent change in labor force was then calculated by dividing total labor force changes by the baseline (with-CAAA) projection of labor force participation. That is, any additional population due to the CAAA will make on average the same labor force participation choices as the population in the baseline scenario, leading to an overall increase in the total time endowment for households in the economy. This percent change is assumed to apply to the full time endowment (labor and leisure) for the labor force. Because of the full employment nature of the model, households may choose to allocate a greater share of their time endowment to leisure (and less to labor) in some scenarios. Since households derive utility from leisure as well as consumption, this can lead to an increase in social welfare, even in the face of a decline in GDP and consumption.

Morbidity impacts were also incorporated into EMPAX-CGE as a percent change in the labor and leisure time available to workers. These impacts were calculated by health endpoint as work days lost using the values in Table 5, which were converted to lost work years, and then expressed as a percent change in the labor force by dividing estimated work years lost by the projected size of the labor force in each target year.

3.1.3 Potential Analytical Limitations

In the prospective study on the costs and benefits of the Clean Air Act from 1990 to 2020 (US EPA, 2011), EPA acknowledged several limitations associated with its incorporation of some benefits into the CGE framework. The study did not include the effects of premature mortality or morbidity for individuals outside the traditional workforce (e.g., children, elderly, informal economy). This notable exclusion biases the estimated welfare changes downwards, particularly since children and elderly populations are known to be more sensitive to health effects associated with changes in air pollution, though even without these effects the net welfare change was found to be positive.

The full-employment nature of EMPAX-CGE, combined with the assumption that additional population due to the CAAA make labor force participation decisions in the same way as existing households, may be an overly restrictive assumption in some circumstances, though the overall impact on results is unclear. It is also unclear how additional years of life due to improved air quality would actually be considered by workers, though the model imposes the restriction that they continue to make choices as they always have. Specifically, if the effect of the CAAA is mainly to reduce mortality rates at advanced (post-retirement) ages, then the effects on labor force participation may be modest.

In addition, non-market and some market benefits, such as visibility improvements, productivity enhancements in agriculture and forestry sectors, reduced materials damage, and reduced pain and suffering from pollution-related illness were not included as inputs to EMPAX-CGE. These types of market and non-market benefits, and how they may be included in CGE analyses, are discussed in Section 4 below.

Table 5: Work Days Lost Per Case, By Morbidity Endpoint

$PM_{2.5}$[2]		
Acute Myocardial Infarction[3]	Age <25: N/A	Age 45-54: 23.7 days
	Age 25-34: 17.7 days	Age 55-65: 137.0 days
	Age 35-44: 14.5 days	Age>65: 0 days
Chronic Bronchitis[3]	Age <25: N/A	Age 45-54: 55.5 days
	Age 25-34: 50.3 days	Age 55-65: 73.5 days
	Age 35-44: 42.2 days	Age >65: 0 days
Hospital Admissions, Cardiovascular[4]	Age 0-14: N/A	Age 45-64: 17.9 days
Hospital Admissions, Respiratory[4]	Age 0-14: N/A	Age 45-64: 30.1 days
Emergency Room Visits, Respiratory[5]	Average across all age groups: 0.2 days	
Work Loss Days	Average among working age population: 1 day	
Ozone[6]		
School Loss Days[7]	Average across all age groups: 0.7 days	
Worker Productivity	Not applicable[8]	
Hospital Admissions, Respiratory[9,10]	Age <2: 0 days	
Emergency Room Visits, Respiratory[5]	Average across all age groups: 0.2 days	

Notes:

N/A indicates that the underlying C-R function does not provide incidence estimates for that age group.

1. Except for chronic bronchitis and acute myocardial infarction, the number of work days lost is applied to the change in annual incidence. For chronic bronchitis and acute myocardial infarction, the work days lost presented in this table are applied to annual changes in the prevalence of each disease.

2. Separate work loss day estimates were not generated for the following PM health endpoints: acute bronchitis, acute respiratory symptoms, asthma exacerbation, lower respiratory symptoms, and upper respiratory symptoms. The lost work days associated with these endpoints are already reflected in the work loss day endpoint included in this table.

3. Derived from Cropper and Krupnick (1999).

4. Agency for Healthcare Research and Quality (2000), as cited in BenMAP user's guide, Abt Associates (2008).

5. We assume that each E.R. visit equals one day of lost work time per worker affected. The estimate of 0.2 days per case reflects the percentage of cases realized by the working-age population, the ratio of workdays to total days in a year (235/365), and the percent of the working-age population in the labor force.

6. We did not estimate the number of work days lost per case of acute respiratory symptoms associated with ozone exposure.

7. Derived from Abt Associates (2008). Note that 0.7 is the estimated average work loss days per school loss day, incorporating work-force participation rates for caregivers.

8. The benefits analysis presented in Chapter 5 does not estimate the number of cases for the worker productivity endpoint. Instead, worker productivity is estimated as the change in income associated with changes in ozone concentrations. We estimated the work days lost per dollar of income lost based on the average daily wages of outdoor workers.

9. Derived from Abt Associates (2008).

10. The dose-response function for ozone-related respiratory hospital admissions does not cover populations older than two years old and

Lastly, environmental quality (outside of the limited health impacts considered) was essentially assumed to be a separable component of the utility function for households. This may not hold, however, as cleaner air may encourage leisure activities, making air quality a complement to leisure.

3.2 Outside Studies Modeling Direct Health Impacts of Air Pollution in CGE Models

Several researchers have used relatively similar approaches as that of EPA (2011) to model the health benefits of improvements in environmental quality in CGE models, though with some interesting variations and for different applications. Vennemo (1997) was one of the first studies to incorporate environmental externalities into an applied CGE model, the Dynamic Resource/Environment Applied Model (DREAM) of the Norwegian economy. The model was based on a small open economy producing a single good available for export and domestic consumption and five input factors. The model estimated the impact of economic activities on emissions of criteria air pollutants using emissions coefficients and then mapped emissions to health, non-market, and production impacts.[7] Morbidity associated air pollution were assumed to impact labor supply at both the extensive and intensive margins. In the model, these two effects were combined and modeled as pollutant specific labor shocks. Changes in mortality risk associated with air pollution along with non-market impacts (discussed in more detail in Section 3.4) were modeled as a separable shock to household utility. Over the course of the model's time horizon the impact of the environmental feedbacks on consumption of goods and services increased from -1.4% in 2030 to -3.5% in 2090, with a welfare estimate of overall damages at 9.2% of full wealth (including leisure).

Mayeres and van Regemorter (2008) extended the GEM-E3 CGE model of the European economy to include non-separable health impacts of air pollution. The version of the GEM-E3 model used for this study was a dynamic model of 14 European countries with a 1995 benchmark. The model included an environmental component that modeled the emissions, transport, and increase in atmospheric concentrations of criteria air pollutants and carbon dioxide from the energy sector. In the standard version of the model all mortality, morbidity, and non-health impacts of air pollution were treated as separable in the utility function for the representative household. Mayeres and van Regemorter incorporated the effects of exposure related morbidity and the medical costs of pollution-induced mortality more fully into the model. Specifically, they considered the effect of air pollution on private and public spending for medical services and the time endowment within a bi-directional coupling of the economy and the environment.

These non-separable health impacts were included by extending both the household's representation and the economy's production functions. To model the effect of air pollution on the household's demand for medical services the top nest of the Stone-Geary utility function, which previously included only composite consumption, C, and leisure, l, was extended to include health, H. Specifically, the utility function was defined as

[7] Vennemo (1997) also considers the feedbacks from changes in traffic volumes, though the treatment is analogous to that of air pollution.

$$U = \alpha_1 \ln\left(C - \bar{C}\right) + \alpha_2 \ln\left(l - \bar{l}\right) + \alpha_3 \ln\left(H - \bar{H}\right) - \sum_{m=1}^{M} \alpha_{H,m} A_m \ , \tag{2}$$

where bars represent subsistence levels and the last component represents separable effects of changes in the ambient concentrations of pollutant from the benchmark, $A_{H,m}$, for the M pollutants, such that $\alpha_{H,m}$ represents the marginal impact of changes in ambient concentration.[8] Health was defined based on a production function approach, such that

$$H = H^* - \sum_{m=1}^{M} \beta_{1,m} A_m + \beta_2 MED \ , \tag{3}$$

where H^* is household health in the absence of air pollution or consumption of medical services, $\beta_{1,m}$ represents the marginal impact of the changes in the ambient concentration of pollutant m on health, and MED is the consumption of medical services. The price of medical services included government subsidies based on the social security systems in place within the European countries modeled, such that an increase in the quantity demanded of medical services affected both the household and government budgets.

The time endowment effect of morbidity was modeled in two parts to recognize the existence of paid sick leave requirements in the countries modeled. Part of the reduction in the time endowment was allocated to reducing the households' time endowment directly, while the remaining lost time endowment was modeled as a labor productivity shock in the model's production functions. The model was calibrated based on the assumption that 58% of the lost time should manifest as a labor productivity shock in the model.

The new parameters of the extended model associated with changes in air quality were calibrated to match the results of studies that have estimated components of marginal WTP for changes in ambient concentration separately (i.e., changes in household income, medical services expenditures, and the separable component). The authors assumed that 10% of the marginal WTP estimate for mortality risk reductions was associated with a change in expected medical expenditures and used that for calibration.

Mayeres and van Regemorter (2008) then compared the results from the extended GEM-E3 model to the case where all air pollution damages were separable. Ex ante marginal WTP for reductions in ambient concentrations were calibrated to be equivalent between the two models. The policy examined was a country specific CO_2 tax commensurate with commitments under the Kyoto Protocol. While climate change impacts were not modeled, the results included the benefits from reduction in energy sector criteria air pollutant emissions correlated with CO_2 emissions. They found the macroeconomic effects (e.g., GDP, labor) to be very similar between the two cases with the extended model estimating a larger increase in consumption. Given the difference in the utility functions between the two models, a direct comparison of estimates for the change in welfare was not possible.

[8] While the GEM-E3 model is dynamic, for convenience we focus on the household problem for only a single period.

Li (2002) applied a similar technique to study the welfare impacts of an economy-wide CO_2 emissions tax in Thailand, accounting for general equilibrium impacts of correlated PM_{10} emission changes. The static CGE model modified the standard International Food Policy Research Institute CGE framework outlined in Lofgren et al. (2001) and had 61 production sectors and three households (agricultural, non-agricultural, and government-employed). The model was extended to include a bi-directional coupling between environmental quality (PM_{10} only) and the economy, though the combined model was only run for one iteration so there may not be convergence in the model. Exposure-response functions were used to map PM_{10} concentrations, derived from emissions, to premature mortality and morbidity endpoints for non-agriculture and government-employed households of working age. In addition to affecting labor supply on the extensive margin it was assumed that exposure also affects labor supply on the intensive margin through a reduction in labor productivity. Premature mortality, morbidity, and labor productivity effects were all represented through a reduction in the households' time endowments. The model also included pollution-induced medical expenditures in the same fashion as Mayeres and van Regemorter (2008), where improvements in environmental quality reduced household and public expenditures on medical services.

Other studies examining the direct health impacts of air pollution using a CGE model extended MIT's Emission Prediction and Policy Analysis (EPPA) model to include health impacts of air pollution.[9] The extended model has been referred to as the EPPA Health Effects (EPPA-HE) model (Matus et al., 2008). To date the EPPA-HE framework has been used to study the impact of historical air pollution (ozone and PM_{10}) in the U.S. (Matus et al., 2008), Europe (Nam et al., 2010), and China (Matus et al., 2012), and future ozone pollution globally (Selin et al., 2009). While there are some slight differences in terms of the data employed and calibration of the model, the general methodology used to incorporate health impacts associated with air pollution and the interaction with the economy in the model was the same across these applications.

The EPPA-HE model incorporated the impacts associated with acute and chronic premature mortality, and pollution-induced morbidity. For premature mortality, the model accounted for the damages to households in terms of total factor consumption, including leisure, while also accounting for the cumulative economic effect of reduced labor, consumption, and investment associated with premature mortality. For pollution-induced morbidity, the model accounted for the reduced time available for labor and/or leisure, the increased demand for medical services, and cumulative economic effects from those changes.

To capture these health impacts the EPPA-HE model modified the original model structure in two ways. First, leisure was introduced into the model by including an additional top-level nest in the household utility function along with a constraint on the time endowment, which was split between labor and leisure. Second, a household healthcare production sector was added to account for the time and medical service expenditures required to respond to pollution-induced morbidity. The household healthcare sectors were

[9] EPPA version 4 is a global recursive dynamic CGE model with a five-year time step and 16 regions built on the Global Trade Analysis Project (GTAP) database.

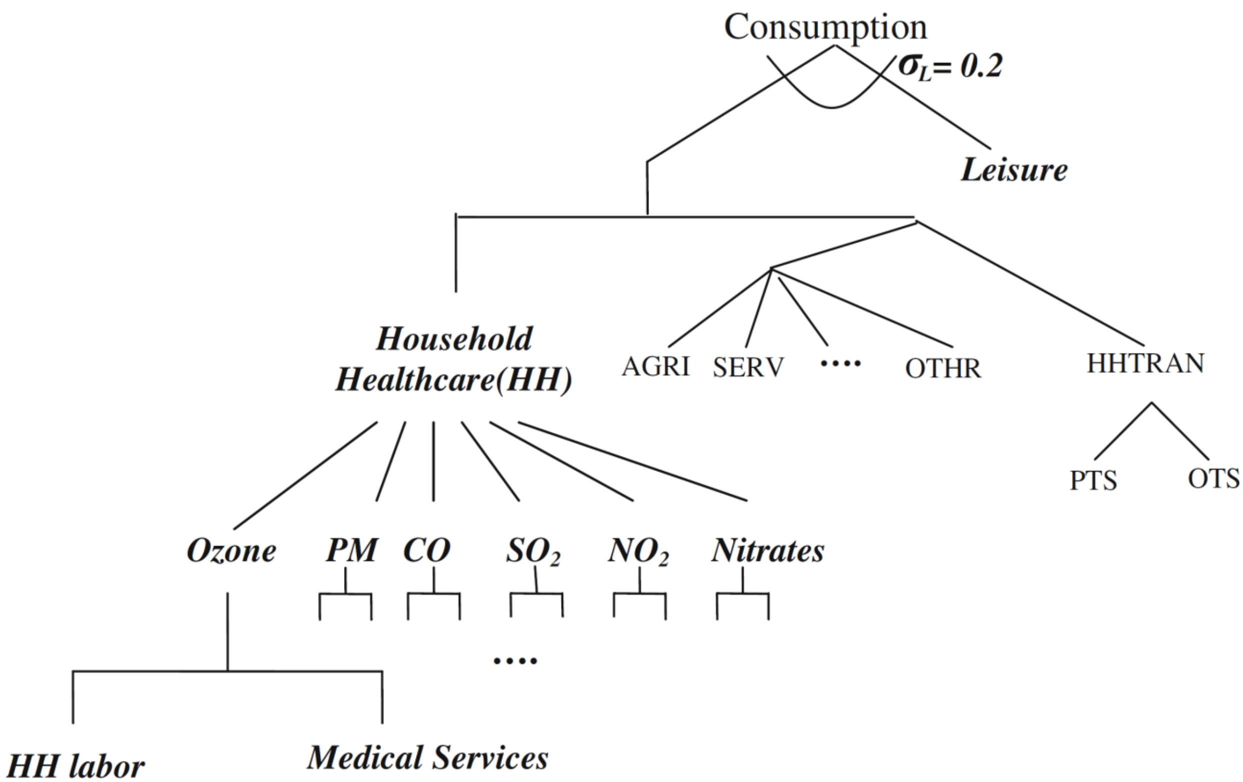


Figure 1: Household and Consumption Structure for EPPA-HE (Source: Matus et al., 2008)

health endpoint and pollutant specific, and Leontief in labor and medical services. The cost shares were calibrated based on external studies that estimate the composite costs of health endpoints. The updated structure of the household consumption function is presented in Figure 1.

To date the studies using the EPPA-HE model have used a unidirectional coupling, where atmospheric concentrations of pollutants are taken as exogenous and fixed, as opposed to the bi-directional coupling of Li (2002) and Mayeres and van Regemorter (2008). The model used a series of exposure-response functions from the epidemiological literature to map the atmospheric concentrations into health endpoints. In all cases, it was assumed that the exposure-response functions are linear without a threshold. For morbidity related endpoints, which were all assumed to be from acute exposure, this stage provided an estimate of the number of cases/events expected to occur in each model year. For premature mortality, the exposure-response functions provided an estimate of the proportional increase in mortality rates in the model year, which must be applied to baseline mortality rates to estimate the expected level of mortality. To account for the effects of premature mortality due to chronic exposure a cohort population model was used to track the number of individuals in a given age group in each model year.

For each cohort the annual average exposure was computed and entered into the appropriate exposure-response function to estimate the change in the mortality rate.[10]

The results of this mapping were then used to adjust the time endowment inputs to account for losses in labor supply due to premature mortality. Care was taken to ensure that for premature mortality of, what were assumed to be, working age individuals (younger than 65) it was effective units of labor that were removed from the time endowment. Effective labor includes time that was adjusted for labor productivity growth, which was included as an exogenous increase in the time endowment over model's time horizon. Based on an assessment of the literature, the model assumed that on average acute mortality occurs 0.5 years earlier than in the counterfactual due to exposure to air pollution. This is representative of the characteristic that premature mortality due to acute exposure primarily affects vulnerable populations. To account for the effects of chronic exposure-induced premature mortality, the model assumed an average lifespan of 75 years. Morbidity-related health endpoints were applied effectively as Hicks neutral shocks to the household healthcare sector to represent an increased need for time and medical services to maintain a given level of health.

In a given simulation year, the modeled welfare loss associated with pollution damages was based on the value of the lost time endowment in that period, the distortion imposed by the additional demand for medical services, and the cumulative economic impacts associated with reduced consumption and investment in previous years due to pollution. Lost time endowment in a given year was valued at the wage rate on the margin. To account for the loss of time endowment from cohorts that are not of working age (children and the elderly), the value of time was adjusted to be 1/3 and 2/3 of the wage rate, respectively.

Matus et al. (2008) used the EPPA-HE model to study the impact of air pollution and the benefits of air pollution regulations in the U.S. from 1970 to 2000.[11] Specifically, they focused on the damages associated with ozone, nitrogen dioxide, sulfur dioxide, carbon monoxide, and particulate matter ($PM_{2.5}$ and PM_{10}). To estimate exposures, it was assumed that all urban populations are exposed to the average urban pollution estimates published by the EPA. Conditional on historic exposure levels and their estimated damages, the EPPA-HE model was calibrated to reproduce observed economic activity. With this calibration, the authors estimated the benefits (reduced damages) associated with air pollution regulations assuming a counterfactual level of pollution exposure and the remaining air pollution damages based on above background pollution levels. Noting the uncertainty surrounding estimates in the exposure-response functions, Matus et al. conducted sensitivity analysis by setting the estimated

[10] To date chronic exposure induced premature mortality in the EPPA-HE model has been based on PM_{10}. The underlying epidemiological studies on the effects of chronic exposure to PM_{10} typically estimate the average exposure-response function for the population over 30 years of age. The EPPA-HE model accounts for cohort specific differences by weighting the exposure response function by the share of the cohorts' historic mortality due to cardiopulmonary and lung diseases relative to the total population over 30 and consistent with the epidemiological literature assumes to increase in mortality rates for those under 30.

[11] Matus et al. (2008) focus on incorporating the health impacts of air pollution mitigation policy in a CGE models and do not explicitly consider the costs of the mitigation in their framework.

exposure-response coefficients at their 95% confidence intervals. (The forthcoming whitepaper on uncertainty in general equilibrium modeling and benefits estimation will include a more detailed discussion of uncertainty analysis in a CGE context.)

Nam et al. (2010) used the EPPA-HE model to study the impact of air pollution in Europe from 1970 to 2005 in a similar fashion. In contrast to Matus et al. (2008), this study focused only on ozone and PM_{10}. However, in this case gridded population and concentration data was combined to calculate population weighted exposure estimates that are entered into the exposure-response functions. Nam et al. focused on the residual damages from air pollution relative to background levels conditional on air regulations already in place that affect the gridded concentration data. They found that the damages measured as a percentage change in overall welfare or consumption, while still notable, declined over time likely due to the strengthening of air pollution regulations.

Matus et al. (2012) use a similar approach as Matus et al. (2008) and Nam et al. (2010) to studied the historic damages in China from ozone and PM_{10} air pollution. The application faced additional challenges relative to the previous studies due to a paucity of concentration data from the pollutants being studied. The cost shares for the household healthcare sectors were recalibrated specifically for China using estimates of the Chinese composite morbidity costs. They considered a series of simulations using historical emissions, background emissions, and two hypothetical policy scenarios. Similar to the previous study the damages as a percentage of total welfare (or consumption) declined over time. However, in this case the results were in part driven by rapid economic expansion. This rapid economic growth also led to a case where losses in previous years account for 29% of the damages in 2005, as opposed to the 12% found by Nam et al. for Europe.

Selin et al. (2009) depart from previous work using the EPPA-HE model to focus on future changes in air pollution worldwide instead of within a single country or region. Specifically, they studied the damages associated with future changes in ozone exposure out to 2050, both with and without climate change. Ozone exposure was based on global atmospheric chemistry modeling with and without climate change and was population weighted by region based on gridded population data from 2000 (i.e., no future migration). The calibration of cost shares in the household healthcare sectors in developed countries was based on the estimates used by Nam et al. but adjusted using purchasing power parity (PPP). For developing countries, cost shares were calibrated based on estimates for China, again adjusted by the PPP. Age distributions were also separately specified for developing and developed regions. While previous studies based on EPPA-HE relied on sensitivity analysis to explore the robustness of their results, Selin et al. took a probabilistic approach and defined distribution over the exposure-response function estimates and cost shares in the household healthcare sectors, and then used a Monte Carlo simulation to integrate over the distributions. Selin et al. found that, when compared to a scenario in which ozone concentrations are assumed to be at background levels, 40% of the economic damages associated with ozone pollution in 2050 were based on the cumulative effect of mortality and morbidity in previous years.

3.2.1 Potential Analytical Limitations

While the studies discussed in this subsection have been important for highlighting the potential importance of the economy-wide effects of environmental quality improvements and providing frameworks for their incorporation into CGE models, their remain a number of potential limitations. To date the treatment of changes in premature mortality risk have been confined to changes in the time endowment of a representative agent. Furthermore, distinctions across space (e.g., distinction between emissions, concentrations, and population locations) and sub-populations (e.g., children and the elderly) that can be highly relevant for estimating health impacts have been only partially addressed to date in CGE models, if at all.

To date the approach taken by most, but not all, studies has been to assume that environmental quality is exogenous to economic activity. Therefore, in the cases where studies found the inclusion of health effects from improvements in environmental quality to have a significant impact on economic activity, there is no feedback from those changes on environmental quality. It may be the case that such feedbacks are of second order importance and the challenges associated with implementing endogenous is environmental quality in a CGE model is not warranted.

In addition, in most cases non-market and some market benefits, such as visibility improvements, productivity enhancements in agriculture and forestry sectors, reduced materials damage, and reduced pain and suffering from pollution-related illness were not included in the analyses. These types of market and non-market benefits may be significant for assessing the economy-wide effects of improvements in environmental quality as discussed in the subsequent section. Additional discussion of potential challenges and room for improvement is provided in Section 4.

3.3 Non-Health Environmental Feedbacks in CGE Models

A number of studies have incorporated non-health environment-economy linkages in CGE models. These linkages have been found to be important for assessing not only the benefits of policies via their impact on environmental quality, but also the costs of certain policies that may indirectly increase the opportunity cost, or reduce the availability, of non-market goods and services.

Vennemo (1997), described in the previous section, was one of the first studies to include non-health environmental feedbacks in an applied CGE analysis. Specifically, Vennemo considered the role of environmental policies on depreciation rate of capital, both through reduced sulfuric deposition on buildings and reduced miles traveled on roadways. The study also considered the impact of acidification on forests by modeling the change as a productivity shock to the forestry sector and use of forests by households for recreation.

Finnoff and Tschirhart (2008) studied the impact of fishery regulation on the economy by coupling ecological and economic general equilibrium models. Specifically, they linked a recursive dynamic CGE model of the Alaskan economy to a food web model of Alaska's marine ecosystem to study the welfare impacts in Alaska of regulating commercial fishing of a prominent species to protect the endangered stellar sea lion's food source. While the economic link is clear with respect to the directly regulated species, the policy also affects other economic sectors as the sea lion, along with its predator the killer whale, play a role in the state's tourism sector.

Their CGE model represented three productions sectors covering fishery, recreation and tourism, and a composite good. Fishing sector operations were constrained by a regulator that selects the total allowable catch and season length for the industry. The regulator's choices defined the equilibrium populations of the various species included in the food web along with the economic output of the fishing sector. The tourism sector depended on populations of some of these species, particularly "charismatic" species – sea lions, killer whales, and sea otters – that were required to create the "recreation experience" sought by consumers. The linear aggregate of charismatic mammal populations derived from the ecological model was incorporated into a Cobb-Douglas production function for the tourism sector along with capital and labor. In the production function for tourism, the exponent on charismatic populations defined the importance of these species for the sector, and was specified exogenously. Other parameters were calibrated to match benchmark data.

Finnoff and Tschirhart (2008) modeled the impact of various regulatory policies and found potentially notable general equilibrium effects occurring from both the change in the directly regulated industry and the tourism sector via its ecosystem connection. They also found that the ecosystem connections with the economy played an important role in economic impacts of potential interest to policy makers. While most of the primary factors (capital and labor) utilized in the directly regulated industry reside outside of Alaska, they still found impacts in the domestic labor market, not only due to the real income effect, but also through changes in the tourism sector due to ecosystem changes that result from the policy.

Antoine et al. (2008) directly introduced non-market recreation services (hunting, fishing, and wildlife viewing) into the EPPA CGE model. While the goal of their study was to establish better estimates for the cost of mitigating greenhouse gas emissions through biofuels policy by representing competing demands for land, the methodological contributions of this work are useful to discuss in the current context. The default category of unmanaged forest in the model was disaggregated into unmanaged but protected forests (e.g., national parks, wildlife reserves) and unmanaged and unprotected forests. Only the latter could be transformed to a managed land type (cropland, pastureland, managed forest, or unmanaged but protected forests).[12] The household sector was expanded to include an outdoor recreation sector, which included hunting and fishing services, wildlife viewing on reserved land, and wildlife viewing on other land. In each case production of the service was a CES combination of land and a Leontief composite of food, transportation, services, leisure, and other expenditures. For wildlife viewing on other land and hunting and fishing, the land input was unmanaged and unprotected forests, while for wildlife viewing on protected land the land input was an improved unmanaged but protected forests, which was a Leontief composite of unmanaged but protected forests and government expenditures.

Antoine et al. (2008) used the extended version of the EPPA model to consider the impact of greenhouse gas mitigation policies on biofuels use and household welfare, both with and without the inclusion of outdoor recreation in the model. They found that, both in the baseline and the policy case, the inclusion of demand for unmanaged forests for outdoor recreation lowered the amount of land converted to biofuel production. This result highlighted how the opportunity cost of biofuels as a mitigation measure was contingent upon the competing demands for land including non-market services, such as outdoor recreation. The authors found this tradeoff associated with non-market services to be a potentially important component for characterizing the social cost of mitigation policy, increasing the welfare costs

[12] Conversion to unmanaged but protected forests is considered permanent.

by around 20% in years where biofuels were considered a relatively cost-effective mitigation strategy by the model.

As noted by Antoine et al. (2008), there are a number of challenges with this approach to incorporating recreation in a CGE model. The CGE model requires benchmark estimates of the value for each of the inputs to the outdoor recreation activities. Even in the U.S., national or regional data for the time spent engaged in these activities is not complete, and for non-U.S. regions, data can be even sparser.[13] There are also data limitations in identifying the type of land used for the outdoor recreation activities. As an approximation, Antoine et al. assumed that all of the outdoor recreation activities take place on forest land, where in reality other land types (e.g., pastureland and grasslands) are also used for some of these activities. The model also assumed that the forest land used for these activities is provided as if supplied by a competitive producer responding to market demand. However, the conversion of land and ability of the owner to effectively price its use might be more challenging in practice. Finally, the literature does not provide readily available estimates for the substitution elasticities that relate the ability for households to substitute between land and other market goods when providing the recreation services and their ability to substitute between outdoor recreation and different activities in their utility function.

Carbone and Smith (2013) tested the quantitative importance of general equilibrium welfare effects from improving environmental quality compared to partial equilibrium measures. In this study, the authors extended the U.S. based CGE model of Goulder and Williams (2003) to include SO_2 and NOx emissions and their potential impacts on human health, non-market ecosystem services, and non-use existence services. The base model is a static CGE model with five consumption goods, four intermediate goods, and one basic factor of production (labor) calibrated to a 1995 benchmark year. Carbone and Smith added sector-specific emissions factors for SO_2 and NOx emissions that have effects on lakes and forests through acidic deposition, along with negative human health impacts. They also included two non-market services associated with recreational fisheries and tree cover and a non-use existence value associated with general habitat services. The preferences of the representative agent were defined with a nested CES utility function that combined the non-market services and output from the consumer services sector in a nest that is then combined with leisure to capture the recreational nature of these services. This aggregate leisure/recreation consumption good is then combined with an aggregate market consumption good. Finally, general habitat services are incorporated into the top-level nest. A diagram of this preference structure is provided in Figure 2.

The inclusion of these non-health environmental feedbacks presented a unique set of challenges in calibrating the parameters of the CGE model. Basic deposition factors were used to convert emissions estimates in tons to deposition rates in kg/ha on a national scale. The benchmark expenditure shares for these goods in the nested CES utility function were then calibrated such that the marginal WTP for those

[13] While the focus of the study was on greenhouse gas mitigation policies in the U.S., the authors noted that incorporating the same demand for outdoor recreation services in other regions was necessary to ensure that food and forestry imports were appropriately priced, such that the opportunity costs of preserving land within the U.S. were appropriately characterized.

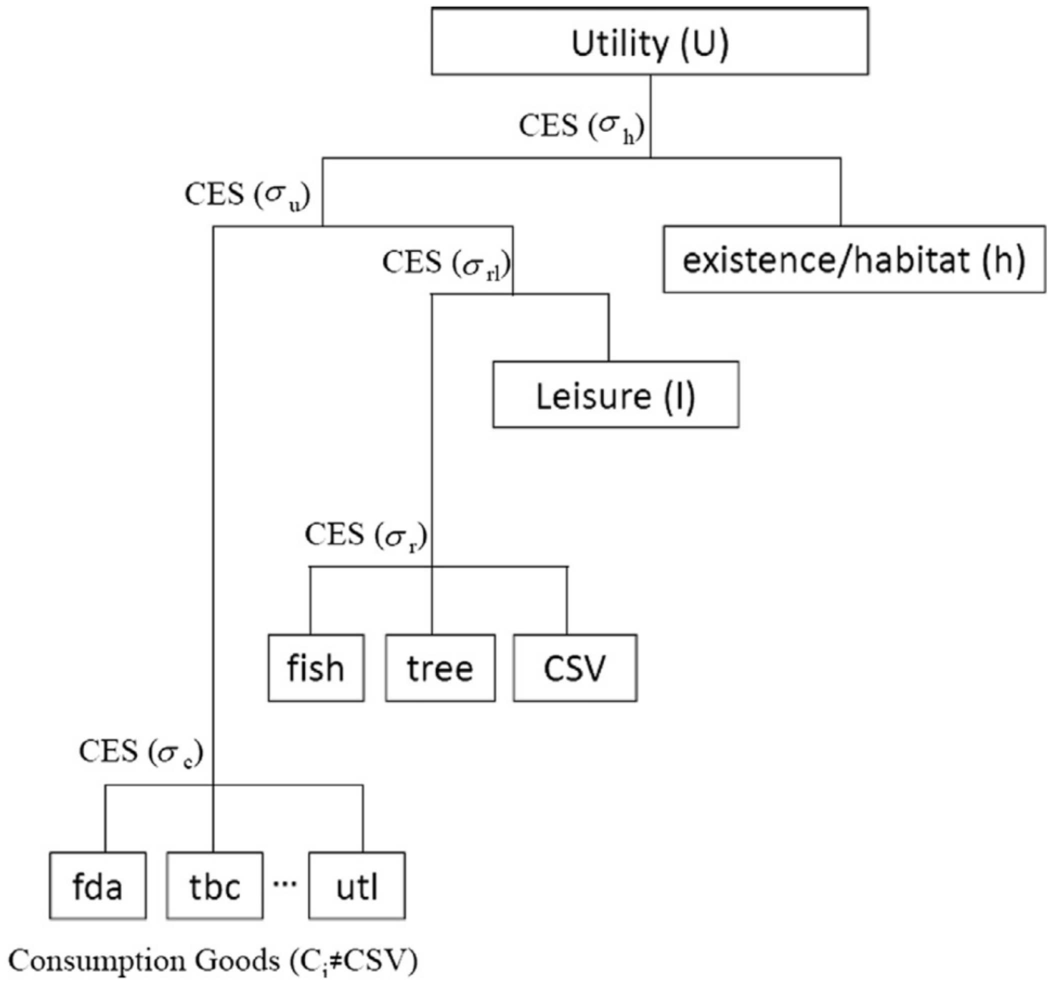


Figure 2: Carbone and Smith (2013) Preference Structure (Source: Carbone and Smith, 2013)

services in the model match estimates from the literature. The non-market and existence services were assumed to be quasi-fixed goods, in that they were endogenous to the model's equilibrium but were considered exogenous by the household. Therefore, analytical expressions relating the benchmark expenditure shares to the marginal WTP were not available, requiring a numerical procedure to calibrate benchmark expenditure shares, along with the other parameters that ensure labor supply elasticities implicit in the model match empirical estimates.

The results of the calibration procedure are conditional on assumptions regarding the substitution elasticities in the model, for which there may not be a significant underlying empirical literature to draw upon with respect to non-market services. Carbone and Smith (2013) assumed that general existence and habitat services are just as substitutable for the aggregate consumption of market and non-market goods as use-based non-market services (combined with consumer services and leisure) are for the bundled consumption good. Furthermore, they assumed that for the nests combining consumer services and non-market services and the combination of that aggregate bundle with leisure, that the cross-elasticities within the nests are one-fourth as substitutable compared to how substitutable the aggregate is with the market-based consumption bundle. In terms of Figure 2, these assumption imply that $\sigma_h = \sigma_u$ and $\sigma_{rl} = \sigma_r = \sigma_u / 4$.

Changes in level/quality of the non-market services have an effect on the economy through changes in agent's consumption patterns and supply of labor. Changes in general habitat services, even as a non-use category, have an effect on market services through the change in marginal utility. Carbone and Smith (2013) conducted a series of experiments with their model to compare the welfare change with and without the general equilibrium effects brought about through changes in environmental quality from polices to reduce SO_2 and NOx emissions. For a policy resulting in a 40% emissions reduction, roughly the magnitude of the CAAA impact, and focusing only on improvements to recreational fisheries, they found that including general equilibrium effects increases the estimate of the welfare improvement by 12% to 55%. The range is based on a sensitivity analysis around the substitutability between market and non-market consumption. The lower end of the range assumes greater substitutability relative to the reference case, while the higher end of the range assumes that they are stronger complements than in the reference case.[14] These results led the authors to conclude that even when the value of non-market services is a small fraction of the aggregate value of the economy, the general equilibrium effects stemming from how changes in environmental quality change household behavior in economic markets may be notable.

Carbone and Smith (2013) also included some health impacts of SO_2 and NOx emissions in a manner similar to those discussed in Sections 3.1 and 3.2. Specifically, they considered the lost time endowment associated with hospital admissions due to respiratory ailments induced by SO_2 and NOx emissions. They found that the welfare effects are small and have a negligible effect on the general equilibrium effects of changes in environmental quality, which is in line with the fact that hospital admissions represent a relatively small portion of the health impacts associated with SO_2 and NOx emissions.

3.4 Benefit-Side Tax Interaction Effects

The effects of environmental policy in a second-best setting that accounts for pre-existing tax distortions has been widely studied. Initial work in this area focused on tax interaction and revenue recycling effects on the cost side of the equation, finding general equilibrium effects to be of first order importance.[15] Subsequently, researchers have examined the role of interactions between pre-existing market distortions and the benefits of improved environmental quality. As noted by Bovenberg and Mooij (1994), if environmental quality enters the utility function in a non-separable fashion, changes to environmental quality will affect the household's labor-leisure choice, thereby working to exacerbate or mitigate, depending on the specifics of the situation, the effects of pre-existing factor market distortions.

[14] Carbone and Smith (2013) recalibrated the model depending on whether the welfare measure includes general equilibrium effects and for the different sets of substitution elasticities so that the marginal WTP implicit in the model remained in line with empirical estimates.

[15] See for example Bovenberg and de Mooij (1994), Bovenberg and van der Ploeg (1994), Goulder (1995), Goulder et al. (1999), Parry (1994), Parry (1997), and Parry et al. (1999).

Williams (2002) provided an intuitive approach to understanding the potential interaction of environmental improvements and distortionary factor market taxes in a static setting. The model considered an economy producing two goods that are consumed by a representative household, and a government that is financed through a labor tax. Production of one good negatively affects environmental quality, which can affect household utility directly (weakly separable with respect to consumption and leisure), the household's time endowment, their demand for medical services, and the efficiency of production. Williams showed that the welfare impact associated with internalizing the pollution externality using a revenue neutral tax on the production of the dirty good can be separated into four effects: the primary welfare effect, the revenue recycling effect, the cost-side tax interaction effect, and the benefit-side tax interaction effect.

The primary welfare effect captures the net benefits of an improvement in environmental quality in the absence of the distortionary labor tax, including any potential general equilibrium effects of the policy. In the case where the government raises revenue from labor taxes, the remaining three effects may be non-zero. The revenue recycling component accounts for the welfare gain associated with the government using revenues from the pollution tax to reduce the distortionary labor tax rate, thereby incentivizing labor.[16] The cost-side tax interaction effect captures the fact that policies may implicitly lower the real wage through higher prices on goods and services, which may further exacerbate the existing labor market distortion through a decrease in labor.

The fourth effect, the benefit-side tax interaction effect, captures the interaction between the impacts of improvements in environmental quality and pre-existing distortionary taxes. The magnitude and sign of the benefit-side tax interaction effect is conditional on how changes in environmental quality affect firms and households. For example, if an improvement in environmental quality increases the production efficiency of firms, this reduces their marginal cost of production, effectively acting as an increase in the household's real wage, which incentivizes labor. In this case, the benefits of the improvement are higher, as they work to mitigate the impacts of the pre-existing distortion. If instead the improvement in environmental quality increases the productivity of a fixed factor (e.g., increase agriculture/land productivity), this acts as an income effect for households, thereby exacerbating the pre-existing distortion and reducing the benefits from the improvement.

The benefit-side tax interaction effect is ambiguous if improvements in environmental quality increase the household's time endowment. This effect is particularly relevant for policies that improve public health through improvements in air quality, since as noted above, this effect may be viewed as increase the amount of time available to the household. This change improves the well-being of the household such that there is an income effect, which increases leisure (decreases labor). However, the household will also have more time to devote to all activities, so the net effect on labor supply is ambiguous.

[16] The role of this effect is unclear with respect to most EPA regulations, as they rarely generate revenue. However, in some cases EPA regulations implicitly relax the government's budget constraint (e.g., by preventing future publicly funded remediation).

In his conceptual model, Williams (2002) assumed that the demand for medical services was a function of environmental quality only and therefore only acts to reduce household income. Thus, an improvement in environmental quality means that a household no longer needs to spend a portion of their income on medical services to address the health effects of pollution. The resulting increase in leisure from the income effect exacerbates the pre-existing distortion and reduces the overall benefits of the environmental improvement.

It is unclear how this last result generalizes to applied settings where a portion of a country's medical expenditures are financed through labor taxes (e.g., Medicare, Medicaid). In their applied studies (which take place outside of the United States), Li (2002) and Mayeres and van Regemorter (2008) took this into account, but in neither case is the benefit-side tax interaction effect investigated. However, a number of theoretical studies have considered the potential for interactions between the health benefits of improved environmental quality and distortions in the healthcare market. Parry and Bento (2000) studied the tax interaction effect in the presence of tax deductions for some consumption goods, and established the intuition that the effect of environmental policies could mitigate such economic distortions if it led to a reduction in the sub-optimal overconsumption of such goods due to the subsidy. Caffet (2005) and Yamagami (2009) built of this work and explicitly incorporated subsidies for medical expenditures into the theoretical model of William (2002). They found that in the case of distorted markets for healthcare goods and services that it is possible for the health benefits of improved environmental quality to have additional impacts through reducing demand for healthcare goods and services and therefore mitigating the economic inefficiencies of subsidies in those markets.

In addition to the previously described interactions, the introduction of environmental quality into a dynamic setting may lead to additional interactions with respect to the way in which changes in mortality risk affect savings decisions, which may interact with other pre-existing capital taxes. The specific magnitude and ultimate impact of the interactions on the net benefits of air regulations in the United States is an empirical question that requires additional study. However, it is possible that benefit-side general equilibrium interactions could be of first order importance for understanding the net welfare impacts of environmental policies. Though studying those effects in an empirical setting may have numerous challenges including the definition of appropriate closure rules in a dynamic setting..

4 Additional Factors Affecting the Incorporation of Benefits into CGE Models

The research reviewed in the previous section provides an important step forward in estimating the economy-wide benefits of environmental policies. However, there remain a number of limitations and challenges associated with estimating a more comprehensive value for the economy-wide effects of improvements in air quality. This section considers a number of these areas.

4.1 Valuing Mortality Risk Reductions

As discussed in Sections 3.1 and 3.2, to date most CGE modeling studies that have examined the macroeconomic implications of human health have represented the proximate effects of changes in mortality risks through changes in the representative individuals' time endowments. The welfare effect of a change in time endowment on the margin will be driven primarily by the wage rate. As noted by Matus et al. (2008), though, the welfare impact will diverge from losses in expected income over time due to the compounding of other general equilibrium effects captured by the models. For analyses examining the impact of changes in premature mortality on economic variables such as gross domestic product (GDP), representing mortality risk changes by modifying the time endowment of the representative agent may be a reasonable starting point. However, in the context of applied welfare analysis, measuring the benefits of changes in premature mortality as a reduction in the time endowment is closely related to the "human capital approach" to valuing loss of life. In general, this approach involves estimating the benefits of mortality risk reductions using the change in an individual's expected remaining lifetime earnings (Linnerooth, 1979; Rosen, 1988; Johansson, 1995; Cropper, 2000). The human capital approach was in common use before Schelling (1968) and Mishan (1971) ushered in the application of willingness to pay measures for mortality risk reductions in benefit-cost analyses. The use of WTP measures is now standard practice, and as discussed in Section 2.2, is the main basis for EPA's benefit-cost analyses of the human health impacts of regulations.

In light of the similarities between the human capital approach and the time endowment approach that has been used in most CGE models to date, in this sub-section we review the distinctions between the human captial approach and contemporary willingness to pay approach to valuing mortality risk changes. We also consider the quantitative difference between EPA's (2011) benefit estimates (based on willingness to pay) and CGE results (based on changing the representative agent's time endowment) in light of these distinctions.

To begin, Schelling's (1968) early discussion of the conceptual differences between the human captial approach and the willingness to pay approach is still apt and worth revisiting:

> "People get hung up sometimes on the apparent anomaly that if a person would yield 2 percent of his lifetime income to eliminate a 1-percent risk of death, he'd have to give up twice his entire lifetime income to save his own life—which he cannot do if his creditors are on their toes. But he doesn't have to. I'd pay my dentist an hour's income to avoid a minute's intense pain—even to prevent somebody else's pain—without having to know what I'd do if confronted with a lifetime of intense pain. This is why the 'worth of saving a life' is but a mathematical construct when applied to an individual's decision on the reduction of small risks... Let me guess. If we ask people what it is worth to them to reduce by a certain number of percentage points over some period the likelihood that they will die, they will find it worth more than that percentage of the discounted value of their expected lifetime income. Arithmetically, if we tell a man that the likelihood of his accidental death over the next three years is 9 percent and we can reduce this to 6 percent by some measure we propose, and ask him what it is worth to reduce the probability of his death by 3 percent over this period (with no change in his mortality table after that period), my conjecture is it is worth to him a permanent reduction of perhaps 5 percent, possibly 10 percent, in his income..."

The quantitative difference between the value of mortality risk changes estimated under the human capital and willingness to pay approaches can be very large. For example, the present value of future earnings for a middle-aged worker paid the average wage in the U.S. is around \$0.75 million (2008 U.S. \$).[17] This figure can be compared to empirical estimates of the VSL based on hedonic wage and stated preference studies. Central estimates of the VSL from recent meta-analyses of both revealed and stated preference studies are between \$2 and \$11 million (2008 U.S. \$) (Cropper et al., 2011), and EPA's central estimate of the average VSL is \$7.9 million (2008 U.S. \$) (US EPA, 2010).[18] Therefore, contemporary central estimates of the willingness to pay for marginal mortality risk reductions are up to an order of magnitude larger than the average remaining lifetime earnings for a middle-aged U.S. worker.

This large quantitative difference raises two important issues for the representation of health benefits in CGE models. The first is that representing expected changes in mortality risks as a change in the representative agent's time endowment may significantly underestimate the welfare effect of air pollution regulations in a CGE analysis. As discussed in detail in Section 3.1, EPA (2011) used a CGE model to examine the macroeconomic impacts of the CAAA. Two scenarios were examined: a "costs only" scenario, in which the impacts of compliance expenditures alone were included, and a "labor force-adjusted" scenario. The latter scenario included both compliance expenditures and beneficial health impacts (specifically, PM induced mortality risk reductions and ozone and PM related morbidity for the working population), where these health impacts were primarily modeled as a change in the representative agent's time endowment. EPA's goal in that study was to examine the macroeconomic implications of the CAAA, not to conduct an applied welfare analysis. Nevertheless, the benefit study and the CGE analysis conducted by EPA (2011) provides an additional illustration of the quantitative difference between benefit estimates inferred from adjusting the expected time endowment in a CGE model and a willingness to pay measure for marginal mortality risk reductions based on the VSL. If one takes the difference between the estimates of Hicksian equivalent variation for the "costs only" and "labor force-adjusted" cases as an alternative measure of the benefits of the CAAA (Smith, 2012), this leads to an estimate of \$65 billion (2006 US \$) in 2010 (US EPA, 2011 Tables 8-7 and 8-8). In the same study, EPA reported a central estimate of the annual WTP for the same health endpoints included in the CGE analysis

[17] This figure was computed based on an individual who is 40 years of age and earns \$43,168 (2008 U.S. \$) per year, which was the annual mean wage in the U.S. in 2013 (http://www.bls.gov/oes/current/oes_nat.htm#00-0000). The individual's real wage is assumed to remain constant for the remainder of her working career (i.e., her nominal wage grows at the rate of inflation), she will retire at age 65, and she discounts future consumption at a rate of 3% per year. Finally, we assume that the individual is subject to a lifecycle mortality risk profile that matches average mortality risks among the U.S. population. The individual's age profile of mortality rates is estimated using a Gompertz function, where her probability of death at age τ, m_τ, increases exponentially with age, i.e., $m_{\tau+1} = (1+\eta)m_\tau$. A good fit to average mortality rates among the general U.S. population ages one and above (Xu et al. 2010) can be achieved using m_0 = 0.0001175 fatalities/yr and η = 0.0780. The individual's expected remaining lifetime earnings, or "human capital" (HC), are then calculated as $HC = \sum_{t=a}^{R} s_{a,t}(1+r)^{-(t-a)}y_t$, where $s_{a,t} = \prod_{\tau=a}^{t}(1-m_\tau)s_{a,t}$ and y_t is her real wage at age t.

[18] Worth noting is that under the same assumptions about the average middle aged U.S. worker used in Footnote X, the conjecture of Schelling (1968) implies a VSL between \$2.6 and \$5.2 million, which is comfortably within the range of estimates summarized by Cropper et al. (2011).

equal to $1.2 trillion (2006 US $) in 2010 (US EPA, 2011 Table 7-1). The difference between EPA's central benefit estimates and the (superficially) analogous welfare measure from the CGE analysis is consistent with the difference between people's marginal WTP for mortality risk reductions and the change in their expected remaining lifetime earnings (as summarized above), accounting also for the fact that the time endowment was adjusted only for working individuals (i.e., health improvements for children, retired individuals, or others not included in the labor force are absent from the CGE analysis).

The second issue raised by the large quantitative difference between the human capital and WTP approaches is what the difference means for modeling behavior. In models that include some degree of foresight and that allow for intertemporal wealth transfers, changes in the expected future time endowment or mortality risks will influence consumption and labor supply patterns. If the agent's WTP for a mortality risk reduction is in fact an order of magnitude larger than the expected value of the change in the agent's time endowment, we might also suspect that the agent's behavior would be notably different if health impacts are modeled explicitly as a change in mortality risk rather than implicitly as a change in the time endowment. If so, then this issue may have important implications not only for the use of CGE models to estimate the welfare effects of environmental regulations but also their ability to accurately represent the effects of environmental regulations on GDP and other macroeconomic outcomes.

4.2 Health Risks to Children

In addition to the mortality risk reductions discussed in the previous section, air quality improvements also impact the health of children, and those impacts are not necessarily the same as those experienced by adults for at least two reasons. First, children are in some cases more highly exposed to environmental contaminants relative to adults due to normal childhood activities and behavior patterns. According to *America's Children and the Environment* (US EPA, 2013a), "children generally eat more food, drink more water, and breathe more air relative to their size than adults do, and consequently may be exposed to relatively higher amounts of environmental contaminants." Second, children may be more vulnerable to environmental contaminant exposure compared to adults since they are still developing physically and may therefore, be more easily harmed.

Executive Order 13045*, Protection of Children from Environmental Health Risks and Safety Risks*, directs all Federal Agencies, including EPA, to make assessing children's environmental health and safety risks a high priority. When suitable exposure-response relationships can be identified, EPA has quantified child-specific benefits in regulatory analyses. As noted in Table 1 the child-specific health impacts that have been incorporated into standard benefit-cost analyses associated with changes in exposure to air pollutants include PM-related infant mortality and acute bronchitis, ozone related hospital admissions for respiratory causes and school absences, and SO_2-related asthma exacerbation and acute respiratory illness.

Fully capturing the benefits of reducing children's health risks can be challenging. As noted previously, standard practice in monetizing adult environmental health risk reductions relies on individual WTP estimates for ex ante risk reductions or cost of illness estimates as a proxy when WTP estimates are unavailable. However, there are a number of reasons why these measures may introduce bias when applied to children, including differences in risk preferences (e.g., risk aversion, involuntariness of exposure), duration of health outcome, and remaining life expectancy (US EPA, 2003). Furthermore, the standard practice of relying on individual, own-preferences for risk reducing behaviors may not be appropriate when considering children since it may not be possible to assume that they are well informed and rational economic agents. As such, the literature on assessing the benefits of reducing children's health risks has evolved around household decision-making models, with parents at the helm.

Gerking and Dickie (2013) reviewed the existing empirical literature (including both stated preference and safety product valuation studies) and categorized studies based on the conceptual model of household production considered: (1) single parent households in which only the child faces a health risk; (2) single parent households in which both the parent and the child face a health risk; and (3) two parent households in which both the parents and the child face health risks. They found that parents' marginal WTP for a one-unit reduction in health risk to their child generally exceeds their WTP for the same absolute risk reduction to themselves. A number of competing explanations exist for this finding beyond parental altruism, for example, the positive perception that protective actions might garner from others. It has also been suggested that this result could be due to differential parental discounting of latent risks over latency periods that differ in length depending on life-stage.

Gerking and Dickie (2013) noted two important limitations with existing studies that value changes in children's health. First, even though mortality and morbidity risks are intertwined (US EPA, 2010b), valuation of morbidity risks and mortality risks are generally modelled separately as if independent, thereby leading to a potential double-counting of benefits.[19] Second, most empirical models of valuing changes in children's health are based on a unitary framework that assumes a single set of preferences across parents in the household, whereas family dynamics may lead to more complicated decision-making scenarios.

Adamowicz et al. (2014) explored alternative models of household decision-making, which allowed them to address this second challenge by modeling households as a collective in which parents may have different preferences and potentially different levels of decision-making power. In this setting, parents may be assumed to work cooperatively or not. Using stated preference data collected from 432 matched pairs of parents' they empirically test these various models. They found Pareto efficiency in health resource allocations across the household and an inconsequential effect of redistributing the household's budget across members of the household on the marginal willingness to pay for health risk reductions for the child. This result lends support to both the unitary and cooperative models of household decision-making.

Gerking et al. (2014) developed an integrated model of mortality and morbidity. The model extended the standard one-period expected utility model with two states (alive and dead) to include a third state, the probability of being ill, with the utility experienced at this third state falling between the other two. The model adopted a family perspective in which the parent makes decisions about risk exposures for the child and herself and spanned multiple periods allowing for differences in latency across parents and

[19] This shortcoming is not exclusive to children's health valuation but more broadly to health valuation at all ages.

children for the health outcomes. Using stated preference data collected from two separate surveys, one on leukemia risks and the other on skin cancer risks, the authors confirmed two important findings from their conceptual analysis of the model. First, parents consider morbidity risk and conditional mortality risk as perfect substitutes. That is "in both studies, total health benefits of reduced health risks for children are entirely captured by the marginal willingness to pay to reduce unconditional mortality risk." Second, a parent's willingness to pay for risk reductions to herself is not the same as that for the same risk reduction in her child, but neither is it systematically lower or higher. Further, they find a potential for double counting if separate morbidity benefits are added to the total benefits calculation and that "adult VSL values may be a questionable guide to corresponding VSL values in children." These findings, taken together, point to the need for additional studies of WTP for children's health risk reductions.

While the approaches discussed above hold promise, WTP estimates for children's environmental health risks are often limited regardless of which model is used. While EPA has identified adequate WTP estimates for some health outcomes, and applies these routinely to child-specific endpoints in its benefits calculations for reduced PM exposures (see Table 3), these WTP estimates are rarely generated specifically for children, and so, do not capture the differences in WTP values described above and the nuances surrounding household decision making. As reported in section 2.2, cost of illness estimates are often used as a proxy for the missing WTP values in benefit-cost analyses. To the extent that age-specific cost of illness estimates can be derived, these are applied where appropriate. Otherwise, adult values – be they based on WTP or COI – are transferred to children's health endpoints in the monetization step in benefit cost analyses when child specific values are lacking.

As discussed in Section 3, previous studies on the benefits of air quality improvements have incorporated changes in children's morbidity and mortality risks to varying degrees using different methodologies. The EPA (2011) analysis, while providing a robust qualitative and in some cases quantitative partial equilibrium analysis of the benefits of reducing children's health risks through air quality improvements, only provided a limited inclusion of these impacts in the CGE analysis. Specifically, the only benefits associated with children's health impacts included in the CGE analysis were reduced school loss days representative of the lost time endowment for the working parent required to care for the child. No benefits representative of the willingness to pay by, or on behalf of, the children themselves for their health risk reductions are captured in the model.

As noted in Section 3.2, the studies using the EPPA-HE model (Matus et al., 2008; Nam et al., 2010; and Matus et al., 2012), accounted for the value of lost non-work (leisure) time associated with morbidity rates in children due to air pollution exposure. Based on the U.S. wage profile by age, lost leisure time for cohorts below the working age due to morbidity impacts are valued at one-third the wage rate. Further, the model incorporated values for market and non-market effects of illness (e.g., pain and suffering and other associated losses) by apportioning WTP estimates among three components: medical expenditures (demand for medical services), lost work time (reduction in labor force), and "damages beyond these market effects" (which as a proxy are considered a loss of leisure). Whereas the valuation estimates used by Matus et al. (2012) do not appear to include any that were derived specifically for children (see Holland et al., 1998), those used by Nam et al. (2010) and Matus et al. (2012) were expanded to include a few child-specific WTP estimates, including one for respiratory symptoms in asthmatic children (see Bickel and Friedrich, 2005).

With this approach, the effect of children's environmental health risks will be below that for adults for the same outcomes. This is in contrast to the recommendations by OMB in Circular A-4 (OMB, 2003) on the valuation of children's health risk changes in benefit cost analyses, where OMB recommends that agencies use values for children in benefit-cost analyses that are "at least as large" as those for adults for the same outcome, barring any compelling evidence to the contrary.

The potential importance of accounting for children's health risks in CGE models follows from the fact that in a dynamic setting, changes in the health risks faced by children can have longer-term implications for the economy than comparable health risks faced by adults. Nevertheless, as noted above there are many challenges associated with developing partial equilibrium WTP measures for improvements in the health risks facing children. Given the challenges of incorporating the changes in adult mortality risks in CGE modeling, as described in the preceding section, and the challenges in developing partial equilibrium estimates for childrens' health benefits, there are many open questions about the appropriate methods for children's health risk changes in CGE models.

4.3 State Dependent Utility Functions

A common choice within CGE models is to represent households as maximizing homothetic utility functions, such as the constant elasticity of substitution or Cobb-Douglas functions. These functions are convenient because they are easily aggregated and can be represented with a relatively parsimonious set of parameters that can be calibrated using existing market data. However, these functions impose restrictions on consumer behaviors that may be unrealistic; specifically, the elasticity of substitution between final goods is fixed and budget shares for final goods are determined by relative prices and not affected by income. Furthermore, in these typical specifications, health does not play a direct role in affecting households' well-being or preferences, as represented by the marginal utility of consumption or leisure and the elasticity of substitution between labor and leisure or between other final goods.

As discussed in Section 3, past studies that have incorporated the health benefits of air pollution regulations in CGE models typically assume that health affects the utility of representative households only through its impact on medical expenditures and the time endowment. While this approach is attractive due to its tractability, it does not take into account the possibility that health status may either, directly affect utility (independent of effects on consumption), or may be a complement to consumption and leisure.

Various hypotheses about the role of health status in the utility function have been examined analytically and to a limited extent tested with empirical data in the health economics literature (Lillard and Weiss, 1997; Finkelstein et al., 2009; Viscusi and Evans, 1990; Sloan et al., 1998). This relationship is commonly discussed in terms of "health state dependence," meaning "the effect of health on the marginal utility of a constant amount of nonmedical consumption" (Finkelstein et al, 2009). While being in an undesirable lower health state must lower the level of utility for an individual at any given level of consumption, the sign and magnitude of the effect on marginal utility (the derivative of utility with respect to consumption) is not known a priori. If consumption goods are predominately complements to health status, then negative state dependence might be likely, whereby a deterioration in health is accompanied by a reduction in the marginal utility of consumption. This could be the case where a deterioration in the health state inhibits the enjoyment of consumption such as the flu reducing the enjoyment of a vacation. However, in the case where final goods and services can substitute for good health there may be positive dependence where the marginal utility of consumption increases with deteriorating health.[20] For example, reduced health states that inhibit non-market production such as cooking increase the marginal utility of prepared meals, or a broken hip may require increased utilization of taxis. The difference between these two cases is depicted in Figure 3.

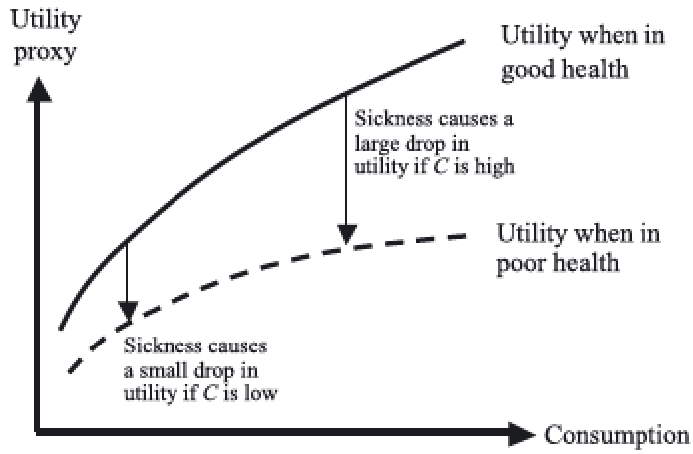
To examine health state dependence, Finkelstein et al. (2013) used panel data on health status and subjective proxies of utility for the elderly and near-elderly to model the way in which utility proxies change across health shocks and consumption levels. They found negative state dependence where the marginal utility of consumption is 10-25% percent lower for individuals in a health state characterized by a chronic disease (e.g., hypertensions, diabetes, cancer) relative to a healthy state, holding the level of consumption constant.

Sloan et al. (1998) used results from a stated preference survey to estimate the scaling coefficients for log utility functions that differ between a healthy state and one with multiple sclerosis. They estimated that the marginal utility of consumption in the health state with multiple sclerosis was 92% lower than in the healthy state for the general population. However, when the sample is restricted to those with multiple sclerosis the marginal utility in the unhealthy state was only 33% lower.

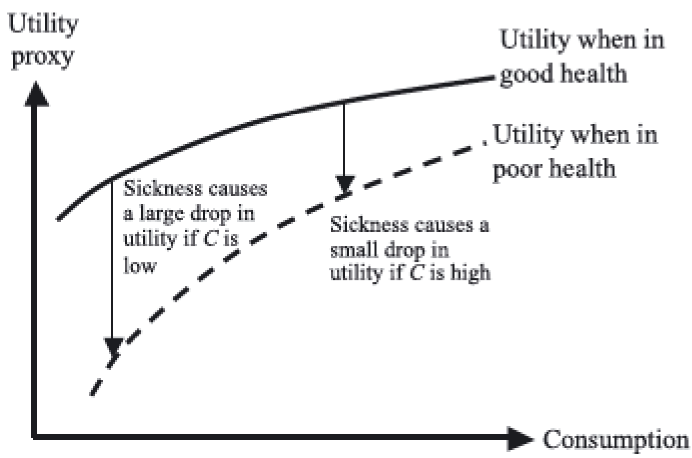
Viscusi and Evans (1990) also used stated preference survey data on compensating differentials for job related injuries in the chemicals sector to test the health state dependence of the utility function. They found evidence of negative state dependence with marginal utility in the injured state being 7% to 23% lower than in the pre-injury state. These results were driven by the potential for more severe injuries lasting more than a month. When the sample was restricted to more minor and temporary injuries they found some evidence of positive state dependence.

Murphy and Topel (2006) calculated the value of improvements in health and life expectancy in the U.S. from 1970 to 2000. They argued that improvements in life expectancy are important for such valuation because of the additional welfare gained from the additional expected years of life, and the improvements in quality of life from health improvements which “raise utility from given amounts of goods and leisure.” This prior was implemented in their framework through the imposition of negative state dependence by multiplying the utility by the current health state measured from zero to one. At its peak effect, Murphy and Topel (2006) suggested that general health improvements over the 30 year time period increased the marginal utility for 55 year old males by nearly 10%.

[20] Specifically, the sign of state dependence requires the cross partial of utility over health and non-medical consumption to be non-zero, such that for a negative cross partial (positive state dependence) the two are Edgeworth-Pareto complements (Samuelson, 1974).



a: Marginal Utility Declines with Sickness



b: Marginal Utility Increases with Sickness

Figure 3: Health State-dependent Utility Functions [Source: Finkelstein et al., (2013)]

Similar to Viscusi and Evans (1990), Evans and Viscusi (1991) used stated preference survey data on compensating differentials for consumer product risk to examine the health state dependence associated with minor and temporary health effects. They assumed a logarithmic utility function and developed an estimation procedure to determine whether the responses exhibit negative state dependence or equivalence between the negative health effect and a loss in consumption. The restricted model does not allow for the possibility of positive state dependence. Evans and Viscusi (1991) rejected negative state dependence for this class of minor health effects.

Lillard and Weiss (1997) examined health state dependence by studying the effect of health state shocks on dynamic consumption patterns of retired households between 1969 and 1979. The authors estimated the parameters of a stochastic dynamic programming framework for household welfare maximization using consumption and health data from the Retirement History Survey. Both the parameter estimates and subsequent simulations using their model showed strong positive state dependence for married couples where one member enters a lower health state, but failed to find evidence of health state dependence for single female households.

Edwards (2008) studied the potential rationale for a continued decline of household exposure to financial risk after retirement. In particular, Edwards (2008) assessed whether this investment pattern is better explained by nearing medical expenditures or due to positive state dependence that would increase marginal utility in the increasingly more likely event of being in a lower health state. Using data from 1992 to 2000 from the Health and Retirement Survey and the Study of Assets and Health Dynamics Among the Oldest Old, the author estimated the parameters of a stylized model of portfolio choice that disentangles the effect of health state dependence and risk aversion, including the effect of potential out of pocket medical expenses. Edwards (2008) found evidence of positive state dependence, though noted some caution is warranted when interpreting the magnitude of the results due to the stylized nature of the model.

In a review of studies examining health state dependence Finkelstein et al. (2009) noted that "a priori, the sign (let alone the magnitude) of any health state dependence is ambiguous" and that "we have relatively little empirical evidence on health state dependence, and what we do have is inconclusive in both sign and magnitude." However, stylized numerical exercises have found that even moderate amounts of health state dependence can have large effects on the optimal level of lifecycle savings (Finkelstein et al., 2009). Moderate amounts of health state dependence may also have a notable impact on the welfare change associated with changes in air quality. Therefore, while there may remain serious concerns about the ability to calibrate any potential approach of incorporating health state dependence into studies on the benefits of air regulations using CGE models, the effect of omitting such impacts has the potential to be significant.

The previous literature on health state dependence offers a few potential approaches, which may be useful for incorporate health state dependence into a CGE framework. Health state dependence could be represented by having multiple representative agents that differ by health status, similar to the way in which some CGE models represent different household demographics. As in Evans and Viscusi (1991) and Murphy and Topel (2006), it could be assumed that total utility is simply scaled between the types of representative households depending on their health state, with lower health states being affected by a multiplier less than one. Such an approach may provide a tractable solution as the literature provides empirical estimates of this scaling factor in some circumstances. However, this approach would necessarily impose negative state dependence, such that the marginal utility associated with consumption of all final goods decreases with health status. An alternative approach would be to adjust the productivity of particular final goods and leisure between the representative households depicting various health states. This approach would provide the ability to allow final goods and leisure to vary with respect to whether they are complements or substitutes for health status. This approach may provide a more realistic depiction on the effect of health status and would leverage the abilities of CGE models. However, calibration of such an approach may be beyond the scope of the current empirical literature on health state dependence.

While such approaches may be tractable within typical CGE frameworks, they would likely assume that transitions between health states are complete surprises to households. In reality, consumers are aware ex ante of the potential risk and potential to transition between health states, and will behave accordingly. An environmental regulation that lowers the risk of entering a lower health state may cause households to adjust their behavior. As shown by Finkelstein et al. (2009) and Edwards (2008) such changes may include adjustments to savings behavior that could have important impacts within a CGE setting if the effect is non-marginal.

4.4 Indirect Impacts on Human Health

The primary pathway for air regulations to impact human health is through improvements in air quality as discussed in Section 2. However, there may also be indirect impacts on human health through other pathways that may also be of interest. One channel by which human health may be indirectly affected by air regulations is through a decrease in the real wage, due to an increase in prices of goods and services relative to the wage rate. An income effect may cause households to reduce their consumption of healthcare goods and services, which in turn may reduce the health state for the household. There may also be a substitution effect whose directional impact on households' purchases of healthcare goods and services, and in turn their health state, will depend upon on the relative effect of the regulation on healthcare prices compared to other final goods and services. Furthermore, improved health through the direct impacts of the regulation may reduce the demand for healthcare goods and services, thereby lowering the price and potentially cause households to substitute towards such goods and services improving their provision of health.

Ex ante, the ultimate impact of these competing indirect effects on household health is ambiguous. However, under the assumption of perfect and complete markets, the value of this change in health to society will be captured in a partial equilibrium welfare measure, as prices will capture the opportunity cost to society of compliance, inclusive of any household indirect health effects. In the more realistic case where distortions exist in the economy, CGE models are particularly adept at capturing changes in relative prices and therefore, it may be worth considering whether these models can offer improvements in capturing the indirect effect of air regulations on households' health. In their review article of tools for comparative analysis, Hofstetter et al. (2002) suggest that tracking such indirect effect may very well require data-intensive general equilibrium models. Implicitly, this class of models already captures the demand by households' for healthcare, which is conditional on the welfare benefits this type of consumption provides through its effect on health. As such, social cost estimates generated by CGE models already implicitly take into account the potential indirect impacts the costs of complying with an air regulation would indirectly have on households' health.

However, improvements may be possible to more robustly capture households' demand for healthcare goods and services. This may require directly modeling the way in which consumption of those goods and services impact health and ultimately household welfare. As noted in Section 3.2, there are a limited number of studies that have considered the endogenous provision of health through the consumption of healthcare. However, additional research may be warranted given the importance of household heterogeneity (across multiple dimensions) and pre-existing distortions in the healthcare market, both of which may be important for understanding the ultimate effect on the endogenous provision of health by households.

Air regulations may also have indirect impacts on human health through their interaction with labor markets. For example, the reallocation of labor across the economy in response to an environmental policy may result in transition costs borne by individual households in moving to the new policy equilibrium. These transition costs could potentially affect their budget constraint and in turn, their endogenous provision of health, though the modeling of such impacts would appear to be subject to the challenges noted above and those associated with capturing transition costs in a CGE model, as discussed in the whitepaper on Economy-Wide Modeling: Social Cost and Welfare. This reallocation in the labor market may also shift the distribution of on-the-job risks across the workforce, which could be relevant for understand the net impact of labor market changes on household health.

4.5 Air Quality and Labor Productivity

Outdoor workers may be at increased risk of experiencing health impacts associated with air pollution due to their increased exposure. As previously discussed, such exposure may lead to effects on the extensive margin as the quantity of labor supplied is reduced due to workdays lost to illness or premature mortality. In addition to these effects, in some cases the elevated exposure for outdoor workers may also lead to less visible effects on the intensive margin as poor air quality could impair worker productivity and therefore reduce the amount of “effective labor” supplied.

There is evidence that in some cases outdoor workers may experience temporary health impacts associated with short-term increases in exposure to air pollution. For example, short-term exposure to elevated ozone concentrations has been shown to notably affect lung function in healthy adults (US EPA, 2013b). Specifically, decreased lung function due to ozone exposure has been observed in outdoor agricultural (Brauer et al., 1996) and forestry (Hoppe et al., 1995) workers. Just as such, health impacts have been found to reduce the speed of hikers (Korrick et al., 1998), they may have an effect on the productivity of outdoor workers.

In an early examination of the subject, Crocker and Horst (1981) used a small case study in southern California to examine the impact of ground-level ozone concentrations on agricultural workers. Specifically, over the course of a single season they examined the productivity of 17 citrus pickers who were paid on a piece-rate basis, i.e., a fixed rate was paid per box of fruit picked. Air quality was found to be a significant determinant of productivity for a subset of the workers studied. In a more recent study, Zivin and Neidell (2012) more thoroughly examined the impact of ground-level ozone concentrations on the productivity of agricultural workers in California’s Central Valley. Using data on the output of workers paid through piece-rate contracts they observe that a 10 parts per billion decrease in ambient ozone concentration for the area was associated with a 5.5% increase in worker productivity.[21] While this result is based on agricultural activity in a particular location, they note that 11.8 percent of the U.S. labor force is associated with industries with regular exposure to outdoor conditions.

[21] The average concentration in the sample was 50 ppb and the current federal NAAQS is 75 ppb.

Given the appropriate sectoral and possibly regional disaggregation, the mechanics of introducing labor productivity shocks conditional on air quality into a CGE model could be relatively straightforward. The larger challenge may be the availability of estimates that would allow for the mapping of air quality levels to sector-specific labor productivity. Such estimates are essential for appropriately calibrating the labor productivity shock that would be introduced into the model conditional on air quality changes. While there has been substantial work examining the influence of air quality on physical health outcomes (e.g., lung function), very little is known about the influence of such health outcomes on labor productivity.

4.6 Material Damages from Atmospheric and Deposition Effects

As briefly noted in Section 2.1, atmospheric and deposition effects of air pollution can lead to materials damages. These damages can be broken into three categories or primary interest: corrosive effects on commercial and industrial buildings, household soiling, and corrosive and soiling effects on structures of cultural and historical significance. Criteria air pollutants can have corrosive effects on some materials leading to increased rates of depreciation. Materials such as paint, galvanized steel, mortar, and stone used to construct walls, roofs, fencing, and window trim (etc.) may all be susceptible to damages from acid deposition effects, resulting in reduced time intervals between repairs or replacement. Estimating the impacts of acid deposition changes because of air regulations is a complicated problem that requires estimating the materials response to acid deposition, modeling the deposition effects at an appropriately fine spatial scale, and estimating the spatial distribution of sensitive materials. Similarly, increases in atmospheric CO_2 concentrations lead to increased depreciation rates for some materials such as concrete (Saha and Eckelman, 2014). If all of this information is available, which is not necessarily the case for most regulations, it may be possible to aggregate the estimates based on the proportion of the capital stock affected to regional impacts on the capital depreciation rate in a CGE analysis.

Household soiling refers to the accumulation of dirt, dust, and ash on exposed residential structures. While some recent studies have sought to estimate this benefits category (Yoo et al., 2008), the majority of the estimates in the literature have been deemed too narrow in their spatial scope or too outdated for use in current analyses of national air quality improvements (US EPA, 1998).

Improvements in air quality will also reduce soiling of, and damages to, buildings and structures of cultural and historical significance. Society's willingness to pay for preventing such damages through improvements air quality may be significant (Grosclaude and Soguel, 1994), in part due to the non-substitutability intrinsic to such assets (Kling et al., 2004). However, the studies estimating the benefits of preserving historical and cultural structures are naturally very site specific (e.g, Willis, 1994; Chambers et al., 1998) limiting the potential for benefits transfer approaches. Furthermore, it is unclear how such values would be incorporated into a CGE framework, though the benefits may have a strong impact on the relative demand for regional tourism services.

4.7 Air Quality and Productivity in the Agricultural and Forestry Sectors

Air pollution has the potential to affect vegetation in multiple ways, including induced foliar injury, reduced annual growth rates, and induced root loss (US EPA, 2007). Of particular concern is ground-level ozone that results from emissions of nitrogen oxides and volatile organic compounds (VOCs). These impacts on vegetation can have meaningful direct effects on the agriculture and forestry sectors of the economy, in addition to non-market welfare effects that may also affect economic activity as discussed in Section 3.3.

Visible foliar injury can reduce the desirability of leafy crops, (e.g., spinach, lettuce) and ornamentals (e.g., petunias, geraniums, poinsettias). The discoloration effects and early shedding of leaves, as a result of air pollution, can also have notable welfare and economic effects in areas where foliage is highly valued (e.g., scenic vistas, fall foliage). Elevated levels of ozone can reduce the overall health of plants through inhibiting their physiological processes (US EPA, 2007). These impacts can directly affect the productivity of agriculture and forestry sectors. Conceptually, the inclusion of the impacts of air pollution on the productivity of land used in the agriculture and forestry sectors may be straightforward, but in practice, there are a number of challenges. Air pollution, such as ground-level ozone, is not uniform across space, and its effects are not uniform across plant species. Therefore, there are potential adaptation opportunities for the economy through substitution. In highly aggregated CGE models, it may be difficult to immediately capture these substitution possibilities, both spatially and sectorally. Previous analyses conducted by EPA, such as the analysis of the benefits and costs of the Clean Air Act from 1990 - 2020 (US EPA, 2011), have considered the impacts of emissions reductions on agricultural productivity, though in detailed partial equilibrium settings that more completely represent some of the substitution possibilities. Incorporating results of such modeling exercises, or the details that allow the estimation of net productivity shocks, is similar to the linkages topic considered in the whitepaper on Economy-Wide Modeling: Social Cost and Welfare, and answers to charge questions on that topic may be applicable here. For the impact of improved air quality on plant health and its implications for non-market services, the challenges are similar to those previously described in Section 3.3.Spatial Challenges in Benefits Assessment

The heterogeneous impact of air quality improvements across space presents unique challenges for assessing the benefits of environmental policy in general, and potentially the use and/or interpretation of CGE-based benefits assessments in particular. In this section we focus on two areas, the degree of spatial heterogeneity in the benefits of air quality improvements relative to the spatial resolution of typical CGE models; second, and the potential first order welfare effects associated with re-sorting in housing markets in response to exogenous changes in the provision of public goods (i.e., air quality).

4.8 Spatial Resolution of Benefits

The impacts of regulations designed to improve air quality have effects that can differ significantly across space due to a number of crucial factors. First, affected sources are not distributed uniformly across space, and in many cases, the change in emissions is not uniform across sources. Second, the relevant changes in air quality may not be associated with the pollutants that are directly emitted but are instead formed through secondary processes. For example, tropospheric ozone forms because of a complex series of non-linear chemical reactions of precursor pollutants, such as volatile organic compounds and nitrogen oxides. Therefore, the impacts associated with changes in emissions depend on the dispersion and transport of the pollutants prior to such reactions, and are influenced by spatial heterogeneity in weather patterns. Finally, the ultimate impact on human populations depend on the correlation between the spatial

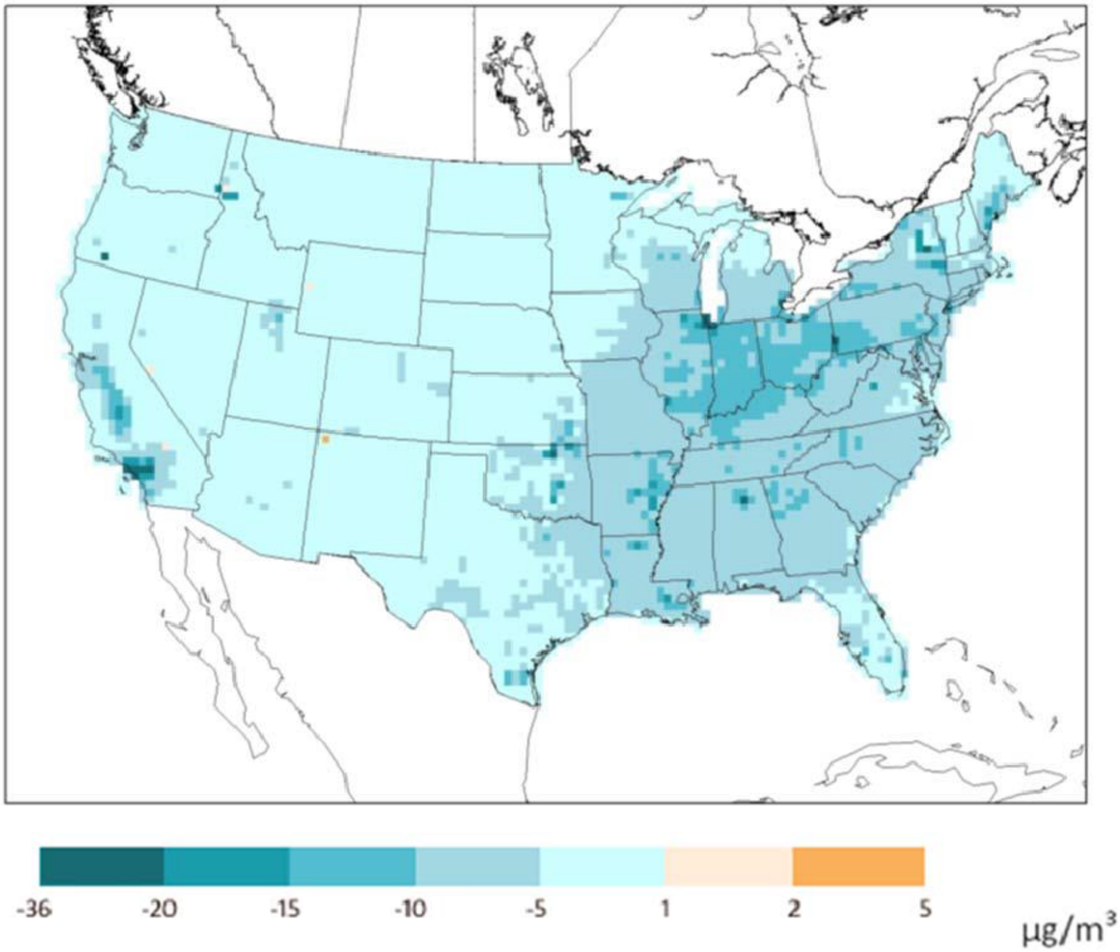


Figure 4: Predicted change in 2020 $PM_{2.5}$ concentrations in EPA (2011)

distribution of air quality impacts and the spatial distribution of populations, demographics, and land uses. As a result, the impacts of air regulations often need to be estimated a relatively fine spatial scale. Often, estimates of the benefits are calculated at the spatial resolution of a 12 to 36 km grid. For example, Figure 4 presents the predicted changes in $PM_{2.5}$ concentrations in 2020 modeled for EPA (2011) based on a 36 km grid.

It is readily apparent from Figure 4 that there is considerable variation in air quality impacts across the United States. As previously noted, this information must be combined with information on the spatial variation in population density and baseline incidence rates across the country, which leads to results for any particular health endpoint that are very location specific. Introducing these results into a CGE model will necessarily require aggregation up to the spatial scale at which economic activity is represented in the model. This can be at the state (e.g., USAGE-R51, Dixon et al., 2012), multi-state regional (e.g., EMPAX-CGE, US EPA, 2008), or national level (e.g., IGEM, Goettle et al., 2009). For comparison purposes, Figure 4 includes a delineation of the five multi-state regions in the EMPAX-CGE model. This figure therefore demonstrates the level of aggregation that was used in EPA (2011) to translate the estimated health impacts into their corresponding effects on the regional time endowments and household healthcare expenditures for inclusion in the CGE model. Given the assumption of perfectly mobile capital and labor within the CGE model regions, this aggregation could lead to cases where, because of the regulation, households in one part of a region supply additional labor to industries located in a different part of the region. Therefore, the degree to which this aggregation affects the results will depend on the degree to which there is any potential mismatch between the air quality patterns and economic activity within regions. This issue of aggregation may also affect the results of the CGE model in other ways, for example if the deviation of local healthcare costs from the regional average are correlated with spatial distribution of health impacts. The extent to which these issues might affect the results is unknown.

4.9 Spatial Sorting and Welfare Effects

The approaches to modeling the benefits of air quality improvements discussed in Sections 2 and 3 are based on the assumption that either the spatial distribution of the affected population remains constant or shifts based on an exogenous projection are independent of changes in air quality. However, the change in air quality at a household's pre-policy location may not be the relevant metric for assessing the impacts of the policy on the household. Within housing markets households sort across neighborhoods based on preferences for public goods, social characteristics, and proximity to labor and other markets. Regulations that directly affect the supply of public goods, in this case clean air, can lead heterogeneous households to re-sort as result of the changes in the available choice set.

Beyond the direct exogenous effect of the regulations, some amenities are endogenously determined, creating a feedback between households and their environment. The initial shock associated with the regulation induces sorting amongst households and leads to a potential redistribution of local amenities, until equilibrium is reached within the market. Applications using equilibrium sorting models designed to characterize preference heterogeneity across households have demonstrated that these may be first-order effects for applied welfare analysis (Kuminoff et al., 2010).

A few studies have applied equilibrium sorting models to analyze the welfare impacts of environmental policies. As opposed to the damage function approaches of Section 2 and the CGE approaches of Section 3, these methods focus on assessing the benefits of air quality improvements based on identifying the effects of the exogenous policies that are capitalized in property values. Sieg et al. (2004) considered air quality improvements in Southern California between 10 and 1995, when ozone concentrations were reduced by 19% on average for the study area, with a range of 3% to 33% across individual communities. Sieg et al. (2004) considered both static WTP estimates absent any household resorting and WTP estimates with equilibrium sorting. At the school district level, they found WTP estimates with (costless) re-sorting is on average 50% higher than the case where re-sorting effects are not considered.

This work was extended by Smith et al. (2004) to assess the ex-ante welfare impacts of the CAAA using the forecast changes in air quality for 2000 and 2010 modeled in EPA (1999). They used the equilibrium model of Southern California developed by Sieg et al. (2004) and simulated potential household resorting and welfare impacts based on the changes in air quality for the region forecast by EPA modeling. When allowing for resorting of households they found WTP estimates are on average around 10% higher however, they found significant variation across neighborhoods: with equilibrium welfare impact estimates up to six times greater than simulated estimates without re-sorting in some school districts.

Because sorting in the housing market will be based upon total environmental conditions including all pollutants, equilibrium sorting models may have difficulty assessing the impacts of changes in the atmospheric concentration of single but correlated air pollutants. Therefore, in can be difficult to use the results of these models to assess the benefits of policies that change emissions of multiple pollutants, as can been done with the damage function approach. This class of models to date have also primarily been focused on local or regional analyses capturing single or tightly linked housing markets, as opposed to a national level analysis. Furthermore, equilibrium sorting models may also have difficulty as currently formulated in handling general equilibrium effects such as changes in labor supply and savings, as would be captured in a general equilibrium setting. However, this class of models provides an ability to incorporate other important aspects of the problem. First, they are able to capture a degree of heterogeneity in household preferences and income effects, which may not be currently possible in CGE models. Second, they allow spatial heterogeneity in certain household adjustments to policy changes at a substantially finer level than CGE models, which represent the United States at a spatial resolution of states in the finest case and a single region in the coarsest case. While directly modeling such aspects in a CGE framework would be prohibitive, allowing for behavioral responses in terms of locational choice may be important for understanding the impacts of changes in air quality on health points, such as changes in mortality and morbidity risk, which are entered into the modeling. Additional linkages between modeling frameworks may also offer opportunity for other improvements.

5 Concluding Remarks

EPA has extensive experience with modeling changes in air quality in response to regulations that act to reduce the emission of pollutants into the atmosphere and subsequently estimating households' willingness to pay for those air quality improvements using a damage-function approach. However, while many of the estimates used for households' marginal WTP for improvements in various endpoints are derived, in part, by leveraging the influence environmental quality has on economic activity, these feedback effects are not typically considered in the analysis of air regulations. As shown by the literature discussed in Section 3, these economy-wide benefits of improvements in air quality can be significant. EPA has some experience estimating the economy-wide impact that some of the health benefits borne by improvements in air quality may have (US EPA, 2011), though, as detailed in the charge, technical challenges and important methodological questions remain:

- What are the main conceptual and technical challenges to representing the benefits of an air regulation in an applied CGE framework, and what would be required to overcome them?
- How do we reconcile more traditionally used estimates of WTP by individuals with estimates from CGE models focused on overall welfare, and what insights does a CGE model provide when benefits of air regulations cannot be completely modeled?

The EPA and outside researchers have worked to incorporate some of the human health benefits of improved air quality in CGE models, by characterizing the main impact of changes in the incidence of morbidity and pre-mature mortality as a change in the time endowment. However, the value to a household of a change in their time endowment is not equivalent to their willingness to pay to reduce morbidity or mortality risk. Furthermore, households' preferences, as represented by their utility function within CGE models, may be conditional upon the health state. These observations lead to questions of:

- Is it technically feasible and appropriate to model mortality and morbidity impacts as a change in the time endowment, and if not, what other approaches are available to incorporate mortality and morbidity impacts into a CGE model?
- Because of changes in their health and/or life expectancy, households may experience changes in their relative preferences for goods and services. Could such changes be captured in a CGE model, and under what circumstances would such modeling add value to the overall analysis?

While the predominant source of benefits from improvements in air quality are typically estimated to derive from households' willingness to pay to reduce the risk of premature mortality, there are a notable number of other positive impacts associated reductions in air pollutions as detailed in Section 2. However, CGE models may have value to add in the estimation of certain benefit categories and/or certain benefit categories may be important for understanding the net economy-wide impacts of air regulations, leading to the charge questions:

- What approaches are available to incorporate additional effects of improved air quality in CGE models, and what are the conceptual and technical challenges to incorporating them?
- What techniques are available for incorporating impacts of improved air quality on non-market resources in existing CGE models, and what are the particular challenges to incorporating non-market benefits into a general equilibrium framework (e.g. non-separability)?

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