OECD Environment Working Papers No. 116

Measuring the economic
value of the effects
of chemicals on ecological
systems and human health

Anna Alberini

<https://dx.doi.org/10.1787/9dc90f8d-en>

ENVIRONMENT DIRECTORATE

MEASURING THE ECONOMIC VALUE OF THE EFFECTS OF CHEMICALS ON ECOLOGICAL SYSTEMS AND HUMAN HEALTH - ENVIRONMENT WORKING PAPER No. 116

By Professor Anna Alberini, University of Maryland

OECD Working Papers should not be reported as representing the official views of the OECD or of its member countries.

The opinions expressed and arguments employed are those of the author.

Authorised for publication by Simon Upton, Director, Environment Directorate

JEL Classification: J17, Q51, Q53, Q57, Q58, D61, K32.

Key words: Economic valuation, monetised benefits, regulation of chemicals, toxics, non-market valuation, willingness-to-pay, Value of a Statistical Life, Cost-of-illness, lost productivity, Value of a Statistical Case of Cancer.

OECD Environment Working Papers are available at www.oecd.org/environment/workingpapers.htm

JT03409442

Complete document available on OLIS in its original format

This document and any map included herein are without prejudice to the status of or sovereignty over any territory, to the delimitation of international frontiers and boundaries and to the name of any territory, city or area.

OECD ENVIRONMENT WORKING PAPERS

OECD Working Papers should not be reported as representing the official views of the OECD or of its member countries. The opinions expressed and arguments employed are those of the author(s).

OECD Working Papers describe preliminary results or research in progress by the author(s) and are published to stimulate discussion on a broad range of issues on which the OECD works.

This series is designed to make available to a wider readership selected studies on environmental issues prepared for use within the OECD. Authorship is usually collective, but principal author(s) are named. The papers are generally available only in their original language -English or French- with a summary in the other language.

Comments on OECD Working Papers are welcomed, and may be sent to:

OECD Environment Directorate, 2, rue André Pascal, 75775 PARIS CEDEX 16, France
or by e-mail to *env.contact@oecd.org*

OECD Environment Working Papers are published on
www.oecd.org/environment/workingpapers.htm

This document and any map included herein are without prejudice to the status of or sovereignty over any territory, to the delimitation of international frontiers and boundaries and to the name of any territory, city or area.

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

© OECD (2017)

You can copy, download or print OECD content for your own use, and you can include excerpts from OECD publications, databases and multimedia products in your own documents, presentations, blogs, websites and teaching materials, provided that suitable acknowledgment of OECD as source and copyright owner is given.

All requests for commercial use and translation rights should be submitted to *rights@oecd.org*.

FOREWORD

This paper was prepared by Professor Anna Alberini, University of Maryland for the OECD Workshop on *Socioeconomic Impact Assessment of Chemicals Management* in Helsinki, 6-8 July 2016.

The workshop was organised in co-operation between the Joint Meeting of the Chemicals Committee and the Working Party on Chemicals, Pesticides and Biotechnology (Joint Meeting) and Working Party on Integrating Environmental and Economic Policies (WPIEEP), and was hosted by the European Chemicals Agency, with funding contributions from the European Commission, the European Chemicals Agency and the American Chemistry Council.

The paper underwent revision and takes into account feedback received from Delegates during and after the workshop and comments received from the Joint Meeting and WPIEEP by written procedure.

The opinions expressed and the arguments employed are those of the author.

TABLE OF CONTENTS

FOREWORD.....	3
ABSTRACT	5
RÉSUMÉ.....	5
1. Introduction	6
2. Valuation Methods.....	8
A. Market methods	8
B. Non-market valuation: The travel cost method	9
C. Non-market valuation: Hedonics.....	11
D. Non-market valuation: Stated preference methods.....	12
3. Valuing human health effects and changes in productivity	13
A. Nature of the human health endpoints	13
B. Valuing morbidity: Acute illnesses	14
C. Effects on productivity	17
D. Developmental effects, low birth weights, and other infant and child outcomes.....	18
E. Reproductive effects	21
F. Mortality effects	22
4. Valuing cancer outcomes.....	27
5. Chemicals as emerging pollutants	30
6. An Example: The US EPA NESHAP emissions rule.....	32
7. Discussion and conclusions	36
REFERENCES	39

Tables

Table 1. Categories of benefits from regulating chemicals.	8
Table 2. Adverse human health effects avoided each year by the rule.....	35
Table 3. Annual Benefits and Costs of the final rule in 2016 (2007 USD).	35

Figures

Figure 1. Consumer surplus change associated with improved environmental conditions at the site. ...	9
Figure 2. Competitive market after the regulation.....	36

ABSTRACT

This paper reviews and discusses the existing methods for placing a value on the effects of chemicals on human health and the environment. It surveys both methods and non-market methods, discussing their advantage and limitations. For example, when valuing non-fatal illnesses, the cost-of-illness approaches captures labour income lost to illness and medical expenditures undertaken to mitigate the illness, but fails to account for the value of the disutility of the illnesses. The paper also discusses mortality risk valuation, and the widely used metric termed the Value of a Statistical Life, the difficulties associated with estimating it, and the appropriateness of any adjustments for futurity, age, and the nature of the risk itself. Finally, the paper takes up the issue whether the source of the health risks (e.g., chemicals versus other forms of pollution versus others) affects how much the public values reducing those risks.

Keywords: Economic valuation, monetised benefits, regulation of chemicals, toxics, non-market valuation, willingness-to-pay, Value of a Statistical Life, Cost-of-illness, lost productivity, Value of a Statistical Case of Cancer.

JEL codes: J17, Q51, Q53, Q57, Q58, D61, K32.

RÉSUMÉ

Cet article décrit et analyse les méthodes existantes destinées à attribuer une valeur aux effets des produits chimiques sur la santé humaine et l'environnement. S'intéressant aux méthodes marchandes aussi bien que non marchandes, il examine leurs avantages et leurs limites. Par exemple, dans l'évaluation des maladies non mortelles, les approches fondées sur le coût des maladies rendent compte de la diminution des revenus du travail due à la maladie et des dépenses médicales engagées pour lutter contre cette dernière, mais pas de la désutilité liée aux maladies. Il est également question, dans cet ouvrage, de l'évaluation des risques de mortalité, de la valeur d'une vie statistique – indicateur couramment employé –, des difficultés rencontrées dans son estimation, et du bien-fondé des corrections apportées au titre de la latence, de l'âge et de la nature du risque lui-même. Enfin, les auteurs se demandent si la source des risques pour la santé (produits chimiques, autres formes de pollution ou autres facteurs) a une incidence sur la valeur que tout un chacun attache à la réduction de ces risques.

Mots clés : Évaluation économique, bénéfices monétaires, réglementation des produits chimiques, substances toxiques, évaluation non marchande, consentement à payer, valeur d'une vie statistique, coût des maladies, perte de productivité, valeur d'un cas statistique de cancer.

Classification JEL : J17, Q51, Q53, Q57, Q58, D61, K32.

1. Introduction

The purpose of this document is to summarise methods for estimating the benefits of regulating chemicals and present the most important categories of benefits, along with current practice, state of the art and difficulties. The costs of chemicals regulations are also briefly discussed in this report. Attention is restricted to chemicals that have potential adverse effects on ecological systems, the environment and/or human health when released or disposed of improperly, through workplace exposure, or through consumer contact with these products. In a companion paper, Chiu (2016) discusses risk assessment procedures used by government agencies and international organisations for three such chemicals—a flame retardant (hexabromocyclododecane), a paint thinner (dichloromethane) and perfluorooctanoic acid and its salts, which have a number of industrial applications, but a broad list exists that includes industrial solvents and lubricants, flame retardants, aluminum can linings, plasticisers, pesticides, various food and cosmetics additives, pharmaceutical agents, and many others.

Benefit-cost analyses of proposed regulations require estimating the *social benefits* of the regulations and comparing them with the *social costs* of the regulations. It is sometimes argued that monetising the effects of a regulation, and looking at total monetised benefits and costs, is in principle easier than comparing many different categories of effects measured in different units (e.g., symptom-days, changes in fish population, etc.).

The social costs of the regulations are the sum of all opportunity costs incurred as a result of the regulations (US EPA, 2010). These opportunity costs are the value lost to society of all of the goods and services that will not be produced and consumed if the regulated entities comply with the regulations and reallocate resources away from those goods and activities and towards pollution abatement or replacement of a chemical with a safer one. Cost analyses can be conducted in a number of ways—for example, with partial equilibrium analyses of a single or few markets when the effects of the regulations are limited to one or few markets, and general equilibrium analyses when the regulations are expected to have significant impacts economy-wide.¹ An example is presented in Section 6 of this paper for a rule affecting emissions of mercury from power plants in the United States (US EPA, 2012).

Economic theory holds that the benefits of regulating chemicals are correctly computed as the beneficiaries' willingness to pay for such regulations. Willingness-to-pay (WTP) is defined as the maximum amount that can be subtracted from an individual's income in the regulated state to keep his or her expected utility at the same level as when the chemical is unregulated.

For example, it may be possible to observe consumers' willingness to pay for a safer variant of the product. Alternatively, it is possible to list the physical effects of eliminating the chemical, regulating it, or replacing it with a safer one on ecological systems, the environment and human health, and to attach a value to each such category of effects. This second approach is normally referred to as "damage function approach." Clearly, when this second approach is used, the analyst's first task is to describe and quantify each such of these physical effects, and to identify the persons affected by them (the beneficiaries). An example of such effects and such parties is provided below.

1. It is noted that benefit-cost analyses are sometimes supplemented by economic impact analyses to assess the employment effects of proposed regulations and to examine its distributional impacts—for example, are the costs disproportionately borne by certain industries, and the benefits experienced by specific subpopulations? Are vulnerable groups taken into due consideration?

In theory, if the users of the product correctly understand the adverse effects associated with the product's chemical(s), the two approaches should result in similar estimates of the benefits. Implicit in the damage function approach is the assumption that the beneficiaries' utility is affected by the endpoints, but not about pollution per se or by the nature of the substance that is causing such endpoints.

In current practice both approaches are used. This makes it difficult to compare studies and has potentially important implications for benefit transfer, the practice of applying benefits estimated at one locale (or class of chemicals) to another locale (or another class of chemicals) (see Navrud, 2016). From the point of view of estimating the benefits of reducing harmful chemical exposures, it is sometimes thought that the damage production function is easier to work with, on the grounds of the fact that people are presumably familiar with these endpoints but not with pollution concentrations per se that would be affected by policy. Moreover, the beneficiaries of the policy are usually members of the general public, and they would find it extremely difficult to quantify all of the possible ecological or health effects of a given chemical. Understanding a chemical's impact on biophysical processes and then on ecological services and endpoints, and human health effects, however, can be extremely complex even for the experts. Moreover, there are high levels of uncertainty about both the physical effects and the monetary values that may be attached to them.

This paper presents methods that can be applied with both approaches, and specific categories of benefits that are important for policy purposes.

This paper expands on the procedures typically followed by the US EPA with water pollution and identifies the following possible categories of benefits from regulating chemicals in products:

A. Environmental benefits

1. Market costs (the losses/gains to commercial fisheries, other industries that harvest natural resources affected by the presence of the chemical in the environment, agricultural production likewise affected by the presence of the chemical);
2. Recreation benefits;
3. Aesthetics benefits (e.g., improvements in visibility, odours, drinking water turbidity);
4. Non-use values (see below);
5. The (avoided) cost of supplying alternate ecosystem services (for example, if a wetland serves as a flood control mechanism or helps recharge groundwater, and these functions are impaired by the presence of pollution, the cost of obtaining similar services from other sources or the cost of developing an artificial wetland);
6. Other avoided costs (e.g., cost of sludge disposal from processing wastewater in the case of chemical resulting in water pollution).

B. Human health benefits

1. Health effects to workers due to occupational exposures;
2. Health effects to consumers exposed to the chemical through contact with the product or through environmental exposures;

C. Other productivity benefits

1. Avoided loss of worker productivity even in the presence of subclinical or no health symptoms.

These categories of benefits are summarised in Table 1.

Table 1. Categories of benefits from regulating chemicals

Type	Type of exposure /effect (population experiencing the effects)	Specific types of benefits	Methods
A. Effects on environment and ecosystems	<ul style="list-style-type: none"> -Effects on agriculture and other harvestable resources sold on regular markets (producers, consumers) -Effects on resources with recreational use (current and potential users of the resource) -aesthetic values -Changes in non-use values (general population) - Effects on ecosystem functions (parties affected by the ecosystem functions) - Avoided treatment costs 	<ul style="list-style-type: none"> - Consumer and producer surplus change -Consumer welfare, WTP for access and use of the resource -WTP -Value of the ecosystem functions 	<ul style="list-style-type: none"> Market methods -Non-market valuation methods -Cost of alternate supplies of ecosystem functions
B. Human health effects	<ul style="list-style-type: none"> -Workers exposed in the workplace -General population exposed through product use and/or environment 	WTP to avoid or reduce: <ul style="list-style-type: none"> - Acute illnesses - Non-fatal chronic illness - Mortality effects - Cancer (fatal and non-fatal) - Reproductive effects - Developmental and neurodevelopmental effects 	Non-market methods
C. Other worker productivity effects	<ul style="list-style-type: none"> -Workers exposed in the workplace -General population exposed through product use and/or environment 	Loss of productivity at work, even when symptoms are subclinical	Market methods.

The remainder of this document is organised as follows. Section 2 presents succinctly the most commonly used valuation methods. Section 3 describes adverse human health effects of chemicals and the valuation of these health endpoints. Section 4 discusses issues related to valuing cancer outcomes. Section 5 discusses valuing emerging risks. Section 6 presents a recent rule on mercury emissions from power plants in the United States and the associated benefit-cost analysis. Section 7 concludes.

2. Valuation Methods

What follows discusses market and non-market valuation methods, following the order in which they appear in Table 1. The emphasis in what follows is on the methods, not on the specific categories of benefits that the methods seek to estimate.

A. Market methods

When harmful chemicals deposit in soil or water, and harm agricultural productivity, timber harvest and other biota that has commercial value (e.g., fisheries), the price of agricultural output or harvest is expected to increase, the quantity traded on the market to decrease and producers are expected to incur additional costs to purchase substitute inputs (e.g., fertilisers, different water supplies, etc.). The benefits of reducing harmful chemicals released into the environment are thus the producer and consumer surplus losses avoided by regulating that chemical.

B. Non-market valuation: The travel cost method

In many cases, however, environmental quality, ecological systems and environmental assets are not bought and sold in regular markets, and so it is necessary to use non-market valuation methods. If a substance affects the quality of natural resources that have recreational use (for example, a chemical or pollutant affects fish abundance in bodies of water used by recreational anglers), the travel cost method can be used to estimate the welfare change associated with a change in quality at the affected sites.

The travel cost method relies crucially on the weak complementarity assumption—namely, that if site quality improves, the demand for site visits will increase. It is limited to capturing use values on the part of current or prospective users of the resources. It can be and has been deployed with a variety of pollutants discharged into the environment or substances “absorbed” by wildlife.

The notion behind travel cost models is that people often pay only a nominal charge (or nothing at all) to access the natural resource site being studied, a nominal charge that bears no relationship with the actual cost of maintaining or managing the resource.² People do, however, incur out-of-pocket and opportunity-cost-of-time costs to visit the resource, and these can be used to construct the price of a visit, which typically varies across visitors, and estimate a demand function for site visits. In a simple, single-site framework, and assuming a linear demand function:

$$(1) \quad \text{Trips}_i = \alpha + \beta \cdot \text{price}_i + \mathbf{x}_i \boldsymbol{\gamma} + \varepsilon_i$$

where i denotes the individual and $\mathbf{x}$ is a vector of individual characteristics that modify the demand function for site visits. When the quality of the site is improved, the demand function is shifted out, in that more visits are demanded for any given level of the price of a visit. In other words, the intercept becomes $\alpha' > \alpha$. The recreational benefits associated with improving environmental quality are thus measured as the change in consumer surplus, i.e., the area of trapezoid ABCD in Figure 1.

Figure 1. Consumer surplus change associated with improved environmental conditions at the site

2. In 2015, for example, the access fee to 133 national parks in the United States ranged from USD 3 to USD 25. Access to many state or county parks is free.

Difficulties associated with the travel cost method include i) the definition of the market, namely the population over which the consumer surplus must be aggregated, ii) the correct calculation of the price of a trip, including the opportunity cost of time, iii) the inclusion of the price of one or more substitute sites in the regression equation, and iv) the selection of the appropriate functional form for the regression.

Much recent work based on the travel cost method has relied on discrete choice models, which allow the analyst to explicitly incorporate substitution across sites when the quality at one is compromised or (conversely) improved. Discrete choice models rely on a random utility framework, which posits that individuals derive utility from the attributes of the sites and from income. The indirect utility experienced by individual i at site j is usually assumed to be a linear function of the attributes of the site and residual income, namely income minus the cost of accessing and using the site:

$$(2) \quad V_{ij} = \mathbf{x}_{ij}\boldsymbol{\beta} + \gamma \cdot (\text{Income}_i - C_{ij}) + \varepsilon_{ij}.$$

Coefficients $\boldsymbol{\beta}$ in equation (2) are the marginal utilities of the attributes and γ is the marginal utility of income. The error term ε captures all unobserved factors that influence utility. These may be known to the individual but not to the analyst. On assuming that ε is independently and identically drawn from the type I extreme value distribution,³ it is easy to show that the probability of selecting site k is

$$(3) \quad \Pr(k) = \exp(\mathbf{w}_{ik}\boldsymbol{\delta}) / \sum_{j=1}^J \exp(\mathbf{w}_{ij}\boldsymbol{\delta})$$

where $\mathbf{w}$ is a vector that subsumes $\mathbf{x}$ and residual income, and $\boldsymbol{\delta}$ is comprised of $\boldsymbol{\beta}$ and γ . Equation (3) is the contribution to the likelihood in a conditional logit model.

Typical attributes include water quality, species abundance, scenic quality, presence of facilities that make visits easier (parking structures, lodging, access, etc.), any environmental health risks, and others. The welfare gain associated with improving a site's environmental conditions are computed as

$$(4) \quad \frac{1}{\gamma} \left[\ln \left(\sum_{j=1}^J \exp(\tilde{V}_{ij}^1) \right) - \ln \left(\sum_{j=1}^J \exp(\tilde{V}_{ij}^0) \right) \right]$$

where $\tilde{V}$ denotes the predictable component of indirect utility (i.e., the right-hand side in (2), without the error term), and the superscripts 0 and 1 refer to the initial and final environmental conditions. The WTP for a marginal change in any given attribute l (for example, environmental quality or fish catch rate) is β_l / γ .

Conditional logit models imply the independence of irrelevant alternative assumptions, which results in a rather restrictive substitution pattern across sites when the attributes of one site are changed. Those wishing to relax this assumption have turned, when appropriate, to nested logit models, which partially relax this assumption, or mixed logit models, which carry the additional advantage of allowing the

3. A type I extreme value random variable has a cumulative distribution function $F(\square) = \exp(-\exp(-\square))$. The conditional logit model assumes that the error term within the indirect utility associated with each alternative is an independent draw from such a distribution. In other words, when examining the alternatives within someone's choice set, the error terms are independent from one another. The assumption that the errors are i.i.d. and from the type I extreme value distribution results in the independence of irrelevant alternatives.

marginal utility of an attribute to vary across individuals.⁴ An additional difficulty is how to specify the choice set faced by an individual for any given choice occasion (Haab and Hicks, 1999).

C. *Non-market valuation: Hedonics*

Hedonic pricing methods seek to identify the value markets place on each attribute of a good or a service – including, when possible, its environmental and health effects. Hedonic housing price methods are one of the best known examples of this approach.

Hedonic housing price approaches can be applied whenever there is reason to believe that environmental or ecosystem quality will be reflected in housing values, such as e.g. the level of a given chemical in the nearby environment. This approach is based on observing housing sales, and typically involves regression analyses where the (log) price of a home is regressed on structural characteristics of the dwelling characteristics (size, number of floors, house type dummies, construction quality dummies, age of the home) and neighbourhood characteristics, including environmental quality. Structural characteristics usually account for most of the variation in the value of the home, whereas environmental quality usually picks up a smaller—and yet, often still meaningful—portion of the variation in the value of the home.

For the hedonics approach to work, there must be sufficient variation in environmental quality over time and across space. The approach is often criticised because 1) it is difficult to account for the neighbourhood characteristics that truly affect property values, 2) environmental quality is sometimes so strongly spatially correlated with other unobserved amenities or disamenities that the analyst attributes effects to environmental quality that are truly due to other factors, and 3) it is assumed, without testing, that real estate market participants are informed about the changes in the environmental quality as measured by the analyst.

Recent high-quality application of hedonics have sought to exploit exogenous environmental quality “shocks” to identify the value of the change in environmental quality (e.g., Davis, 2004; Currie et al., 2012) or have resorted to the repeat sale approach. Assuming that nothing else changes (or that, if it does, it is adequately controlled for in the regression analysis), with a repeat sales model the value of the change in environmental quality is captured by the difference in the value of the same home before and after the change in environmental quality. One advantage of this approach is that it filters out any other unobserved factors that influence property values. In practice, it is assumed that

$$(5) \quad \ln P_{it} = \alpha_i + x_{it}\beta + E_{it}\gamma + \varepsilon_{it}$$

where t denotes the time period, α_i is a house-specific fixed effect, x represents time-varying factors affecting the value of the home (e.g., retrofits and modifications to the home, or changes in the neighbourhood), and E is the time-varying measure of environmental quality.

1. On taking first differences, one obtains:

$$(6) \quad \Delta \ln P_{it} = \Delta x_{it}\beta + \Delta E_{it}\gamma + e_{it}$$

which allows the analyst to identify the effects of the change in environmental quality through an ordinary least squares (OLS) regression.

4. Latent class models are an alternate approach for allowing the marginal utilities to vary in a unobservable fashion across recreationists.

One concern with the repeat sale approach is that little attention has been paid to whether homes that are sold repeatedly enough to be included in this type of analysis are sufficiently representative of the housing stock and the real estate transactions at a given location.

Hedonics have been used to examine the effects of water contamination, air pollution, health risks of unknown origin (Davis, 2004) and have been recently proposed to study the effects of “fracking,” the process of drilling and injecting fluid into the ground at a high pressure in order to fracture shale rocks to release natural gas inside (Bennet and Loomis, 2015; Muehlenbachs et al., 2013). Fracking fluids often contain toxic chemicals that end up in groundwater used for drinking or other domestic purposes.

Hedonics can be used with a single or multiple categories of effects from changed environmental conditions. Guignet et al. (2015) and Guignet et al. (2016) use property value hedonics to capture the depreciation in housing values associated with groundwater contamination due to agricultural runoff and groundwater contamination due to leaking underground storage tanks (such as those at gasoline stations), respectively. (It is noted that in the latter case, there are risks of explosions, fires, etc. associated with the volatile and flammable by-products of gasoline leaks. The change in property values may capture these as well as the risks of illness and cancer.)

Importantly, Guignet (2014) shows that conventional hedonics approaches (where one simply looks at the change in property values associated with changed environmental conditions or human health risks) capture the benefits of changed environmental conditions only when owners would sell their homes regardless of the change in environmental quality. If the shock to environmental conditions affects the decision to sell, then welfare effects calculations are much more complex and demanding, in that they require specific information about household characteristics, income, and preferences.

Both travel cost and housing price hedonic methods share one common difficulty—the definition of the “market.” In other words, how broad a geographical area is it reasonable to assume to be affected by changes in environmental quality at the study locale?

To conclude this discussion of hedonic pricing methods, it should be noted that hedonics can be, and indeed often are, conducted by regressing the price of any given product on its characteristics. The coefficients on the characteristics are the marginal WTP for the characteristics. When the product is a car and the attributes include safety (in the case of a car crash), the approach can be used to value health risk reductions (see Andersson, 2005). It is also possible to use hedonics to value reduced health risks from other types of products. For example, Maguire et al. (2004) apply hedonics to baby food (which can be made with conventional or organic ingredients), and infer the value of avoiding cancer outcomes from the marginal WTP for organic baby food, combined with assumptions about intake and potency of the pesticide residue.

D. Non-market valuation: Stated preference methods

Stated preference methods can be used to value changes in pollution levels per se, the presence and/or quality of environmental assets (e.g., groundwater in pristine condition), changes in ecosystem health, quality, and appearance, access to environmental or natural resources that have recreational uses, and other environmental and health “goods.” Because they rely on what people say that they would do under specified hypothetical circumstances, they can be applied with levels of environmental quality, ecosystem or human health that do not currently exist, and with both users and non-users of the resources. Indeed, only stated preference methods capture non-use values, i.e., the value that people place on a resource because of its existence (“existence value”), because they wish to preserve it for future generation (“bequest values”), and just in case they may want to use it themselves in the future (“option value”).

Contingent valuation is a popular survey-based stated preference method. It relies on asking individuals to report information about their willingness to pay for a specified change in environmental quality, level or quality of a resource, etc. These must be clearly defined to the respondent in the course of the survey.

Choice experiments create hypothetical and stylised environmental “goods” defined by attributes. Respondents are usually asked to indicate which is their most preferred “good” out of a choice set with K goods, where $K \geq 2$. In many cases, the choice set allows for a “current situation” or “opt out”.

Contingent behaviour questions are sometimes used in combination with travel cost methods. They ask people if they would continue to visit, or stop visiting altogether, a recreational site if the price of a visit was \$ X (where \$ X is greater than the current cost of accessing the resource) and thus can be used to figure out an individual’s “choke price” or improve information about the slope of the individual’s demand function for site visits.

It should be noted that there has been little attempt to see if the public’s preferences and WTP are affected by the nature of the pollutant, the origin of the substance (natural or man-made), and the level of certainty about the chemical’s causation of the effects on ecological systems and human health that analyst wishes to value.

It is also important to note that the same categories of benefits can be captured through more than one valuation method, and that indeed deploying more than one method provides an excellent check of the validity of the benefit estimates. One must be careful, however, to avoid double counting (US EPA, 2010).

3. Valuing human health effects and changes in productivity

Some chemicals have been linked with cardiovascular effects, neuro-developmental effects (leading to a loss of IQ, ADHD, and autism), cancer, chronic renal and liver disease, diabetes, obesity, reproductive health problems, bone fragility, and other health endpoints. See Trasande (2016) for a discussion of the uncertainty about causation and attributable disease burden.

Exposure may occur via the environment through air, soil, dust, or water that contain these chemicals, or via food or other products, by dermal, oral or inhalation exposure. Some chemicals pose both acute and long-term risks to the persons that use them on an occupational basis or work in the facilities that manufacture them, and others affect consumers or members of the general public through environmental exposure or use of the products that contain them.

For many chemicals, dose-response functions are available that predict the expected number of cases of illness, or the probability of developing cancer, other diseases or dying, associated with a given level of exposure or ambient concentration of the chemical. These functions have been developed on the basis of epidemiologic studies at single locale or selected populations, meta-analyses of studies done in different places and on different populations, or by extrapolating to humans the findings from bioassays based on laboratory animals. In some cases the effect of exposure to a substance are computed from dose-response responses developed for a different type of pathway. For example, the US EPA applies the cessation lag model estimated from cigarette smokers to predict the reduction in the cases of bladder cancer deriving from reducing the level of arsenic in drinking water.

A. *Nature of the human health endpoints*

Health endpoints can be classified in several ways. One classification might be based on the severity of the effects – morbidity effects (acute and chronic) and mortality effects. Morbidity effects can further be

classified on the basis of the impact that they have on normal daily activities. For example, a restricted activity day would be one where normal daily activities are impaired to some extent, and a bed disability day would be one in which the illness forces the individual to stay in bed.

Even when attention is restricted to mortality effects, much current valuation and epidemiologic literature classifies mortality effects as acute (such as excess mortality associated with a spike in air pollution) and chronic (due to chronic conditions caused or exacerbated by the chemical).

Another type of classification may be done depending on the organs affected or the specific type of functions affected (respiratory, cardio-circulatory system, reproductive system, etc.). Effects can be also be examined by the age of the affected persons – children, adults and the elderly –and by whether the effects of exposure are immediate or delayed over time.

B. Valuing morbidity: Acute illnesses

This section presents a simple model for valuing morbidity linked with pollution or chemical exposures. The model is suited for minor, acute illnesses, and is an application of the household production function framework.^{5, 6}

Assume that utility (U) increases with aggregate consumption (X) and leisure (L), but is negatively affected by sick days (D).

$$(6) \quad U = U(X, L, D; \mathbf{Z}_U)$$

where $\mathbf{Z}_U$ is a vector of individual characteristics capturing preferences for income, leisure, and health.

In utility function (6), pollution does not influence utility directly, but does have an indirect influence by triggering illness and causing sick days. The dose-response function summarises the relationship between pollution or chemical concentrations and health outcomes:

$$(7) \quad D = D(P, A; \mathbf{Z}_D)$$

Equation (7) shows that sick days depend on pollution P and averting activities (A), undertaken by the individual to reduce exposure to pollution.⁷ $\mathbf{Z}_D$ is a vector of individual characteristics allowing for

-
5. In the household production function approach, an individual household derives utility from combining private and public goods (e.g., air quality, transportation infrastructure, etc.). Some of these private or public good do not provide utility per se, but only via other goods or services used by the household. In the model described in this section, it is assumed that the household does not experience utility or disutility from pollution per se, but does experience disutility from the sick days that pollution causes.
 6. A simple allergic reaction or respiratory ailment that clears relatively quickly and does not recur, for example, would be covered by this model. The model is suited for a working adult, or for a household with working adults that must curtail their work hours to take care of a sick family member. In this form, the model does not allow for different utility functions for different members of the household (as might happen, for example, when a parent experiences disutility from having to take care of a sick child and from knowing that the child is sick and suffering because of the illness).
 7. In practice, the literature typically produces reduced-form concentration-response functions without disentangling the contribution of pollution, and the offsetting effect of averting activities (see Moretti and Neidell, 2009). The air pollution literature, for example, links hospital admissions, restricted activity days, work loss days, cases of lower respiratory tract illnesses or new cases of chronic bronchitis to particulate matter or ozone.

individual predisposing factors and baseline health. The individual chooses the levels of L , X , and A to maximise utility subject to budget constraint:

$$(8) \quad y + w \cdot [T - L - W(D(P, A))] = X + p_M \cdot M(D(P, A)) + p_A \cdot A.$$

Equation (8) posits that the individual must allocate his time between work and leisure and spend income on aggregate consumption, medical care M , which in turn depends in the number of sick days, and on the averting activity A . The prices per unit of M and A are equal to p_M and p_A , respectively. Work time lost to illness (W) is in the budget constraint because it reduces the work time available to the individual.

Based on this model, what is the individual's willingness to pay (WTP) to reduce pollution? Higher pollution levels make individuals worse off because they increase sick days, so, all else the same, individuals experience higher utility levels when pollution is lower. The WTP is defined as the maximum amount of money that can be subtracted from the individual's income at the new, lower level of pollution to keep him or her at the same utility level as with the initial levels of pollution and income. Formally, WTP satisfies:

$$(9) \quad V(y - WTP, w, p_M, p_A, P_1) = V(y, w, p_M, p_A, P_0)$$

where $V(\cdot)$ is the indirect utility function and P_0 and P_1 are the initial and final levels of pollution with $P_0 > P_1$ so that environmental quality improves. Harrington and Portney (1987) show that WTP for a small change in pollution can be decomposed into:

$$(10) \quad WTP = w \frac{dW}{dP} + p_M \frac{dM}{dP} + p_A \frac{dA^*}{dP} - \frac{U_D}{\lambda} \cdot \frac{dD}{dP}$$

where A^* is the demand function for A , and dA^*/dP gives the optimal adjustment of A to a change in pollution. Equation (10) indicates that the WTP for a small reduction in pollution includes four components: (i) marginal lost earnings, (ii) marginal medical expenditures, (iii) the marginal cost of the averting activity, and (iv) the disutility of illness, converted into dollars through dividing by the marginal utility of income.

Equation (10) can be rearranged as follows:

$$(11) \quad WTP = \frac{dD}{dP} \left[w \cdot \frac{dW}{dD} + p_M \frac{dM}{dD} + p_A \frac{dA^*}{dD} - \frac{U_D}{\lambda} \right]$$

7. (cont.) To illustrate, an increase of $10 \mu\text{g}/\text{m}^3$ of PM_{10} raises population-wide respiratory hospital admissions by about 7 people per 100 000 per year, and cardiac respiratory admissions by 4.34 people per 100 000 per year. The same change in air pollution is predicted to increase restricted activity days by 606 per 1000 people per year and work loss days by 207 per year for every 1000 adults aged 15-64 (Rabl et al., 2014, p. 100-107). To the extent that a substance is part of particulate matter, these concentration-response function slopes could be used to predict the air pollution impacts of reducing releases of that chemical into the environment. With heavy metals, dioxins and PCBs, benzene and other volatile organic compounds, polycyclic aromatic hydrocarbons and formaldehyde, the focus is on predicting the cases of cancer associated with exposure. Dioxins can cause acute effects like chloracne, but considerable uncertainty surrounds whether it truly causes cancer. It is conjectured that exposure to high levels of formaldehyde may result in acute effects such as irritation of the eyes and airways, but no reliable dose response are available (Rabl et al., 2014), and there is no convincing evidence about cancer risk at low ambient levels.

showing that the marginal WTP can be equivalently expressed as the slope of the dose-response function multiplied by the marginal value of illness.

Equation (11) has important implications for valuation purposes. First, it provides support for the damage function approach, namely for assessing dose- or concentration-response functions and combining them with the WTP to avoid illness per se – whether or not individuals know that the illness is associated with pollution exposures.

Second, it shows that one possible way to estimate the value of morbidity associated with pollution exposures is to focus on the first two items in the brackets in (11), ignoring the rest. This approach is termed the cost of illness approach, and is used extensively in policy analyses and in various assessment efforts. The US EPA, for example, has issued a handbook with cost-of-illness figures for a variety of health endpoint linked with pollution (air pollution and otherwise) (US EPA, 2007). The ExternE project⁸ and its sequels have deployed the cost-of-illness approach to monetise a variety of health endpoints, including chronic bronchitis.

Cost-of-illness calculations are relatively easy to do, have been compiled by government agencies for a variety of illnesses, and do not require conducting surveys of individuals. It is clear, however, that they provide only a lower bound for the true WTP to avoid the illness, and as such they understate the true costs of inaction. Chestnut et al. (1996), Dickie and Gerking (1996), and Alberini and Krupnick (2000) compared full WTP for angina pectoris, doctor visits, and minor respiratory illnesses, respectively, with the cost of those illnesses as reported by survey respondents in the US and Taiwan, and concluded that the full WTP is two to four times larger than the cost of illness alone.

Bartik (1988) has focused on the third item in the brackets in equation (11), namely $p_A \frac{dA^*}{dD}$, but stringent assumptions about the household production function technology are required for his approach to produce estimates of the WTP to avoid illness. Simpler averting costs calculations are conducted by simply tallying the expenditures associated with purchasing and running air filters, purchasing bottled water when the domestic water supply is unsafe, etc. These simple calculations – much like the cost-of-illness approach – underestimate the true WTP to avoid the illness, but provide useful lower bounds to the value of avoiding illness.

The only way to capture the full WTP to avoid illness is to conduct a stated-preference study, i.e., surveys where individuals are asked to report their WTP to reduce chemical pollution or risk or avoid the illness associated with exposure. Contingent valuation methods do just that, and WTP figures based on this and other non-market methods (such as discrete choice experiments) have been used in assessment efforts.

The model described by equations (6)-(8) contains a number of important assumptions. The individual does not derive disutility directly from pollution, but only indirectly because pollution causes sick days. An implication of this assumption is the decomposition of total WTP into the components shown in equations (10) and (11), and the fact that the value of illness does not change with the nature of the pollutant. This is important for benefit transfer and for valuing illnesses associated with previously unregulated (or merely unstudied) chemicals: If the symptoms associated with them are the same as for other illnesses, then the WTP is the same.

The individual is also assumed to have flexible work hours, and while pollution can force the individual to adjust his labour supply, it does not otherwise influence productivity and the wage rate. These

8. www.externe.info/externe_d7/.

assumptions may be reasonable when the individual is self-employed, when the effects of pollution on the labour market are sufficiently small and localised, and when individuals are not exposed to pollution at their workplace. It is relatively straightforward, however, to amend the above model to accommodate for the situation when individuals are required to work a fixed number hours, and for sick leave, the cost of which is generally absorbed by the employer and/or taxpayers.

If one wishes to amend the model to allow for pollution or the nature of the chemical to influence utility directly, then an additional term must be added to the right-hand side of equation (11), namely the marginal utility or disutility from the chemical, divided by the marginal utility of income. This additional term stays outside of the bracket in the right-hand side of (11). An extensive literature has studied the perceptions of environmental or health risks, and the effect of attributes such as voluntariness, blame, controllability, and others (Tsuge et al., 2005; Cropper and Subramanian, 2000), suggesting that this additional term may well be present with different chemicals. Its exact magnitude is likely to depend on chemical (and likely on exposure pathway as well), and at this stage, not enough information is available to be able to apply it with confidence for every new chemical in products to be regulated. (Also see Section 5 for a discussion of the attributes of chemicals.)

Extensions of this basic model are applied to study the damage costs of non-fatal chronic illnesses, such as obesity, bone fragility associated with chemicals exposure, diabetes, obesity, hypertension, etc. (Drake, 2016; Trasande, 2016). These applications have focused on the cost of illness (work income and productivity lost to illness, medical expenses, cost of home care for those needing assistance, etc.) but have not incorporated the value of the disutility from being ill and suffering. These applications have usually explicitly recognised this limitation.

In practice, studies that have elicited the full WTP of individuals to avoid illness have deployed one of two possible approaches. The first simply describes the illness (and its attending symptoms, severity, and duration of the episode or frequency of bouts over a specified period of time) to the respondents, asking them to imagine that they are about to experience such an illness – with probability 1. How much would they be willing to pay to avoid it altogether?

The second approach tells respondents that they have a probability q of experiencing such an illness, and asks them to indicate their WTP to reduce such risk by Δq . The value of a “statistical” case of the illness is thus the WTP divided by Δq . One problem with this approach is that – probably because of well-known difficulties with processing probabilities (see Corso et al., 2001, for the effectiveness of visual displays in aiding respondents with probabilities) – the values per case are sometimes implausible. Hammitt (2016), for example, reports WTP of the order of USD 6 800 – 10 000 to avoid a one-day episode of food poisoning with relatively mild symptoms.

Both approaches have struggled with illnesses that respondents are not familiar with.

C. *Effects on productivity*

This section draws a parallel between the valuation of the effects of chemicals and the valuation of air pollution. In the context of air pollution, it is often assumed that air pollution causes no other impact on labour and labour productivity other than the loss of work days. Estimates of the cost of absenteeism are available from ExternE and related research projects. A study by the Confederation of British Industry, which surveyed representatives of the manufacturing, service, and public sector, estimates the median direct cost to business per employee-day absence to be EUR 85. The mean cost is EUR 114. The survey respondents were asked to estimate the salary costs of absent employees, the costs of replacement staff, and lost service and production time. The study also provided an estimate of the indirect cost of absence, due to lower customer satisfaction and poorer quality of products and services. The mean indirect cost is

EUR 168, but this figure should be used with caution due to the low item response rate for this particular question. Adjustments of these figures to the mean earnings of workers in different EU countries have resulted in total figure of EUR 295 per employee absence day (Rabl et al., 2014, page 384-385).

It should be emphasised that in projects like ExternE and its sequels, air pollution affects productivity in only one pathway: It gets people sick and forces them to miss work. No other pathways are covered.

However, that air pollution affects labour supply has been observed at a variety of locales. Hanna and Oliva (2011) examine an exogenous variation in air pollution due to the closure of a large refinery in Mexico City to understand how air pollution impacts labour supply. The closure led to an 8% decline in pollution in the surrounding neighbourhoods. They found that a 1% increase in sulphur dioxide results in a 0.61% decrease in the hours worked. The effects do not appear to be driven by labour demand shocks nor differential migration as a result of the closure in the areas near the refinery, and so Hanna and Oliva conclude that it is correct to ascribe the changes in labour supply to changes in air pollution levels.

One important but understudied issue is whether air pollution affects not only hours worked, but also the productivity of labour. This might be the case if, for example, pollution causes subclinical symptoms that do not result in loss of work time, but reduce productivity at work.

Zivin et al. (2011) measure the hours worked and labour productivity of farm workers in Southern California. Chang et al. (2014) study the effect of outdoor air pollution on the productivity of indoor workers at a pear-packing factory in Northern California. They find that an increase in fine particulate matter (PM_{2.5}) outdoors leads to a statistically and economically significant decrease in packing speeds inside the factory. Specifically, a 10-unit change in PM_{2.5} decreases worker productivity by 6%. They find that pollutants that do not travel indoors, like ozone, have little effect and that PM_{2.5} has little effect on hours worked or decision to work. They also note that PM_{2.5} begins to affect productivity at levels well below current US air quality standards.

It would be interesting to assess if similar effects are observed with chemicals associated with consumer products or products used in workplaces. Were one to observe such reduced productivity, even in the absence of clinical symptoms, the value of the loss of productivity should be added to the damages of the chemicals (and hence to the benefits of regulating them).

D. Developmental effects, low birth weights, and other infant and child outcome

The simple model of equations (1)-(3) does not lend itself to situations where pollution interferes with the accumulation of human capital, and thus affects future labour supply and the wage rate.

Exposure to certain chemicals may cause i) infant mortality directly, ii) low birth weight, which in turn may cause infant mortality and illness, and has been linked with lower IQ, school absences, worse educational attainments, and lower earnings (see Black et al., 2007, for a survey of the literature on this link and for an original and convincing study based on Norwegian twins), iii) neurodevelopment problems, iv) congenital malformations and v) other physical impairments.

Exposure to lead and (methyl) mercury in utero and/or as a child has been linked with neurotoxic effects such as hyperactivity, behavioural and attention difficulties, delayed mental development and motor and perceptual skill deficiencies (US EPA, 1997; Spadaro and Rabl, 2008). What is the loss to society associated with these effects?

Landrigan et al. (2002) and Grosse et al. (2002) use labour market data to estimate the wage losses associated with lower IQ, and compute the total losses to society from a change in IQ attributable to lead exposure. Landrigan et al. begin with asking experts to estimate the percentage of cases of certain pediatric

illnesses (lead poisoning, pediatric asthma, childhood cancer, and neurobehavioural disorders) that is attributable to environmental exposures. All cases of lead poisoning are attributed to environmental exposures, and so for lead this share is 100%. Using a sample of 5-year-olds from the National Health and Nutrition Examination Survey, Landrigan et al. determine that the average blood lead level is 2.7 µg/dL, which is predicted to reduce IQ by 0.675 points (1 µg/dL = 0.25 IQ points lost). Earlier estimates by Salkever suggest that 1 IQ point lost results in a 2.39% loss in lifetime earnings, so a 0.675 IQ point loss is estimated to cause a 1.61% reduction in lifetime earnings. If lifetime earnings are USD 881 027 for boys and USD 519 631 for girls, then based on the population of 5-year-old boys and girls, the total losses are USD 27.8 billion and USD 15.6 billion (USD 43.4 billion altogether). The cost per IQ point is thus USD 21 014 for boys and USD 12 394 for girls.

Amendments to this approach include adding the cost of treatment, and special instruction required by subjects with such developmental difficulties. Again, this approach is *ex post*, and fails to include the disutility and suffering of parents and children. Moreover, it does not capture the utility that the rest of society places on avoiding a loss in IQ in a more or less sizable portion of its members. Another difficulty is that the Landrigan et al. (2002) and Grosse et al. (2002) estimates are based on exposure from earlier years and on older data from US labour markets. Productivity has been increasing over time among US workers, and so it is unclear how these estimates should be extrapolated into the future.

Other estimates of the wage and other losses per IQ point include Muir and Zegarac (2001), who report USD 14 700 per IQ point (1999 USD); Grosse et al. (2002) (USD 14 500 per IQ point, 2000 USD); Rice and Hammitt (USD 16 500 per IQ point, 2000 USD), Trasande et al. (2005) (USD 22 000, 2000 USD); Griffiths et al. (2007) (USD 11 245, 2000 USD). Rabl et al. average these figures to USD 18 000 per IQ point (2005 USD), but feel that it would be best to compute the loss of GDP per capita for each IQ point, rather than the loss of private wages, since GDP per capita is a broader measure of productivity that also captures firms' profits, returns to investments, etc. They produce a worldwide measure of USD 3 890 for each IQ point.

It is of course possible to apply stated preference methods to find out how much parents (or prospective parents) or society as a whole are prepared to pay to avoid the cognitive impairments associated with exposure to certain chemicals. The effects are generally expressed in terms of shifts of the distribution of IQ in the population, which poses difficulties when trying to create scenarios for respondents to value in stated preference surveys.

Focusing on PCB exposures and associated health endpoints, Von Stackelberg and Hammitt (2009) use contingent valuation, time trade-offs and standard gamble methods to elicit the WTP for an IQ point, which is then converted into QALY weights.⁹ Rice et al. (2010) focus on two endpoints associated with

9. The quality-adjusted life year or quality-adjusted life-year (QALY) is a generic measure of disease burden that includes both the quality and the quantity of life lived. It is used to assess the cost-effectiveness of medical interventions, and thus to allocate limited resources to different medical procedures. One QALY equates to one year in perfect health. If an individual's health is below this maximum, QALYs are accrued at a rate of less than 1 per year. To be dead is associated with 0 QALYs, and in some circumstances it is possible to accrue negative QALYs to reflect health states deemed "worse than dead." The exact weight between 0 and 1 to be attached to a health state is computed from surveys of individuals using a variety of approaches, including "time trade-offs" (e.g., what would you choose, seven months in perfect health or one year with a specified less-than-perfect state health?), standard gambles (choose between staying in a given impaired health state or being restored to perfect health through an intervention that has chance (1-p) of being successful and chance p of being fatal), and visual analogue scales, where the respondent indicates the current health state on a scale from 0 to 100. The notion of QALY is not very helpful for cost-benefit analysis, because it rests on very restrictive assumptions about individual preferences (see Hammitt, 2002).

methyl mercury intake in the US – reduction in IQ and risk of a heart attack – and develop joint distribution of health endpoints and values.

One way to value outcomes ii)-v) in the list developed at the beginning of this section is to look at the reduced schooling attainment often experienced by children and adolescents affected by these problems, which in turn results in lower lifetime earnings. School absences are one way of capturing some of these effects. They impose costs on society in terms of remedial programs, future lower wages due to worse educational attainment, lost wages and reduced productivity for parents (Currie et al., 2009).

Currie et al. (2009), for example, look at school absences associated with fluctuations in air pollution in Texas, and, using the estimated elasticity between birth weight and earnings of 0.1 from Black et al. (2007), the average earnings of full-time workers in 2003, and the total number of births in 2003, conclude that a decline in carbon monoxide concentrations of the magnitude observed in Texas from 1989 to 2003 resulted in a nationwide increase in earnings of USD 720 million for the 2003 birth cohort. This calculation assumes participation in the labour force for 30 years and a discount rate of 6%, and almost certainly understates the true increase in earnings, since future earnings of this cohort are likely to be higher than current earnings.

One important question is how to attach a monetary value to low birth weight outcomes. One approach (recently deployed in Ščasný and Zverinova, 2014) is to ask prospective parents how much they would be willing to pay to reduce the risk of a low birth weight new-born. This is part of a broader study where Ščasný and Zverinova ask respondents to provide information about their WTP for a healthy baby, thus reducing the chance of i) minor birth defects, ii) defects in internal organs, iii) defects in external body parts, and iii) very low birth weight. Based on a conservative approach, they estimate the Value of a Healthy Baby for EU 28 countries to be EUR 13 000 in case i), EUR 216 000 in case ii), EUR 151 000 in case iii) and EUR 132 000 in case iv). These are figures based on a private good scenario. Their public good scenario counterparts are EUR 51 000, EUR 792 000, EUR 529 000, and EUR 644 000, respectively. (In the public good scenario, people report information about their WTP for risk reductions that they could experience themselves or by others, especially if the respondent does not plan on having children or is no longer in reproductive age.)

Alternatively, using a cost-of-illness approach, it is possible to quantify the medical expenses associated with treating illnesses and health problems attributable to a low birth weight, the additional education expenditures associated with addressing poor attendance and frequent absences from school due to illnesses attributable to a low birth weight, and the lower earnings associated with worse educational attainment (Trasande, 2016). One problem with the latter approach, however, is that it misses the suffering experience by children and the stress, anxiety and suffering of parents.

Turning to infant mortality (item i) in the list developed at the beginning of this section), it is not clear what metric should be used to monetise infant mortality effects. Rabl et al. (2014, page 383) are generally proponents of using the Value of A Statistical Life Year (VOLY) to monetise the mortality benefits of air pollution reductions (or the mortality costs of inaction with respect to poor air quality), but in the case of infant mortality they conclude that it is preferable to use the Value of a Statistical Life (VSL) because i) there are no estimates of life expectancy loss associated with infant mortality (especially if the infant is frail), and ii) there are no estimates of the VOLY for children.

There are hardly any credible estimates of the VSL for infants either, although studies (summarised in OECD, 2008) have elicited child VSL using a variety of approaches (ranging from stated preference studies to consumer purchases of safety equipment, such as bicycle helmets) and with mixed results, in the sense that some have found child VSL to be greater than adult VSL (see Alberini and Ščasný, 2011), others have found it to be lower, and others yet have concluded that there are no statistically significant

differences between the two. Currie et al. (2009) use the adult VSL estimates from the US EPA to compute the reduced infant mortality benefits from reducing carbon monoxide levels.

E. Reproductive effects

Many chemicals have been linked, at least tentatively, with fertility and reproductive effects. In principle, it may be possible to use government-collected individual data or develop a new study to document the expenses associated with fertility treatment, assisted reproduction, and pre-natal care. But these represent ex post expenditures that are not appropriate for cost-benefit analysis and fail to capture the disutility and suffering associated with impaired reproduction.¹⁰ The difference between the ex ante and the ex post values is well illustrated in a stated preference study by Neumann and Johannesson (1994), who asked respondents how much they would be willing to pay for in-vitro fertilisation (IVF) under various conception probability scenarios (but knowing that they cannot conceive unless assisted). WTP for IVF increased with the probability of conception, and was different from the WTP from the ex ante perspective (the WTP for IVF insurance, assuming that respondents do not know yet their fertility status). In particular, the WTP per “statistical baby” was much higher in the ex ante case than in the ex post case.

Ryan (1999) did a conjoint choice experiment to estimate the marginal WTP for attributes that were identified as determinants of the utility of undergoing IVF. His analysis focuses on different in WTP across low and higher income individuals.

Gardino et al. (2010) estimates the relative economic value of fertility by eliciting the WTP for the ovarian tissue cryopreservation procedure v. treatment for other health problems, and for a variety of goods and services. Gardino et al. (2010) found that 1) individuals valued very highly the possibility of being able to bear children, independently of the probability that this will actually occur, suggesting a high “option value” for fertility, and 2) the WTP for the fertility procedure was in line with the probability of success of the fertility procedure. One possible limitation of the Gardino et al. study, however, is that the sample was comprised exclusively of 18-25 year-olds. In other words, these persons may have had limited income and responsibility for financial decisions. It is recognised that. At the same time, these persons may be familiar with, and care for, issues of reproductive health and fertility.

For policy purposes, the theoretically correct values are those for ex ante changes in the probabilities of conception, carrying the pregnancy to term (avoiding miscarriage) and delivery a “normal” child.¹¹ The appropriate methods, it seems, are stated preference methods, or revealed preference methods based on the observed trade-offs between money and behaviours, products or interventions that protect *future* fertility. It is also important to determine how much society as a whole, not just potentially affected individuals, care about protecting fertility.

Ščasný and Zverinova (2014; 2016) present respondents from four EU countries with discrete choice experiments offering increases in the probability of conception at a given monthly cost. In split samples, the delivery modes are vitamins (in which case the improved fertility is a private good) or chemical-free products (in which the improved fertility is a public good). The Value of a Statistical Pregnancy (VSP) is approximately EUR 40 000 in the private good scenario, EUR 33 000 in the public good scenarios and with the general public, and EUR 40 000 in the public good scenario but the sample is restricted to persons

-
10. These are ex post expenditures based on the behaviours of persons who know their fertility status. But risk assessments used for benefit-cost analysis purposes produce at best predictions of cases of impaired fertility, and do not identify the persons who will be experiencing such effects, which in turn implies that all analyses must be done from an ex ante perspective.
 11. If the risk assessment procedures are capable of computing the risks of different degrees of impairment from a “normal” baby, then benefit-cost analysis should try to find the WTP to avoid each such outcome.

who wish to have a child. All VSPs decline considerably, when the econometric analysis controls for co-benefits the respondent admits having thought of (for example improved health and fitness for himself or herself), – to about EUR 25 000, EUR 11 000 and EUR 20 000, respectively.

F. Mortality effects

Some chemicals are thought to cause premature mortality. Since mortality benefits often account for the lion's share of an environmental policy's benefits (see US EPA, 1998, 1999; ExternE, 1999; Clean Air for Europe, 2005), in this section of the report, the focus is on mortality benefits, their estimation, factors that potentially affect monetised values, and subpopulations of interest.

The discussion assumes that the damage function approach is followed, which first estimates the reduction in mortality risks (or increase in life expectancy) attributable to a proposed policy. This is usually accomplished by reviewing evidence in the epidemiological or clinical medical literature, or through expert assessment. In the second step, the risk changes (or the increase in an individual's life expectancy) are aggregated over the population of beneficiaries of the policy, and then multiplied by the (economic) value of each such unit.

This section focuses on the value of each unit that makes up the monetised mortality benefits of economic policies. As explained in detail below, the key metric used in the literature and in policy practice for this purpose is the Value of a Statistical Life (VSL).¹² An alternate metric used in policy analyses, namely the Value of a Statistical Life Year (VOLY), is also discussed.

The Value of a Statistical Life

First the VSL is derived in a simple static, one-period model, where individuals maximise the expected utility of consumption:

$$(12) \quad E(U) = (1 - R) \cdot U(y) + R \cdot V(y)$$

In equation (12), $U(\cdot)$ and $V(\cdot)$ denote the state-dependent utilities of consumption when alive and dead, respectively, and R denotes the baseline chance of dying in that period.

The VSL is the willingness to pay for a marginal reduction in the risk of dying, and is therefore defined as the rate at which the people are prepared to trade off income for risk reduction:

$$(13) \quad VSL = \frac{\partial WTP}{\partial R}.$$

It is straightforward to show that

$$(14) \quad VSL = \frac{U(y) - V(y)}{(1 - R) \cdot U'(y) + R \cdot V'(y)},$$

where the numerator is the utility differential between the two states, and the denominator is the expected marginal utility of income. If the utility of income is assumed to be zero when dead, the expression for the VSL is simplified to

12. Definitions and some of the materials covered in this section can also be found in OECD (2012).

$$(15) \quad VSL = \frac{U(y)}{(1-R) \cdot U'(y)}.$$

In policy analyses, the VSL is regarded as a summary measure of the WTP for a mortality risk reduction, and a key input into the calculation of the benefits of policies that save lives. The mortality benefits of the policy are computed as $VSL \times L$, where L is the expected number of lives saved (i.e., not lost) by regulating the chemical.

The concept of VSL is generally deemed as the appropriate construct for ex ante policy analyses, and/or when the identities of the people whose lives are saved by the policy are not known yet.¹³ As shown in the above mentioned example, in practice VSL is computed by first estimating WTP for a specified risk reduction ΔR , and then by dividing WTP by ΔR .

How is the VSL estimated?

The “traditional” approach holds that the value of a life is the present value of the stream of income generated by a person over his or her remaining lifetime. This approach has been criticised because it places a very low value, or no value at all, on individuals that are not gainfully employed, such as homemakers or retired persons, even if these people *are* willing to pay for a reduction in their own risk of dying.

Alternative estimates of the VSL can be derived from compensating wage differentials in labour markets. The rationale of compensating wage studies is that workers must be offered higher wages for them to accept jobs with greater risks of dying, and that employers are willing to do so to the extent that this is cheaper than installing safety equipment in the workplace. This approach too misses the value placed on improved safety by people that are not in the labour force, such as home-markers, retirees and children (Cropper et al., 2011).

In a typical compensating wage study, data are gathered on the wage rate, education, experience, occupation, and other individual characteristics of workers and workplace characteristics. These data are then used to run a regression relating the wage rate to the risks of fatal and non-fatal injuries, while controlling for education and experience of the worker, and other job and worker characteristics thought to influence wages. Viscusi (1993) argues that the correct specification of the wage regression is:

$$(16) \quad w_i = \beta_0 + \mathbf{x}_i \boldsymbol{\beta}_1 + p_i \beta_2 + q_i \beta_3 + (q_i \times WC_i) \beta_4 + \varepsilon_i$$

where w is worker i 's wage rate (or its log transformation), and $\mathbf{x}$ is a vector of individual, workplace and occupational characteristics, such as experience, education, age, gender, marital status, union status of the worker, industry dummies, occupation dummies, and geographical dummies. The variable p measures the risk of dying on the job, while q is the risk of non-fatal injuries. The β s are unknown coefficients to be estimated, and the VSL can be inferred from β_1 . Viscusi (1993) recommends that equation (16) include expected worker compensation in the event of a non-fatal workplace accident, i.e., $(WC \times q)$, where WC is the level of workers compensation paid out to the worker if he experience an accident at work.

Based on their survey of the US literature, Viscusi and Aldy (2003) recommend VSL figures of USD 5-9 million (2000 US dollars). Estimates of the VSL based on compensating wage studies are available for

13. The concept is also used in retrospective benefit-cost analyses, which compute the number of deaths “attributable” to air pollution, for example, but cannot attribute an identified person’s death to air pollution.

several European countries, including the UK, where they usually range between USD 4 and USD 11 million,¹⁴ and Switzerland (USD 6.5 – USD 9.5 million 2000 dollars; Baranzini and Ferro Luzzi, 2001).¹⁵

Estimates of VSL based on compensating wage studies are rife with econometric identification problems and biases (Leigh, 1995; Arabsheibani and Marin, 2000, 2001; Black and Kniesner, 2003, and Black et al., 2003). Kniesner et al. (2006) offer a nice discussion of the many econometric difficulties that afflict wage-risk studies, and propose an approach to filter out some of the measurement errors affecting workplace risks. Hintermann et al. (2010) use a panel dataset documenting wages and work conditions in the UK labour market in recent years, and find no evidence of compensating wages when heterogeneity of worker preferences and endogeneity of workplace risks are accounted for in the econometric analysis. In other words, when more sophisticated econometric procedures were used, higher job risks were not associated with a positive and statistically significant wage premium. Kniesner et al. (2012) estimate the VSL using panel data.

Clearly, when using the VSL estimated from labour market studies, one is implicitly assuming that the trade-offs between the risk of *accidents* and income observed in labour markets can be applied in other contexts, such as workplace risks associated with chemicals exposures or non-worker exposures through consumer products or environmental exposure. This assumption is questionable and untested.

Despite concerns about the econometric identification of compensating wage differentials and the appropriateness of transferring the VSL from labour markets to other safety and environmental policy context, government agencies do rely on VSL figures from wage-risk studies. For example, in its 2010 Guidelines for Economic Analyses, the US EPA relied on 21 compensating wage studies, out of a total of 26 studies, to produce a VSL figure (USD 7.4 million 2006 dollars). It should be noted that these wage-risk studies are based on US labour market data from the 1970s and 1980s, when mining and manufacturing (two industries with comparatively large mortality and injury risks) employed larger shares of the labour force than they do today, and when labour markets were different than nowadays. They also assumed that the labour market is nationwide and that there is limitation to accessing a certain occupation.¹⁶

-
14. In some cases, studies based on UK data have found the VSL to be much larger than the upper bound of this range. For example, Siebert and Wei (1994), Sandy and Elliott (1996), Arabsheibani and Marin (2000) and Sandy (2001) peg the VSL in the range between EUR 4.3 million and EUR 74.4 million (equal to USD 4.0 million to USD 68.5 million at the 2000 exchange rate). A meta-analysis by CSERGE (1999) generates a range of VSL figures between EUR 2.9 million and EUR 100 million, resulting in weighted average equal to EUR 6.5 million (all 2000 EUR, the corresponding dollar amounts being USD 2.7, USD 92.1 and USD 6.0 million).
 15. Using the 1995 Swiss Labour Force Survey (SLFS) and the 1994 Swiss Wage Structure Survey (SWSS), Baranzini and Ferro Luzzi peg the VSL implicit in the choices of Swiss workers in the range of CHF 10 to CHF 15 million (Swiss Francs, equivalent to USD 6.5 to USD 9.5 million 2000 US dollars). They find that the VSL depends on risk level, union coverage, and age.
 16. It also seems likely that the composition of the labour force in industries with relatively high injury risks has changed since those studies took place. A recent study by the Pew Charitable trust (www.pewtrusts.org/~media/assets/2015/12/variation_brief.pdf) shows that immigrant workers are 1.5 times more likely than native-born to be employed in construction and agriculture, and 0.9 times as likely in manufacturing. Labour Department statistics point to the lower educational attainment of foreign-born workers compared to native-born (for example 23.9% of foreign-born workers aged 25 and older has not completed high school (vs. 4.6% among native-born workers), 16.9% has an associate degree or some college (vs. 29.9% for native-born workers). The proportions for foreign-born and native-born persons that had a bachelor's degree and higher were more similar, at 34.9% and 39.1% respectively (www.bls.gov/news.release/forbrn.nr0.htm). Foreign-born workers are also paid less than native-born. This is interesting because the construction industry is indeed that with the highest fatality rate, followed by

An alternative approach to estimating the VSL is to relate the price of a product to the product's attributes, including its safety. Atkinson and Halvorsen (1990) and Andersson (2005) apply this approach with cars (where the risk is the risk of a fatal accident), where they further control for other car characteristics (e.g., horse power, size, etc.) that explain the price of the car.

Consumer behaviour studies examine the time spent by an individual in activities that increase safety, or the amount of money spent on items that reduce risk, to estimate WTP for a reduction in the risk of death. A meta-analysis by Blomquist (2004) concludes that the VSLs from averting behaviour studies are typically smaller than those produced by other approaches.

Ashenfelter and Greenstone (2004) infer the VSL from states' decisions about speed limits. They assume that the states will adopt a speed limit if the costs (the value of the additional travel time) are greater than or equal to the benefits of the speed limits (the value of reduced accident risks). Their estimates imply that the states were willing to accept risks that resulted in a savings of USD 1.54 million (1997 dollars) per fatality. This figure thus represents an upper bound for the VSL implicit in State legislatures' decisions. It should be noted that this figure represents a VSL only to the extent that the State legislatures mirrors the preferences of the residents they wish to protect.

Finally, it is possible to estimate the VSL using stated preference studies. Stated preference studies rely on what people *say* that they would do under well-specified hypothetical circumstances. They do not involve actual money transactions, and they are useful as an alternative (or a supplement to) to revealed preference studies when (i) it is not possible to implement the latter, (ii) there is no guarantee that people's perception of risks matches objective risks, and (iii) there is little or no variation in the risk to be valued.

Stated preference methods have indeed been used to place a value on mortality risk reductions and thus to infer the VSL. Researchers have generally used one of four main research methods: (i) contingent valuation, (ii) conjoint choice experiments, (iii) the chained approach first illustrated in Carthy et al. (1999), and (iv) risk-risk questions, paired with a valuation question (Viscusi et al., 1991). OECD (2012) present a meta-analysis of VSL estimates obtained in various countries using stated preference methods, which generally produce lower VSL figures than those from labour market studies.

The effect of age on the VSL

Some observers suggest that the VSL should be lowered when a policy reduces risks for older persons or persons in compromised health, because they have a shorter remaining life expectancy.¹⁷ However, economic theory shows that this is the case only under very restrictive assumptions.¹⁸

transportation and warehousing, and agriculture, forestry and mining (www.bls.gov/iif/oshwc/foi/cfch0013.pdf). Moreover, Hispanics-Latino account for two-thirds of the fatalities in 2014. In his book *Fast Food Nation* (2001), Eric Schlosser describes the nature of the injuries experienced by workers in the meat packing industry (many of whom are Hispanic and immigrants). Recent statistics further suggest that the nature of the fatal injuries is likely to be different nowadays from what it was decades ago: The Bureau of Labour Statistics and the Occupation Safety and Health Administration report that half of the fatalities are transportation-related.

17. In the transportation safety and environmental policy contexts, it has been noted that deaths occur disproportionately in certain age groups. For example, the majority of the people dying in road traffic accidents are young males, whereas epidemiological evidence from the United States (Pope et al., 1995) indicates that over 75% of the lives saved by the Clean Air Act are those of persons 65 years old and older. Likewise, heat waves have been linked with increased premature mortality among the elderly (see, for example, Basu and Samet, 2002; Currero et al., 2002; Ebi et al., 2004; Median-Ramon et al., 2006;

Empirical studies (reviewed in Krupnick, 2007) suggest the WTP for any given risk reduction declines by a relatively modest amount only among the very elderly, and Alberini and Chiabai (2007) report that persons surveyed in Italy about cardiovascular mortality risks are willing to pay more if they are in poor health. This finding is in sharp contrast with the practice of some government agencies (e.g., in the United States, the Food and Drug Administration) and in medical decision-making of relying on quality-adjusted life years (QALY), whereby survival with a chronic illness would be deemed as less desirable than survival in perfect health. In a meta-analysis of VSL values from stated preference studies, OECD (2012) did not find any impact of age on the VSL.

Viscusi and Aldy (2007) estimate the relationship between VSL and age using labour market data, finding that older workers have greater risk vulnerability and face flatter wage-risk gradients than younger workers, and that the relationship between VSL and age follows an inverted-U shape. Their estimates of the VSL, based on US labour markets, range from US\$ 6.4 million for younger workers to a peak of USD 9 million for workers aged 35-44, and then decline with age, to the point that it is USD 3.8 million for the oldest workers in the sample (persons aged 55-62).

The Value of a Statistical Life Year

Some of the illnesses (e.g., diabetes) linked with exposure to certain chemicals result in a considerable loss of life expectancy. Drake (2016), for example, reports that diabetes implies a loss of life expectancy of 10 years. When effects are expressed in terms of loss of life expectancy, the natural valuation metric when assessing proposed regulations would therefore seem to be the willingness to pay for an extension in remaining life expectancy. When that life expectancy extension is one year, the metric is termed Value of a Statistical Life Year (VOLY).

Unfortunately, it is difficult to reconcile valuation of life-years with economic theory (see Alberini et al., 2015, OECD report). Moreover, there are only very few studies that have specifically inquired about individual preferences for remaining life expectancy and the willingness to pay for an improvement in it. Johannesson et al. (1997), Morris and Hammit (2000), Chilton et al. (2004), and Desaigues et al. (2011) are exceptions to this dearth, but critics (including the author) have raised concerns about the credibility of some of these studies.¹⁹

Finally, the notion of VOLY is used in policy analyses in addition to or instead of that of VSL, but, depending on the age of the people whose lives are saved by the policy, can offer recommendations in conflict with those obtained by using VSL. Consider for example two alternative public programs, and suppose that both save 100 lives. But suppose that with one, the lives saved are those of young adults, whereas the other saves the lives of the elderly. Then, as long as the VOLY is constant with respect to age, the policy that saves young adults, who have a longer life expectancy, would be concluded to offer greater

Median-Ramon and Schwartz, 2007). This has led to the question whether the VSL should be adjusted for age.

18. Alberini et al. (2016) show that the appropriate framework is the life-cycle model, which posits that the individual receives expected utility from income in each period of his or her remaining lifetime. Adjusting the VSL for age or other factors affecting remaining life expectancy relies on two restrictive assumptions: (i) that the utility divided by marginal utility does not vary with age, and (ii) that the discount rate is zero. There is no particular reason to believe that these assumptions should be true in practice. For example, if the marginal utility of consumption increases with age, then it is no longer appropriate to assume that WTP is proportional to remaining life years.
19. The Desaigues et al. study was conducted in nine countries for a total of only 1 463 respondents. Individuals were presented with a graph that clearly mislabels the change in life expectancy and the econometric analysis of the responses is well below acceptable standards.

benefits if the VOLY is used. By contrast, if the VSL is used, and a single figure is applied to people of all ages, the two policies would be concluded to provide the same benefits.

In the absence of direct estimates of the WTP for a life expectancy extension, researchers have derived the VOLY from estimates of the VSL. Specifically, assuming that a VOLY is constant over the rest of one's remaining lifetime, and letting T be the number of expected remaining life years, the VOLY and the VSL are related as follows:

$$(17) \quad VSL = \sum_t^T VOLY \cdot (1 + \delta)^{-t},$$

where δ is an appropriate discount rate. The limitations of this approach should be apparent. The VOLY obtained in this fashion may be sensitive to the choice of the discount rate and is forced to be constant with a person's age. The latter restriction can be overcome if one is willing to make an assumption about the shape of the relationship between the VOLY and the age of the individual. (Again, this will be an untested assumption.)

4. Valuing cancer outcomes

Many chemicals are proven or suspected carcinogens, and risk assessments of proposed chemicals regulations include excess cancer lifetime risks, or the number of cases of cancer expected in the population with and without the regulations.²⁰

Many types of cancer can be fatal, and there has been much debate in academic and policy circles on whether cancer mortality benefits should be based on a different VSL figure than that used for other causes of death. Policy practice varies across countries and government agencies, with the US EPA using the same VSL for all causes of death, and the EU's DG-Environment and the UK's Health Executive applying a 50% cancer "premium."

Economic theory suggests several reasons why the VSL for one cause of death might be different from that for another. First, the VSL should increase with baseline risks (Pratt and Zeckhauser, 1996). All else the same, the VSL for a specific cause of death might be larger simply because the baseline risk of dying for that cause is higher.

Second, the existence of competing risks might be another reason for different VSLs. Eeckhoudt and Hammitt (2001) consider competing risks and show that if the utility of a bequest at death is positive, then the marginal WTP for reducing one type of risk (i.e., the VSL for that cause of death) depends on the magnitude of the other risks of dying. Based on their model, a person in poor health with a high risk of dying from a chronic illness would have a very low WTP for a small reduction in the risk of dying for another cause (e.g., pollution exposures) if the latter accounts for a very small share of that person's total risk of dying (the "why bother" effect). Evans and Smith (2006) show that the effect of a competing risk is potentially ambiguous because it depends on how the competing risks enter in the expected utility.

Another reason why people might be willing to pay different amounts of money to reduce the risk of dying from different causes may simply lie in the timing of the risk reduction. Economic theory shows that the VSL at time t for a risk reduction to be incurred L periods later is equal to the VSL for an immediate risk reduction in period $(t+L)$, discounted back to the present (Cropper and Sussman, 1990). Chemicals

20. See Rabl et al. (2014) for cancer concentration-response functions for a number of chemicals and substances often found in air pollution, including heavy metals, certain volatile organic compounds, polycyclic aromatic hydrocarbons, and formaldehyde (p. 107-122). These dose-response functions are based on the IRIS database of the US Environmental Protection Agency.

regulations would presumably take place immediately, but the risk reductions may take place only a number of years thereafter because of the possible “latency” (also called “cessation lag”) between exposure and onset of cancer. Latency considerations, however, imply that, all else the same, one would be prepared *less* for cancer risk reductions because they are incurred in the future.

These reasons alone could explain why individuals may value cancer risk reductions differently from reductions in the risk of dying from other causes. Other reasons are suggested by the psychometric literature (Fischhoff et al., 1978; Slovic, 1987; Chauvin et al., 2007). Cancer is associated with suffering and pain, and is highly dreaded (see Starr, 1969; Fischhoff et al., 1978; Slovic, 1987), which is often taken to imply that the VSL should be *greater* when the cause of death is cancer (Revesz, 1999; US EPA, 2000).

Evidence from the literature is, however, mixed and inconclusive as to whether the cancer VSL is larger, smaller or about the same as the VSL for other causes of death (Jones-Lee et al., 1985; Mendeloff and Kaplan, 1989; McDaniels et al., 1992; Savage, 1993, and Tolley et al., 1994; Magat et al., 1996; van Houtven et al., 2008; Hammitt and Liu, 2004; Hammitt and Haninger, 2010; Tsuge et al., 2005; Adamowicz et al., 2011; Alberini and Ščasný, 2011).

Some researchers believe that the difference between cancer and other causes of death lies in the morbidity associated with cancer, and have been trying to isolate the “pure mortality” from the “morbidity” component of the cancer VSL.²¹ This distinction is not mere intellectual curiosity, as it is important to avoid double counting, especially when risk assessments of proposed regulations quantify lifetime cancer excess risks, breaking them down into cases of non-fatal and fatal cancers.

When a risk assessment quantifies the cases of cancer associated with a chemical, or the reduction in the number of cases of cancer attributed to the policy, but does not specify how many of them are fatal, then the appropriate metric to use in benefit-cost analyses is the Value of a Statistical Case of Cancer (VSCC), namely the willingness to pay for a marginal change in the risk of getting cancer.

This figure has been estimated, for example, through hedonic housing pricing studies. Gayer et al. (2000, 2002) observe the prices of homes near Superfund sites (sites contaminated by uncontrolled hazardous waste where remedial investigation and clean-up activities are performed by or under order of the US EPA), and combine them with cancer risks predicted using pollutant concentrations, dilution and groundwater models, and resident exposure assumptions. Using regression analyses and assumptions about people’s beliefs about risks, they disentangle the Value of a Statistical Case of Cancer (VSCC) implicit in the value of homes at different distances from the contaminated site. Davis (2004) uses housing price hedonics to estimate the VSCC for paediatric leukaemia. Alternatively, it is possible use stated preference methods, as in Tonin et al. (2008), who value reductions in the risk of *developing* cancer (whether or not one dies from it).

To understand the relationship between the VSCC and the cancer VSL (and hence the appropriate figures to use in benefit-cost analysis), it is useful to understand how much people are prepared to pay to avoid cancer per se, and how much they are prepared to pay to avoid dying from it, once they have developed it in the first place. Alberini and Ščasný (2015) conduct a dichotomous-choice contingent

21. For example, in DeShazo and Cameron (2004), respondents examine alternatives defined by mortality risk reduction and illness and recovery from illness profiles, among other attributes. The results of this study do not seem very helpful for policy purposes, however. For starters, respondents are told that they receive a certain risk reduction, and that they die at age X (e.g., at 63). The fact that people are told at what age they die lacks credibility and goes against the very notion of mortality *risk*. Moreover, the same life expectancy is compatible with a virtually infinite number of instantaneous hazard paths, where hazard is the risk of dying in the next period. Finally, the distribution of illness, severity of illness and recovery period is not known to the policy maker and is irrelevant when estimating the mortality benefits of a policy.

valuation survey in four European countries where they ask people to indicate which they would prefer – the status quo (which implies a given chance of getting cancer, and a given chance of dying from it, assuming that one gets it to begin with) or a situation where, for a cost, either the chance of getting cancer, or that of dying from it (conditionally on getting it), or both, are reduced. People were asked to engage in seven such choice situations, which differed from one another for quality of life and pain experienced if cancer is developed.²²

The responses to these choice questions can be used to estimate two key metrics—the Value of a Statistical Case of Cancer (VSCC), which is the WTP for a small reduction in the chance of getting cancer, and the cancer VSL. To see this, the reader is reminded that the unconditional chance of dying of cancer is the product of the chance of getting cancer times the chance of dying from it, conditional on getting it in the first place. Formally, respondents were to consider a risk-reducing option or program where the cancer mortality risk reduction was:

$$(17) \quad \Delta M = M_0 - M_1,$$

where M_0 is the risk of dying from cancer in the status quo, and M_1 that if the risk reduction takes place ($M_0 > M_1$). ΔM can also be expressed as

$$(18) \quad \Delta M = R_0 \cdot D_0 - R_1 \cdot D_1 = \Delta R \cdot D_0 + R_0 \cdot \Delta D - \Delta R \cdot \Delta D,$$

where R denotes the risk of cancer, D the chance of dying from it, subscripts 0 and 1 the initial and final situation, $\Delta R = R_0 - R_1$ and $\Delta D = D_0 - D_1$.

Expression (18) shows clearly that reducing the risk of getting cancer, or that of dying from it, or both, unequivocally reduces the unconditional cancer mortality risk.²³ The WTP for the scenario with the risk reduction is $\alpha \cdot \Delta M$, where α is the VSL.

The VSCC is thus $\partial WTP / \partial \Delta R = \alpha \cdot (D_0 - \Delta D) = \alpha \cdot D_1$, which means that the higher the chance of surviving cancer, the lower the VSCC, and the WTP for a cancer mortality risk reduction is

$$(19) \quad \alpha \cdot \Delta M = \Delta R \cdot VSCC + \alpha \cdot R_0 \cdot \Delta D.$$

The first term in the right-hand side of (19) should capture the disutility of cancer treatment, suffering, loss of income due to missed work, etc., and the second one should capture the additional disutility of, and suffering from, when treatment fails.

Alberini and Ščasný administer the questionnaire to a sample from Italy, the UK, the Netherlands and the Czech Republic, and estimate the cancer VSL to be EUR 2.144 million. The VSCC is EUR 0.208 million when the probability of surviving cancer is kept unchanged; raising it by five percentage points implies a VSCC of EUR 0.177 million, by 10% (from 0.60 to 0.70 at five years)

22. The risk-reducing commodity described in Alberini and Ščasný is thus much simpler than that in Cameron and DeShazo (2013), who use 1) the reduction in the risk of occurrence, 2) age at death, 3) illness (cancer or other), 4) illness, recovery and recurrence, and 5) severity of symptoms and pain. The willingness to pay per micromort depends systematically on all of these factors. Unfortunately, it is unclear how helpful these results are in policy analyses, as agencies must assess benefits and costs ex ante and without information about the specifics of every single case avoided.

23. The partial derivatives of ΔM with respect to ΔR and ΔD are both positive.

points to a VSCC of EUR 0.147 million, and by 20% a VSCC of EUR 0.093 million. It is interesting that when estimation is restricted to the choice cards which posits only reductions in R but no change in D, the cancer VSL is EUR 5.676 million and the VSCC is EUR 0.551 million. The results suggest that the “death” part plays a much more important role in the cancer VSL than the morbidity per se.

Since Alberini and Ščasný have respondents look at cancer risks only, they cannot establish whether the magnitudes of the illness vs. death components are similar for other illnesses that tend to cause premature mortality (e.g., diabetes; see Drake, 2016). It would be certainly be interesting to pursue this line of inquiry.

It is noteworthy that in the Alberini and Ščasný study, pain and quality of life did not seem to affect the VSCC or VSL.

McDonald et al. (2016) take up the issue of latency (cancer risk reductions are typically latent, once a policy is implemented), which should lower the VSL for cancer, all else the same, versus the cancer premium. They use choice questions that do not involve costs or other monetary attributes, and motivate their analysis with the relationship:

$$(20) \quad C_{t+k} = \frac{R_t(1+x)}{(1+d)^k}$$

where x is the cancer “premium” and d the discount rate. They estimate the former to be 0.43 (i.e., 43%) and the latter to be 7.5% in sample of U.K. respondents. They further experiment with adding a morbidity period immediately before death. The length of such morbidity period is varied for both cancer and road traffic fatalities, so that the researchers can assess how morbidity figures in the ratio between the cancer VSL and the road traffic VSL. They find that, once morbidity is controlled for, there is no difference between the cancer and the road traffic VSL for a given latency period. McDonald et al. conclude that morbidity accounts for the majority of the cancer context premium.

5. Chemicals as emerging pollutants

How should valuation be done when the effects of the chemicals are unknown or only tentative? The answer to this question is not simple, and this section offers some simple points for further reflection and analysis.

First, studies have observed and measured a market response, and estimated the corresponding WTP, even for situations where causation had not been established or the consequences of exposure to certain chemicals are not clear. For example, Davis (2004) conducts a hedonic pricing study at a locale where a cluster of paediatric leukaemia cases took place. These were initially suspected to be associated with contaminated groundwater, but this link was eventually ruled out. Yet, a considerable depreciation in property values was observed, compared to a control group county where no cluster of paediatric cancers existed, which led Davis to estimate the VSCC for paediatric leukaemia to be about USD 5 million (2000 dollars).

Second, the public has consistently demonstrated to be willing to switch to different products that do not contain potentially harmful chemicals, and to be willing to incur costs in doing so. Florax et al. (2005) present a meta-analysis of studies documenting the willingness to pay to avoid pesticides, and studies about the willingness to pay for no-pesticide, organic foods is well documented.

A survey conducted by Industrial Economics for Health Canada (2016) shows that some 50% of the general public in Canada is “very concerned” about chemicals and certain properties exhibited by these

chemicals, such as persistence, bioaccumulation, and their potential effects on air, soil and water. About two thirds of the respondents are very concerned about the effects on human health; such as cancer, cardiovascular illnesses, and developmental and reproductive system problems.

A discrete choice experiment conducted within the same survey finds that people would be willing to pay CAD 49 (2015 CAD) extra a month to switch to products that are not carcinogenic, and CAD 17 - CAD 35 extra a month for products that eliminate each of the other adverse attributes of the products or the effects on soil, air or water.

Third, when different sources announce different assessments of the human health risks associated with certain chemicals, pollutants, or merely dietary habits, people may simply average those risks (perhaps after weighting them according in proportion to the credibility and reputation of the source; see Viscusi 1985, 1997), or, in contrast, may exhibit ambiguity aversion. In general, ambiguity aversion means preference for risks with known probability distributions over risks with unknown probability distributions. All else the same, a “lottery” or risk (a situation where someone earns money with probability p and loses money or earns nothing with probability $(1-p)$) would thus be surrendered for a lower cash amount if there is uncertainty about p than if p is known. Goldberg et al. (2009) summarise hypotheses and empirical evidence about ambiguity aversion. For example, Fox and Tversky (1995) argue that ambiguity aversion is present when subjects evaluate both clear and ambiguous prospects jointly, but diminishes or disappears when the prospects are evaluated in isolation.

Based on this argument, one would expect that subjects attach a higher cash equivalent to risky prospects with unambiguous probabilities in experiments that also contain risky prospects with ambiguous probabilities, but that the cash equivalents would be similar, all else the same, if subjects examine only one type of risky prospects. In practice, empirical evidence is mixed. Chow and Sarin (2001) find that unambiguous prospects are always priced higher than ambiguous ones, whether considered in isolation or within settings that include both ambiguous and unambiguous prospects.

One important question is whether these behaviours carry over the situation where the uncertainty is about the occurrence of an adverse health outcome, rather than financial gains or losses. Goldberg et al. (2009) conduct experiments with treatments specifically designed to test the effect of uncertainty about the probability of pathogens in commercial baby formula. Here, the ambiguous situation is that in which no rate of incidence of pathogenic contamination is presented to the study subjects. The experiment contains both between- and within-subject ambiguous health risks, and elicits the willingness to pay for baby formula that is certified to be pathogens-free. Goldberg et al. find that the WTP is virtually the same across all treatments. We find this result intriguing, because it suggests that the monetised benefits of reducing health risks are not strongly affected by uncertainty about the risks themselves, which simplifies benefit-cost analysis tasks. However, these results should be interpreted with caution, due to the small number of subjects in this study, the fact that WTP under hypothetical circumstances was elicited, and the fact that the scenario completely eliminated risks (whereas many policies can at best be argued to reduce risks, but not completely eliminate them).

Treich (2010) derives the VSL in the presence of ambiguity with respect to the baseline probability of dying (whereas the risk reduction is not ambiguous, meaning that there is no uncertainty about it). He uses a simple static model where the decision maker’s utility function is

$$(21) \quad W = \phi^{-1} \{ E[\phi((1 - \tilde{p})U(w) + \tilde{p}V(w))] \},$$

where $\tilde{p}$ is random variable representing the ambiguity over the baseline mortality risks ($\tilde{p} = p_0 + \varepsilon$), and $\phi(\cdot)$, with $\phi' > 0$ and $\phi'' < 0$ captures ambiguity aversion (much like a utility function that is concave

in income or consumption captures risk aversion). If $\phi(x) = x$, then the individual is ambiguity-neutral and for all practical purposes an expected utility maximiser. Treich shows that if an individual is ambiguity-averse, then introducing ambiguity (i.e., uncertainty about the baseline probability of dying) increases the VSL with respect to its counterpart in the absence of ambiguity. Ambiguity effectively acts much like an increase in baseline risks. How the VSL changes when the ambiguity is increased, however, is not clear. Courbage and Rey (2015) derive conditions under which the VSL increases with ambiguity—for both ambiguity-averse and ambiguity-loving persons.

Treich (2010) further discusses what ambiguity aversion does in the presence of competing risks: The decision-maker's VSL is higher for the risk that has the highest baseline probabilities, a result that, he argues, is consistent with many decisions actually made by policy makers and regulators. He further notes, based on numerical exercises, that the VSL “premium” associated with ambiguity for an ambiguity-averse person is likely to be modest at best, and concludes that this is in contrast with the sometimes high VSLs adopted by government agencies. He also shows that the theory cannot conclusively show whether ambiguity aversion increases or decreases the VSL, relative to ambiguity neutrality, when self-protection is present.

These results in Treich (2010) and Courbage and Rey (2015) may help explain the wide range of VSL values sometimes observed across studies.

As noted above, ambiguity may arise in situations where announcements by authorities conflict with one another, or where the authorities state that consuming a product is both beneficial and potentially dangerous. An example of the latter situation occurred when in 2001 the Food and Drug Administration warned the public about the risks of ingesting methylmercury through the consumption of fish, while at the same time explaining that omega-3 fatty acids contained in fish are healthful. Shimshack and Ward (2010) use supermarket scanner data and information about the methylmercury and omega-3 fatty acids contained in different species of fish to study how the public reacted to such announcements. They devise a difference-in-difference study design, where the control group is represented by households without pregnant women or small children, and use data from before and after the announcement. Using quantile regressions to examine differential impacts depending on baseline fish consumption levels, they find that the announcement did reduce methylmercury *and* omega-3 fatty intake, resulting in a welfare *loss* of USD 30 (2007 USD) per at-risk household.²⁴

6. An Example: The US EPA NESHP emissions rule

In the United States, chemical substances are to be regulated under the authority of different statutes and by different agencies. For example, certain chemical substances are regulated by the Toxic Substances Control Act (1976), which is generally interpreted as requiring agencies to conduct benefit-cost analyses. The US EPA, for example, is in charge of pesticide registration, re-registration and cancellation. In registering or denying (re-) registration to existing pesticides, the EPA examines the health risks to persons that are exposed to pesticides on a professional basis (mixers and applicators), consumers (who ingest pesticide residues in fruits and produce), and possible consequences on ecological systems. Risk assessments typically contain estimated excess lifetime risks on the basis of certain exposure assumptions. Cropper et al. (1992) examine a number of pesticide re-registrations and cancellation decisions by the EPA in the 1980s, concluding that the agency was placing a heavier weight on applicators' risks than on

24. In a related article that focuses on canned tuna purchases documented in the Consumer Expenditure Survey, Shimshack et al. (2007) find that the announcement did not have an effect on consumers at risk with low levels of education and newspaper readership, and that it had spill-over effects, in that consumers not considered at risk likewise reduced their consumption of canned fish.

consumers' or mixers risks.²⁵ The EPA's decisions based on 242 regulatory decisions implied a VSL of USD 35 million (1986 USD) for applicators, but only USD 60 000 for consumers.

Some substances are covered by other statutes which define them as hazardous wastes. Hazardous waste disposal, treatment and recycling is heavily regulated under the Resource Conservation and Recovery Act (1976), and by Superfund (Comprehensive Environmental Response, Compensation and Liability Act, 1980), which triggers liability when uncontrolled hazardous waste pose a threat to human and environmental health. Both of these statutes require risk assessments to determine whether a corrective action (i.e., clean-up) is necessary. A number of studies have examined how EPA conducts such risk assessments and whether it seeks to protect maximally exposed individuals or examines population-weighted exposure (see, for example, Gupta et al., 1996).

The remainder of this section describes the NESHAP rule established by the EPA in 2012. The rule concerns national emissions standards for hazardous air pollutants and covers fossil-fuel fired power plants, and industrial-commercial-institutional and small industrial-commercial-institutional steam generating units. The authority to develop and issues this rule comes from section 112(b) of the Clean Air Act.

2. In the Federal Register 77, n. 32, of 16 February 2012, the EPA first identifies the entities subject to the rules. It then proceeds to identify benefits and costs of the rule. The rule is expected to reduce emissions of hazardous air pollutants, including mercury (Hg), from the electric power industry. The technology necessary to meet this rule will also produce co-benefits: It will reduce the emissions of PM_{2.5} and SO₂, which is a precursor of PM_{2.5}. Specifically, the EPA estimated that the rule would reduce emissions of mercury in coal-fired plants by 90%, reduce acid gas emissions from power plants by 88%, and abate SO₂ emissions by 41% above and beyond the reductions expected from other existing legislation.

The EPA identifies the benefits of such emissions reductions as displayed in Tables 2 and 3.²⁶

The majority of the benefits are in reality co-benefits, and they come from reducing premature mortality risks associated with exposure to PM_{2.5}. These account for some 90% of the total benefits. The US EPA acknowledges a relatively wide range for these benefits, due to the assumption that all particulate matter is equally potent in causing mortality. Non-fatal illnesses avoided because of lower PM_{2.5} levels include heart attacks, asthma attacks, and cases of chronic bronchitis, plus over three million restricted activity days.

The US EPA also quantified the risks associated with deposition of mercury, and its subsequent conversion into methylmercury, in bodies of water. Methylmercury is then accumulated in fish. Attention was restricted to freshwater bodies, where mercury from power plants accounts for a relatively large share of the total stock of mercury, especially in the Ohio River Valley. The pathway of concern was consumption of freshwater fish by women of childbearing age because, depending on the level of prenatal exposure, children may experience a range of neurodevelopmental effects, including decrements on a number of neuropsychological measure. The EPA focused on populations that practice subsistence fishing because of their higher consumption rates and exposure to high doses of methylmercury than recreational

25. Cropper et al. fit a probit model that includes the excess lifetime cancer risks of maximally exposed persons in each category, producers' benefits, yield loss in the event of cancellation, and dummies for whether EPA's assessments examined effects on reproductive health and marine ecological systems. Political economy factors such as that academics, growers, and environmental organizations commented on the proposed cancellation, and a dummy for an EPA administrator subsequently criticized for promoting the industry's interests were also included in certain specifications of the model.

26. Regulated entities were given four years to reach compliance. Hence the benefits and costs summarized in Table 3 refer to 2016, the first year when the rule would be fully applied.

anglers. The risk assessment procedure began with modeling mercury dispersion and deposition using EPA's Community Multi-scale Air Quality (QMAC) model, which produced maps of the United States and mercury deposition throughout the country. Populations were then selected that engage in subsistence fishing (e.g., low-income persons, Great Lakes tribal fishers, Laotians and Vietnamese communities nationwide), and assumptions were made about fish (and the methylmercury therein) consumption rates, taking cooking and fish preparation into account.²⁷

Specifically, the annual average daily exposure per kg of body weight was computed as the fish Hg concentration (FC),²⁸ times the Hg conversion factor (MCF), times the food preparation factor (FPCAF), times the fish consumption rate, and then divided by body weight in kg (BW):

$$(22) \quad IR = \frac{FC \times MCF \times FPCAF \times FCR}{BW}$$

This is compared with the reference dose, which is 0.0001 m³/kg-day and identifies a level of exposure above which there is the risk of adverse health effects, and is then converted into a hazard quotient (HQ) for each watershed, due to concerns that the shifts in IQ alone may underestimate the full range of effects of methylmercury. A hazard quotient of 1.5 or greater is regarded as potential harmful, while one below 1.5 is considered not to exceed the reference dose.²⁹ The EPA then seeks to apportion the power generating units contribution to Hg and MeHg, and hence the reductions in HQ attributable to the rule. The EPA also quantifies the gains in IQ points attributable to the rule, but heeded the agency's Scientific Advisory Board that IQ is not especially sensitive to methylmercury and hence tends to understate this substance's true risks.

The benefits computed by the EPA also include the effect of lower levels of emissions of greenhouse gases, which contribute to climate change. These effects were valued using the estimated social cost of carbon.

The EPA also recognised, but did not attempt to quantify, other benefits from the technology that would reduce air toxics—namely, improvements in visibility in national parks and other class I areas, other health effects of mercury exposures, health effects of ozone and direct exposure to SO₂ and NO₂, plus ecosystem effects, health effects from consumption of fish from commercial operations or non-freshwater bodies, and other health risks from exposure to air toxics other than mercury.

-
27. The EPA relied on three studies that measured methylmercury in cooked fish. Another difficulty the Agency encountered while completing the risk assessment is that fish mercury concentration data are available only for a small percentage of all waterbodies. Computing the risks associated with eating fish with high methylmercury content is challenging, because fish also contains beneficial nutrients (e.g., omega fatty acids) that help reduce health risks.
28. The EPA used the 75th percentile of the distribution of Hg concentrations in fish tissue in an effort to select a concentration that would be representative of larger edible fish for watersheds with one or more fish concentration values.
29. Rabl et al. (2014) write that Axelrod et al. (2007) report a loss of 0.18 IQ points per part per million (ppm) of mercury in maternal hair (p. 115).

Table 2. Adverse human health effects avoided each year by the rule

Item	Number of cases
Premature deaths	Up to 11 000
Chronic bronchitis	2 800
Heart attacks	4 200
Asthma attacks	230 000
Hospital and emergency room visits	5 700
Restricted activity days	3 200 000

Source: (Source: US EPA, 2012)

Table 3. Annual Benefits and Costs of the final rule in 2016 (2007 USD)

Category of benefits or costs	3% discount rate	7% discount rate
Total monetised benefits	USD 37 to USD 90 billion	USD 33 to USD 61 billion
Partial Hg-related benefits (consumption of fish and its effects through maternal exposure)	USD 0.004 to USD 0.006 billion	USD 0.0005 to USD 0.001 billion
PM _{2.5} -related co-benefits (mortality, non-fatal illnesses, hospitalisations, restricted activity days)	USD 36 to USD 89 billion	USD 33 to USD 80 billion
Climate related co-benefits	USD 0.36 billion	USD 0.36 billion
Total social costs	USD 9.6 billion	USD 9.6 billion
Net benefits	USD 27 to USD 100 billion	USD 24 to USD 71 billion
Non-monetised benefits	Visibility in class I areas; other neurological effects of Hg exposure; other health effects of Hg exposure; health effects of ozone and direct exposure to SO ₂ and NO ₂ ; ecosystem effects; health effects from commercial and non-freshwater fish consumption; health risks from exposure to non-Hg hazardous air pollutants.	

The benefits must, of course, be compared with the costs of the rule. The costs include all of the resources that must be invested to comply with the rule, plus the value of lost output. If only one or few markets were affected by the rule, then a partial equilibrium analysis would be deemed sufficient to capture most of these costs, which would be equal to the sum of the surplus lost to producers and consumers. Figure 2 displays the inverse demand and supply function for a good in a competitive market. The demand function is, of course, decreasing in the market price, while the supply function is the same as the marginal cost of producing the good and is increasing in price. The market equilibrium prior to introducing an environmental regulation is given by quantity Q_0 and price P_0 . Introducing the regulation means that the firms are now incurring abatement costs. This will shift the supply function inwards, and result in a new equilibrium characterised by less output being traded (Q_1) and a higher price (P_1). The new consumer surplus is the area of the green triangle—clearly much less than the pre-regulation surplus. The firms' abatement costs are equal to the area of the white rhomboid, and an additional source of costs is the black triangle—the deadweight loss—due the loss of output.

Figure 2. Competitive market after the regulation

Source: US EPA, Guidelines to Economic Analyses (2012).

But since the mercury and air toxics rule (MATS) affects power generators, its potential effects may be felt economy-wide, which calls for a general equilibrium type of analyses. The EPA estimated that the annual compliance costs would be 2.7 – 3.5% of annual electricity sales (based on 2000 – 2011), and that the annual compliance capital expenditures would be 3.0 – 5.9% of total annual capital expenditure over a ten-year period. The expected impact on retail electricity prices was estimated to be about USD 0.003/kWh, or 3% of price, which is well within annual fluctuations in electricity prices.

This latter consideration, taken together with the fact that electricity is only a small factor in the production of goods and services, led the EPA to conclude that any downstream economic effects of the rule would be small. As is often done with general equilibrium type of analyses, employment effects were also examined, and the EPA concluded that, if anything, the rule would require 46 000 job-years of one-time construction labour (where a job-year is the equivalent of one person employed for one year). This figure would include jobs in the steel and cement industries, and in the pollution abatement fabrication and installation industries, and would more than offset the lost jobs at power plants (about 15 000 job-year), for a net gain of some 30 000 job-years.

The EPA rule was struck down by the Supreme Court in 2015 (see www.wsj.com/articles/high-court-strikes-down-epa-limits-on-mercury-emissions-1435590069), although this did not technically prohibit the agency from limiting mercury emissions from power plants and did not repeal the mercury rules, leaving a lower court to decide how the case should proceed.³⁰ That lower court ruled in late 2015 the EPA could use its powers to enforce the *Clean Air Act* rules, which then prompted the latest challenge from the collection of states and business interests. The Supreme Court in 2016 withdrew its support of the states that had filed against the EPA.

7. Discussion and conclusions

This paper has presented and discussed the main categories of social benefits associated with regulation of chemicals, the social costs of regulations, and their estimation. Benefits can be estimated with market methods if the chemicals, once dispersed into the environment or contained in products, affect the production of goods and the productivity of labour. When the chemicals have adverse effects on ecological

30. Masur and Posner (2015) discuss the role of unquantified benefits in this decision and others.

systems or human health, it may become necessary to use non-market valuation methods. These methods may rely on visitation rates to natural resources that have recreational use (travel cost method), on the prices of other goods (e.g., housing, cars, foods and products) when these prices change in the presence of the chemicals or the ecological and human health risks associated with them (hedonics), and/or on the expenses sustained by individuals to mitigate the health symptoms associated with chemicals exposure or prevent them in the first place.

It is also possible to ask people to report information about their willingness to pay for a specified improvement in environmental quality or reduction in health risks through surveys of individuals. These survey-based approaches do not observe actual market transactions and merely record what people say that they would do under well-specified hypothetical circumstances. For this reason, they are flexible, suited to measure the WTP for changes that have not been observed in real life yet and capable of capturing non-use values of environmental quality and natural resources, i.e., those experienced by persons who do not use the resources and are not planning to do so in the future, but care to preserve it in its own right and for future generations to enjoy. It is possible to deploy more than one method to value the same health risk or ecological system change. If the methods are deployed correctly, one would expect the results from one to confirm those from the others (or to exhibit the relationship between suggested by economic theory). Different methods may also be deployed in a complementary fashion – the travel cost method, for example, focuses on current or prospective visits, whereas stated preference methods would capture the benefits experienced by a broader population. (Stated preference methods may, of course, also be applied with current visitors to measure their welfare change if the site conditions were to change.) It is important, however, to avoid double-counting of benefits.

Implicit in much valuation work is the assumption that people care about the end outcomes –health risks or ecological system conditions – regardless of what the source of the impacts, the specific nature of the chemical, and the pathway exposure are. But these assumptions need to be tested, and relatively little work seems to have been done in this area. It is recommended that research be done to see if the WTP and benefits depend on these factors. Psychometric research suggests that perceptions of risks might be affected by them, but it is not clear that in practice the WTP to reduce these risks is appreciably changed by these perceptions.

A considerable degree of “transferring” benefits takes place within research and policy circles –across locales, populations and subpopulations, and types of chemicals (see Navrud, 2016). But the dose-response relationships used by government agencies to predict the effect of chemicals, and hence the expected physical impacts of regulating them, are subject to a similar extent of transfer, with effects of a chemical studied for one population (e.g., smokers), chemical (e.g., arsenic) and pathway (e.g., inhalation) extrapolated to predict others (the effect on bladder cancer from arsenic in drinking water). This means that the physical effects are predicted with a considerable amount of uncertainty.

Nevertheless, the public is often prepared to pay to reduce health or ecological risks even when they are highly uncertain. There is ample evidence of this from a variety of settings, and this report’s interpretation of the existing research about ambiguity aversion in the context of health risks is that ambiguity aversion is unlikely to have a major effect on the willingness to pay to avoid or reduce these risks, and on the associated benefit-cost analyses. It is suspected that the human health benefits are likely to account for the majority of the benefits of regulating chemicals, and point that there are many unresolved issues in the valuation of mortality risks, gains and losses in remaining expected lifetime, and cancer. It is also noted that efforts at valuing morbidity would be much simplified in future policy analyses if systematic research was done about the relationship between the WTP to avoid illness and the cost-of-illness figures for that illness (which are often available from past research and administrative data). Presumably the ratio between these entities, which should be greater than one, would vary with the illness

itself and its effects on quality of life, but at present only a handful of studies have attempted to estimate such ratio.

It is also recommended that attention be paid to studying the labour productivity impacts of exposure to chemicals, should they exist even when workers experience only subclinical symptoms and do not miss work – they just make it less productive.

Finally, this report presented briefly procedures typically used in assessing risks and estimating the costs of regulations, using the recent mercury rule by the US EPA (2012) as an example.

REFERENCES

- Adamowicz, W., D. Dupont, A. Krupnick and J. Zhang (2011), “Valuation of cancer and microbial disease risk reductions in municipal drinking water: An analysis of risk context using multiple valuation methods”, *Journal of Environmental Economics and Management*, 61 (2), 213-226.
- Alberini, A. and M. Ščasný (2016), “The Benefits of Avoiding Cancer (or Dying from Cancer): Evidence from a Four-country Study,” draft paper, University of Maryland, College Park, October.
- Alberini, A. and M. Ščasný (2016), “The Benefits of Avoiding Cancer (or Dying from Cancer): Evidence from a Four-country Study,” presented at the Experts Workshop on Valuing the Health Impacts of Chemicals, European Chemicals Agency, Helsinki, January, available at http://echa.europa.eu/news-and-events/events/event-details/-/journal_content/56_INSTANCE_DR2i/title/expert-workshop-on-valuing-the-health-impacts-of-chemicals.
- Alberini, A. et al. (2016), “Approaches and issues in valuing the costs of inaction of air pollution on human health”, *OECD Environment Working Papers*, No. 108, OECD Publishing, Paris, <http://dx.doi.org/10.1787/5j1ww02k83r0-en>.
- Alberini, A. and M. Ščasný (2011), “Context and the VSL: Evidence from a stated preference study in Italy and the Czech Republic.” *Environmental Resource Economics*. 49, 511–538.
- Alberini, A. and A. Chiabai (2007), “Urban Environmental Health and Sensitive Populations: How Much are the Italians Willing to Pay to Reduce their Risks?” *Regional Science and Urban Economics*, 37(2), 239-258.
- Alberini, A. and A. Krupnick (2000), “Cost-of-Illness and WTP Estimates of the Benefits of Improved Air Quality: Evidence from Taiwan,” *Land Economics*, 76(1), 37-53.
- Andersson, H. (2005), “The Value of Safety as Revealed in the Swedish Car Market: An Application of the Hedonic Pricing Approach”, *Journal of Risk and Uncertainty*, 30(3), 211-239.
- Arabsheibani, G.R. and A. Marin (2000), “Stability of the Estimates of the Compensation for Danger,” *Journal of Risk and Uncertainty*, 20(3), 247-269.
- Arabsheibani G.R. and A. Marin (2001), “Self-selectivity bias with a continuous variable: potential pitfall in a common procedure,” *Applied Economics*, 33, 1903-1910.
- Ashenfelter, O. and M. Greenstone (2004), “Using mandated speed limits to measure the value of a statistical life.” *Journal of Political Economy*. 112(S1): S226-S267.
- Atkinson, S.E. and R. Halvorsen (1990), “The Valuation of Risks to Life: Evidence from the Market for Automobiles,” *The Review of Economics and Statistics*, 72(1), 133-136.
- Baranzini A. and G. Ferro Luzzi (2001), “The Economic Value of Risks to Life: Evidence from the Swiss Labour Market,” *Swiss Journal of Economics and Statistics*, 137(2), 149-170.

- Bartik, T.J. (1988), “Evaluating the Benefits of Non-Marginal Reductions in Pollution Using Information on Defensive Expenditures”, *Journal of Environmental Economics and Management*, 15: 111-127.
- Bennett, A. and J.C. Loomis (2015), “Are Housing Prices Pulled Up or Pushed Down by Fracked Oil and Gas Wells? A Hedonic Price Analysis of Weld County, Colorado,” *Society and Natural Resources*, 28(11), 1168-1186.
- Black, S., P. Devereux and K. Salvanes (2007), “From the Cradle to the Labour Market? The Effect of Birth Weight on Adult Outcomes”, *Quarterly Journal of Economics* 122: 409-439.
- Black, D.A., J. Galdo and L. Liqun (2003), “How Robust are Hedonic Wage Estimates of the Price of Risk?”, Final Report to the USEPA, June 2003, R 82943001.
- Black, D.A. and T.J. Kniesner (2003), “On the Measurement of Job Risk in Hedonic Wage Models, *Journal of Risk and Uncertainty*,” 27(3), 205-220.
- Blomquist, G. (1979), “Value of Life Saving: Implications of Consumption Activity,” *Journal of Political Economy*, 96(4), 675-700.
- Blomquist, G. (2004), “Self-Protection and Averting Behaviour, Values of Statistical Lives, and Benefit Cost Analysis of Environmental Policy,” *Review of Economics of the Household*, 2, 89-110.
- Cameron, T.A. and J.R. Deshazo (2013), “Demand For Health Risk Reductions,” *Journal of Environmental Economics and Management*, 65(1), 87–109.
- Carthy, T. et al. (1999), “On the Contingent Valuation of Safety and the Safety of Contingent Valuation: Part II – The CV/SG ‘Chained’ Approach,” *Journal of Risk and Uncertainty*, 17(3), 187-213.
- Chang, T. et al. (2014), “Particulate Pollution and the Productivity of Pear Packers”, NBER Working Paper 19944.
- Chestnut, L. G. et al. (1996), “Measuring heart patients’ willingness to pay for changes in angina symptoms”, *Medical Decision Making*, 16(1), 65-76.
- Chilton, S. et al. (2004), *Valuation of Health Benefits Associated with Reductions in Air Pollution: Final Report*, DEFRA, London.
- Chiu, W. (2016), *Chemical Risk Assessment and Translation to Socio-Economic Assessments*, ENV/EPOC/WPIEEP(2016)9.
- Chow, C.C. and R.K. Sarin (2001), “Comparative Ignorance and the Ellsberg Paradox,” *Journal of Risk and Uncertainty*, 5, 325-370.
- Corso, P.S., J.K. Hammitt and J.D. Graham (2001), “Valuing Mortality-Risk Reductions: Using Visual Aids to Improve the Validity of Contingent Valuation,” *Journal of Risk and Uncertainty*, 23(2), 165-184.
- Courbage, C. and B. Rey (2015), “Value of Statistical Life and Changes in Ambiguity,” draft paper, available at <http://afse2015.sciencesconf.org/61892/document>.
- Cropper, M. and F. Sussman (1990), “Valuing Future Risks to Life”, *Journal of Environmental Economics and Management*, 19, 160-174.

- Cropper, M. et al. (1992), "The Determinants of Pesticide Regulations: A Statistical Analysis of EPA Decisionmaking", *Journal of Political Economy*, 100(1), 175-197.
- Cropper, M. and U. Subramanian (2000), "Public Choices Between Life-saving Programs: The Tradeoff Between Qualitative Factors and Lives Saved," *Journal of Risk and Uncertainty*, 21(1), 117-149.
- Cropper, M., J.K. Hammitt and L. Robinson (2011), "Valuing Mortality Risk Reduction: Progress and Challenges", *Annual Reviews of Resource Economics*, 3, 313-336.
- CSERGE (1999), "*Benefits transfer and the economic valuation of environmental damage in the European Union with special reference to health*", CSERGE (Centre for Social and Economic Research on the Global Environment), IOS-NLH, IVM, CAS and DAE-UoV.
- Currie, J. et al. (2012), "Do Housing Prices Reflect Environmental Health Risks? Evidence from More than 1600 Toxic Plant Openings and Closings", *NBER Working Paper* 18700.
- Currie, J. et al. (2009), "Does Pollution Increase School Absences", *The Review of Economics and Statistics* 91(4): 682-694.
- Davis, L.W. (2004), "The effect of health risk on housing values: Evidence from a cancer cluster", *American Economic Review*, 94 (5), 1693-1704.
- Desaigues, B. et al. (2011), "Economic valuation of air pollution mortality: A 9-country contingent valuation of a life year (VOLY), *Ecological Indicators*, 11, 902-91.
- DeShazo, J.R., and T.A. Cameron (2004), "Mortality and morbidity risk reduction: an empirical life-cycle model of demand with two types of age effects", Unpublished paper, Department of Policy Studies, University of California at Los Angeles.
http://faculty.spa.ucla.edu/deshazo/pdf/14/Age_paper_allparts.pdf.
- Dickie, M. and S. Gerking (1996), "Formation of Risk Beliefs, Joint Production and Willingness to Pay to Avoid Skin Cancer," *Review of Economics and Statistics*, 78(3), 451- 463.
- Drake, L. (2016), "Health Economic Costs of Diabetes and Bone Fractures Caused by Exposure to Chemicals," presented at the Experts Workshop on Valuing the Health Impacts of Chemicals, European Chemicals Agency, Helsinki, January, available at http://echa.europa.eu/news-and-events/events/event-details/-/journal_content/56_INSTANCE_DR2i/title/expert-workshop-on-valuing-the-health-impacts-of-chemicals.
- Eeckhoudt, L.R. and J.K. Hammitt (2001), "Background Risks and the Value of a Statistical Life," *Journal of Risk and Uncertainty*, Vol. 23(3), 261-79.
- Evans, M.F. and V. Kerry Smith (2006), "Do We Really Understand the Age-VSL Relationship?" *Resource and Energy Economics*, 28, 242-261.
- Fischhoff, B. et al. (1978), "How safe is safe enough? A psychometric study of attitude towards technological risks and benefits", *Policy Sciences*, 8, 127-152.
- Florax, R.J.G, M. Chiara Travisi and P. Nijkamp (2005), A Meta-Analysis of the Willingness to Pay for Reductions in Pesticide Risk Exposure, available at <http://purl.umh.edu/19524>.

- Fox, C.R. and A. Tversky (1995), “Ambiguity Aversion and Comparative Ignorance”, *Quarterly Journal of Economics*, 110, 585-603.
- Gardino, S.L., A. Sfekas and D. Dranove (2010), “Anticipating ovarian tissue cryopreservation in the health care marketplace: A willingness to pay assessment”, *Cancer treatment and research*, 156 (5), 363-370.
- Gayer, T., J.T. Hamilton and W. Kip Viscusi (2000), “Private Values of Risk Tradeoffs at Superfund Sites: Housing Market Evidence on Learning About Risks”, *Review of Economics and Statistics*, 82(3), 439–451.
- Gayer, Ted, James T. Hamilton, and W. Kip Viscusi (2002), “The Market Value of Reducing Cancer Risk: Hedonic Housing Prices with Changing Information”, *Southern Economic Journal*, 69(2), 266–289.
- Goldberg, I., J. Roosen and R.M. Nayga, Jr. (2009), “Parental Response to Health Risk Information: Experiment Results on Willingness-to-Pay for Safer Infant Milk Formula”, *Health Economics*, 18, 503-518.
- Grosse, S.D. et al. (2002), “Economic gains resulting from the reduction in children’s exposure to lead in the United States”, *Environmental Health Perspectives*, 110 (6), 563-569.
- Guignet, D. (2014), “To Sell or Not to Sell: The Impacts of Pollution on Home Transactions,” *National Center for Environmental Economics Working Paper 14-01*, US EPA, Washington, DC.
- Guignet, Dennis, Rachel Northcutt, and Patrick Walsh (2015), “The Property Value Impacts of Groundwater Contamination: Agricultural Runoff and Private Wells”, *National Center for Environmental Economics Working Paper 15-05*, US EPA, Washington, DC.
- Guignet, D. et al. (2015), “Do Housing Values Respond to Underground Storage Tank Releases? Evidence from High-Profile Cases Across the United States”, *National Center for Environmental Economics Working Paper 16-01*, US EPA, Washington, DC.
- Gupta, S., G. van Houtven and M. Cropper (1996), “Paying for Permanence: An Economic Analysis of EPA’s Cleanup Decisions at Superfund Sites”, *RAND Journal of Economics*, 27(3), 563-582.
- Haab, T.C. and R.L. Hicks (1999), “Choice Set Considerations in Models of Recreation Demand: History and Current State of the Art”, *Marine Resource Economics*, 14(4), 271-281.
- Hammitt, J.K. (2016), “Valuing Health Risks: Monetary and Health-Utility Measures”, presented at the Experts Workshop on Valuing the Health Impacts of Chemicals, European Chemicals Agency, Helsinki, January, available at http://echa.europa.eu/news-and-events/events/event-details/-/journal_content/56_INSTANCE_DR2i/title/expert-workshop-on-valuing-the-health-impacts-of-chemicals.
- Hammitt, J.K. (2002), “QALYs versus WTP”, *Risk Analysis*, 22(5), 985-1001.
- Hammitt, J.K. and K. Haninger (2010), “Valuing fatal risks to children and adults: Effects of disease, latency and risk aversion” *Journal of Risk and Uncertainty*, 40, 57-83.
- Hammitt, J.K. and J-T. Liu (2004), “Effects of disease type and latency on the value of mortality risk” *Journal of Risk and Uncertainty*, 28(1), 73-95.

- Hanna, R. and P. Oliva (2011), “The Effect of Pollution on Labour Supply: Evidence from a Natural Experiment in Mexico City”, *NBER Working Paper* 17302.
- Harrington, W. and P.R. Portney (1987), “Valuing the Benefits of Health and Safety Regulation”, *Journal of Urban Economics*, 22: 101-12.
- Hintermann, B., A. Alberini and A. Markandya (2010), “Estimating the Value of Safety with Labour Market Data: Are the Results Trustworthy?”, *Applied Economics*, 42(9): 1085-1100.
- Industrial Economics (2016), “Economic Valuation Study for Canada’s Chemicals Management Plan: Final Survey Implementation and Summary Analysis”, Cambridge, MA, March.
- Johannesson, M., P-O. Johansson and K-G. Löfgren (1997), “On the Value of Changes in Life Expectancy: Blips Versus Parametric Changes”, *Journal of Risk and Uncertainty*, 15(3), 221-239.
- Kniesner, T.J. et al. (2006), “Pinning Down the Value of a Statistical Life”, *Center for Policy Research paper #85*, Maxwell School of Citizenship and Public Affairs, Syracuse University, Syracuse, NY, September.
- Kniesner, T. J., W. Kip Viscusi and C. Woock (2012), “The Value of Statistical Life: Evidence from Panel Data”, *Review of Economics and Statistics*, 94(1), 74-87.
- Krupnick, A. (2007), “Mortality-risk Valuation and Age: Stated Preference Evidence”, *Review of Environmental Economics and Policy*, 1, 261-282.
- Landrigan, P.J. et al. (2002), “Environmental Pollutants and Disease in American Children: Estimates of Morbidity, Mortality, and Costs for Lead Poisoning, Asthma, Cancer, and Developmental Disabilities”, *Environmental Health Perspectives*, 110 (7), 721-728.
- Leigh, J.P. (1995), “Compensating Wages, Value of a Statistical Life, and Inter-Industry Differentials,” *Journal of Environmental Economics and Management*, 28(1), 83-97.
- Maguire, K.B., N. Owens and N.B. Simon (2004) “The Price Premium for Organic Babyfood: A Hedonic Analysis,” (with K.B. Maguire and N. Owens), *Journal of Agricultural and Resource Economics*, 29(1): 132-149.
- Magat, W.A., W. Kip Viscusi and J. Huber (1996), “A reference lottery metric for valuing health”, *Management Science*, 42 (8), 1118-1130.
- Masur, J.S. and E.A. Posner (2015), “Unquantified Benefits and the Problem of Regulation Under Uncertainty”, University of Chicago Law School, available at http://papers.ssrn.com/sol3/papers.cfm?abstract_id=2646063.
- McDaniels, T.L., M.S. Kamlet and G.W. Fischer (1992), “Risk perception and the value of safety”, *Risk Analysis*, 12 (4), 495-503.
- McDonald, R.L. et al. (2016), “Dread and Latency Impacts on a VSL for Cancer Risk Reduction”, *Journal of Risk and Uncertainty*, <http://link.springer.com/article/10.1007/s11166-016-9235-x>.
- Mendeloff, J.M. and R.M. Kaplan (1989), “Are large differences in “lifesaving” costs justifies? A psychometric study of the relative value placed on perverting deaths”, *Risk Analysis*, 9 (3), 349-363.

- Moretti, E. and M. Neidell (2009), “Pollution, Health and Avoidance Behaviour: Evidence from the Port of Los Angeles”, *Journal of Human Resources*, 46(1), 154-175.
- Morris, J.A. and J.K. Hammitt (2000), “Using Life Expectancy to Communicate Benefits of Health Care Programs in Contingent Valuation Studies,” *Medical Decisionmaking*, Nov/Dec., 468-478.
- Meuhlenbachs, L., E. Spiller and C. Timmins (2015), “The Housing Market Impacts of Shale Gas Development”, *American Economic Review*, 105(12), 3633-3659.
- Navrud, S. (2016), Possibilities and challenges in transfer and generalisation of monetary estimates for environmental and health benefits of regulating chemicals, ENV/EPOC/WPIEEP(2016)11.
- Neumann, P. and M. Johannesson (1994), “The willingness to pay for in vitro fertilization: A pilot study using contingent valuation”, *Medical Care*, 32 (7), 686-699.
- OECD (2012), *Mortality Risk Valuation in Environment, Health and Transport Policies*, OECD Publishing, Paris, <http://dx.doi.org/10.1787/9789264130807-en>.
- OECD (2010), *Valuation of environment-related health risks for children*, OECD Publishing, Paris, <http://dx.doi.org/10.1787/9789264038042-en>.
- Pratt, John W. and Richard J. Zeckhauser (1996), “Willingness to Pay and the Distribution of Risk and Wealth”, *Journal of Political Economy*, 104(4), 747-763.
- Rabl, A., J.V. Spadaro and M. Holland (2014), *How Much is Clean Air Worth? Calculating the Benefits of Pollution Control*, Cambridge University Press, Cambridge, United Kingdom.
- Revesz, Richard L. (1999), “Environmental regulation, cost benefit analysis, and the discounting of human lives”, *Columbia Law Review*, 194 (4), 941-1017.
- Rice, G.E., J.K. Hammitt and J.S. Evans (2010), “A probabilistic characterization of the health benefits of reducing methyl mercury intake in the United States”, *Environmental Science and Technology*, 44(13), 5216-5224.
- Rowe, R. D. and L. G. Chestnut (1986), “Oxidants and asthmatics in Los Angeles: A benefits analysis”, EPA-230-09-86-018, Office of Policy Analysis, Washington, DC.
- Ryan, M. (1999), “Using conjoint analysis to go beyond health outcomes and take account of patient preferences: an application to in vitro fertilisation”, *Social Science and Medicine*, 48, 535–546.
- Sandy R. and F. Elliott (1996), “Unions and Risk: Their Impact on the Level of Compensation for Fatal Risk”, *Economica*, 63(250), p. 291-309.
- Sandy R. (2001), “Measurement Error and the Effects of Unions on the Compensating Differentials for Fatal Workplace Risks”, *Journal of Risk and Uncertainty*, 23(1), p. 33-56.
- Savage, I. (1993), “An empirical Investigation into the effects of Psychological Perceptions on the Willingness-to-pay to reduce risk”, *Journal of Risk and Uncertainty*, 6, 75-90.
- Ščasný, M. and I. Zverinova (2016), “The ECHA Study on Fertility, Borth Defects, and Developmental Toxicity: How Much Are Improving Fertility and Reducing Risk of Borth Defects Worth?” presented at the Experts Workshop on Valuing the Health Impacts of Chemicals, European

Chemicals Agency, Helsinki, January, available at http://echa.europa.eu/news-and-events/events/event-details/-/journal_content/56_INSTANCE_DR2i/title/expert-workshop-on-valuing-the-health-impacts-of-chemicals.

Schlosser, E. (2001), *Fast Food Nation*, Houghton Mifflin, Boston, MA.

Shimshack, J.P., M.B. Ward and T.K.M. Beatty (2007), “Mercury Advisories: Information, Education, and Fish Consumption”, *Journal of Environmental Economics and Management*, 53, 158-179.

Shimshack, J.P. and M.B. Ward (2010), “Mercury Advisories and Household Health Tradeoffs”, *Journal of Health Economics*, 29, 674-685.

Siebert, W.S. and X. Wei (1994), “Compensating Wage Differentials for Workplace Accidents: Evidence for Union and Non-union Workers in the U.K.”, *Journal of Risk and Uncertainty*, 9(1), 61-76.

Slovic, P. (1987), “Perception of risk”, *Science*, 236 (4799), 280-285.

Spadaro, J.V. and A. Rabl (2008), “Global health impacts and costs due to Mercury emissions”, *Risk Analysis*, 28 (3), 603-613.

Tolley, G.S., D. Kenkel and R. Fabian (eds.) (1994), *Valuing health for policy: An economic approach*, University of Chicago Press, Chicago.

Trasande, L. (2016), “Burden of Diseases of Costs of Endocrine Disrupting Chemicals in the European Union,” presented at the Experts Workshop on Valuing the Health Impacts of Chemicals, European Chemicals Agency, Helsinki, January, available at http://echa.europa.eu/news-and-events/events/event-details/-/journal_content/56_INSTANCE_DR2i/title/expert-workshop-on-valuing-the-health-impacts-of-chemicals.

Treich, N. (2010), “The value of statistical life under ambiguity aversion”, *Journal of Environmental Economics and Management*, 59, 15-26.

Tsuge T., A. Kishimoto and K. Takeuchi (2005), “A Choice Experiment Approach to the Valuation of Mortality”, *The Journal of Risk and Uncertainty*, 31(1), 73–95.

U.S. Environmental Protection Agency (2012), National Emission Standards for Hazardous Air Pollutants From Coal- and Oil-Fired Electric Utility Steam Generating Units and Standards of Performance for Fossil-Fuel-Fired Electric Utility, Industrial-Commercial-Institutional, and Small Industrial-Commercial-Institutional Steam Generating Units, *Federal Register*, Vol. 77, No. 32, February 16, 2012.

US Environmental Protection Agency (2010), Guidelines to Preparing Economic Analyses, US EPA, National Center for Environmental Economics, Washington, DC, December, available at [https://yosemite.epa.gov/ee/epa/erm.nsf/vwAN/EE-0568-50.pdf/\\$file/EE-0568-50.pdf](https://yosemite.epa.gov/ee/epa/erm.nsf/vwAN/EE-0568-50.pdf/$file/EE-0568-50.pdf).

U.S. Environmental Protection Agency (2007), *US EPA Cost of Illness Handbook*, National Service Center for Environmental Publications, <http://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=901A0E00.TXT>.

U.S. Environmental Protection Agency (1997), *Locating and estimating air emissions from sources of Mercury and Mercury compounds*, EPA-454/R-97-012, Washington, DC.

- Van Houtven, G., M.B. Sullivan and C. Dockins (2008), "Cancer premiums and latency effects: A risk tradeoff approach for valuing reductions in fatal cancer risks", *Journal of Risk and Uncertainty*, 36, 179-199.
- von Stackelberg, K. and J. Hammitt (2009), "Use of Contingent Valuation to Elicit Willingness-to-Pay for the Benefits of Developmental Health Risk Reductions", *Environment and Resource Economics*, 43, 45-61.
- Viscusi, W. Kip (1985), "Are Individuals Bayesian Decision Makers?", *American Economic Review*, 75(2), 381-385.
- Viscusi, W.K., W.A. Magat and J. Huber (1991), "Pricing environmental health risks: Survey assessments of risk-risk and risk-dollar tradeoffs for chronic bronchitis", *Journal of Environmental Economics and Management*, 21(1): 32-51.
- Viscusi, W. Kip (1993), "The Value of Risks to Life and Health," *Journal of Economic Literature*, 31(4), 1912-1946.
- Viscusi, W. Kip (1997), "Alarmist Decisions with Divergent Risk Information", *The Economic Journal*, 107(445), 1657-1670.
- Viscusi, W. Kip and J.E. Aldy (2007), "Labour Market Estimates of the Senior Discount for the Value of a Statistical Life", *Journal of Environmental Economics and Management*, 53(3), 377-392.
- Viscusi, W. Kip and J.E. Aldy (2003), "The Value of a Statistical Life: A Critical Review of Market Estimates Throughout the World", *Journal of Risk and Uncertainty*, 27(1), p. 5-76.
- Zivin, J. S. G. and M.J. Neidell (2011), "The impact of pollution on worker productivity", *NBER working paper no. 17004*, National Bureau of Economic Research, Washington DC.