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POSSIBILITIES AND CHALLENGES IN TRANSFER AND GENERALISATION OF MONETARY ESTIMATES FOR ENVIRONMENTAL AND HEALTH BENEFITS OF REGULATING CHEMICALS - ENVIRONMENTAL WORKING PAPER No. 119

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FOREWORD

This background paper was prepared by Dr. Ståle Navrud of the School of Economics and Business, Norwegian University of Life Sciences[1] 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.

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ABSTRACT

This paper reviews and discusses existing methodologies for transferring and extrapolating the economic value of health and environmental impacts across chemicals, and identifies challenges with such value transfer and when it can be suitable. The value transfer methodologies describes can be used to estimate the economic benefits of chemical management regulatory frameworks as a whole, as well as in cost-benefit analyses (CBAs) of risk management measures for individual chemicals.

For economic valuation of *mortality* risks from chemicals, the OECD database of Stated Preference (SP) studies of Value of Statistical Life (VSL) , which should be continuously updated with new valuation studies, has a sufficient number of primary studies internationally to conduct value transfer using meta-analytic regressions. However, the empirical evidence on acute and chronic *morbidity* endpoints, especially concerning all costs components of chronic illnesses, seems to be scarce. The same is true for chemical-related environmental impacts, especially related to ecosystem services, for the multitude of chemicals. Thus, the main methodological and informational challenge for valid value transfer of environmental and health impacts from chemical regulations seems to be new primary valuation studies of morbidity and ecosystem services impacts caused by exposure to (groups of) chemicals.

These new primary valuation studies should be designed with value transfer in mind, and cover several countries, in order to extrapolate and generalise the economic values to evaluate international chemical regulations in CBAs. These new primary studies should ideally cover all relevant scales of the impacts, in order to develop generalised adjustment factors for differences in scale of the impacts between the study sites and the policy site. This would improve the *spatial* transfer of values. The same is true for the combination of Geographical Information System (GIS) data with existing primary studies of impacts at different scales. Furthermore, these new primary studies should be repeated over time in order to provide more information about how values for the relevant impacts change over time; as preferences, scarcity of the public good and the real income of the affected population change. This would improve *temporal* transfer.

JEL codes: Q51, J17, Q53, Q57.

Key words: Value transfer, benefit transfer, chemicals regulations, health benefits, ecosystem services.

RÉSUMÉ

Le présent rapport présente et examine les différentes méthodes disponibles à l'heure actuelle pour transférer et extrapoler la valeur économique des impacts environnementaux et sanitaires de l'ensemble des substances chimiques et met en évidence les difficultés que pose ce transfert de valeurs et les cas dans lesquels il peut être pertinent. Les méthodes de transfert de valeurs décrites peuvent être utilisées pour estimer les avantages économiques induits par les cadres réglementaires de gestion des substances chimiques dans leur ensemble ainsi que pour effectuer des analyses coûts-avantages (ACA) des mesures de gestion des risques pour chacune de ces substances.

Lorsqu'il s'agit d'estimer la valeur économique des risques de *mortalité* due aux substances chimiques, la base de données de l'OCDE des études de préférences déclarées (PD) estimant la valeur d'une vie statistique (VVS) – qu'il faudrait veiller à enrichir sans cesse de nouvelles études de valorisation – comporte un nombre suffisant d'études primaires de différents pays pour permettre de réaliser un transfert de valeurs au moyen de régressions dans le cadre d'une méta-analyse. Il semble néanmoins que l'on dispose de très peu de données factuelles sur les critères de jugement de la *morbidité* aiguë et chronique, en particulier sur l'ensemble des composantes de coût des maladies chroniques. Il en va de même pour les impacts environnementaux liés à tout l'éventail des substances chimiques, en particulier s'agissant des services écosystémiques. Ainsi, d'un point de vue méthodologique et informationnel, il semble que la principale étape à franchir pour permettre un transfert de valeurs valide concernant les impacts environnementaux et sanitaires de la réglementation des substances chimiques soit de réaliser de nouvelles études primaires de valorisation des impacts sur la morbidité et les services écosystémiques dus à l'exposition à des (groupes de) substances chimiques.

Ces nouvelles études primaires doivent être conçues en gardant à l'esprit l'objectif de pouvoir réaliser un transfert de valeurs ; elles doivent en outre porter sur plusieurs pays pour qu'il soit possible d'extrapoler et de généraliser les valeurs économiques et ainsi d'évaluer la réglementation internationale sur les substances chimiques dans les ACA. Dans l'idéal, elles doivent couvrir toutes les échelles pertinentes des impacts étudiés afin de faciliter la mise au point de facteurs d'ajustement généralisés permettant de tenir compte des différences d'échelle entre les sites d'étude et le site de mise en œuvre des politiques. Le transfert *spatial* des valeurs s'en trouverait amélioré. Il le serait également en associant les données des systèmes d'information géographiques (SIG) aux études primaires d'impacts disponibles à diverses échelles. Par ailleurs, il est souhaitable que ces nouvelles études primaires soient renouvelées régulièrement afin de fournir des informations sur l'évolution dans le temps des valeurs associées aux impacts étudiés ; en effet, les préférences, la rareté des biens publics et le revenu réel de la population concernée évoluent. Cette fois, c'est le transfert *temporel* qui s'en trouverait amélioré.

Codes JEL : Q51, J17, Q53, Q57.

Mots clés : transfert de valeurs, transfert d'avantages, réglementation sur les substances chimiques, avantages sanitaires, services écosystémiques.

TABLE OF CONTENTS

POSSIBILITIES AND CHALLENGES IN TRANSFER AND GENERALISATION OF MONETARY ESTIMATES FOR ENVIRONMENTAL AND HEALTH BENEFITS OF REGULATING CHEMICALS

1. Introduction

1.1 Background

This is one of four background papers for the July 6-8 2016 workshop at European Chemicals Agency (ECHA) in Helsinki as part of the OECD project SACAME on the methodologies and information requirements for quantification of social costs of chemicals, and associated social benefits assessments of chemical management frameworks, like the EU REACH directive and similar national frameworks. It aims at reviewing and evaluating existing methodologies for value transfer of health and environmental impacts of chemicals, and their use in assessing the social benefits of regulating chemicals in cost-benefit analyses.

Value transfer (VT) involves transferring economic estimates from previous studies (often termed *study sites*) of similar changes in environmental quality and public health to value the change in the quality or quantity of theses public goods at the *policy site*. This procedure was initially termed *benefit transfer* (Desvousges et al., 1992), and although the term has stuck in the literature, a more general term increasingly used is *value transfer*, as damage estimates can also be transferred (see e.g. Navrud and Ready, 2007).

The *policy uses* of these transferred economic estimates in the area of management of chemicals include:

1. *Raising awareness* of the social costs of chemicals, and the social benefits of regulations of chemicals;
2. *Cost-benefit analysis (CBA)* of regulatory frameworks for managing chemicals, but also of individual measures to reduce use of chemicals and investment projects with impacts on chemicals use;
3. *Environmental accounting* at the national level, in terms of including externalities of chemicals use in green national accounts;
4. *Environmental costing*, i.e. calculating marginal external costs of chemicals as a basis for determining the optimal emission level and design of optimal regulatory instruments (e.g. environmental taxes);
5. *Natural Resource Damage Assessment (NRDA)* according to e.g. the US Oil Pollution Act or the Environmental Liability Directive of the European Union; i.e. calculate compensation payments for environmental and environmentally related health impacts from acute releases of chemicals.

The demand for accuracy in the environmental and health value estimates increase as one moves down this list of policy uses (Navrud and Pruckner, 1997), and the most frequent use of value transfer will probably be in CBAs of individual measures to reduce emissions of chemicals and regulatory policies,

including chemical management frameworks like the EU REACH directive and similar national frameworks.

1.2 Aim

The main aim of this paper is to review and discuss existing methodologies for transferring and extrapolating the economic value of health and environmental impacts across chemicals, and identify challenges with such value transfer and when it can be suitable. The paper will also contribute to estimating the economic value of chemical management frameworks as a whole, and not just the value of individual risk management measures, by identifying value transfer methodologies suited for assessing the social benefits of chemical regulations.

2. Requirements for value transfer (VT) and VT techniques

2.1 Requirements for value transfer

In order to perform benefit transfer, one needs:

1. Databases of primary valuation studies (to transfer values from);
2. Guidelines for assessing quality of primary valuation studies;
3. Value transfer techniques;
4. Value transfer guidelines.

The first requirement concerns a *database* for primary valuation studies with enough detail to judge similarity between the impacts valued in primary studies and the policies one is using value transfer to evaluate. Instead of having to conduct detailed literature searches and reviews every time the environmental and health impacts of new chemicals need to be assessed, a database of valuation studies of relevant environmental and health impacts would greatly ease this task. While such detailed databases have been constructed for meta-analyses of valuation studies of specific environmental and health impacts, they are usually not updated nor easily available. Thus, databases like the OECD database of Stated Preference (SP) studies of mortality risk reductions estimating the Value of Statistical Life (VSL) constructed for a global meta-analysis of VSL estimates from SP studies, should be continuously updated with new valuation studies (Lindhjem et al., 2011; OECD 2012). The Environmental Valuation Reference Inventory (EVRI), www.evri.ca, remains the most comprehensive and updated international database for both primary studies and meta-analyses valuing both environmental and environmentally related health impacts. However, the EVRI entries of primary studies often contain less detailed data than the databases of studies constructed for meta-analysis; and the information collected about each primary valuation study in EVRI will in some cases not be sufficient for the value transfer techniques and guidelines. Thus, the reporting of the valuation estimates and data for variables known to affect the value estimates should be specified in detail by the authors (e.g. as electronic appendices to journal articles and reports), and in a way that allows both a detailed quality assessment of the study and improved value transfer.

Regarding the second requirement listed above, guidelines for assessing the quality of Stated Preference (SP) and Revealed Preference (RP) methods valuing environmental impacts can be found in e.g. Söderqvist and Soutukorva (2006), while publications like the recent US EPA (2016) provide guidance for judging the quality of SP and RP studies valuing mortality risk.

Value transfer techniques are the third requirement. The three main techniques for spatial and temporal value transfer; i) unit value, ii) function transfer and iii) meta-analysis; are described in Sections 2.2-2.4, respectively. The fourth requirement is then to apply these value transfer techniques in a structured framework for value transfer, in terms of *value transfer guidelines*.

The remainder of this background paper is organised as follows: Section 3 presents the fourth requirement for value transfer in terms of the 8-step value transfer guidelines. Section 4 summarises the experiences from testing the validity of value transfer for health and environmental impacts. Section 5 discusses the possibilities and challenges in VT, while Section 6 concludes.

2.2 Unit value transfer

There are two main types of unit transfer: i) Simple (naïve) unit value transfer, and ii) Unit value transfer with income adjustments.

Simple (naïve) unit value transfer (in terms of a mean value from one study, or as a mean value estimate from several studies) is the simplest approach to transferring value estimates from a study site (or as a mean from several study sites) to the policy site. This approach assumes that the wellbeing experienced by an average individual at the study site is the same as will be experienced by the average individual at the policy site, and that the environmental or health impact in question is valued the same at the two sites. One directly transfers the mean value estimate;[2] usually expressed as mean willingness-to-pay (WTP) per household per year (or as a one-time amount) to avoid or obtain a specified change in the quantity or quality of an environmental good (or as individual consumer surplus per activity day for recreational use values), from the study site to the policy site.

For many decades, simple unit value transfer, in terms of the mean value of previous national valuation studies, was routinely used by the United States Department of Agriculture (USDA) Forest Service to establish a set of recreational use values per activity day for recreational activities to be used in CBAs of forest management practises in the United States. However, in the last decade or so, these unit values have been based on meta-analyses (Loomis, 2015). In the United States, simple unit value transfer has also been used in Natural Resource Damage Assessments (NRDA), called Type A assessments, of toxins, heavy metals, etc., from hazardous waste sites and mines, as well as for impacts on marine and coastal resources from oil spills (Loomis, 2015). The obvious problem with simple unit value transfer for recreational activities is that individuals at the policy site may not value recreational activities the same as the average individual at the study sites.

There are two principal reasons for this difference. First, people at the policy site might be different from individuals at the study sites in terms of income, education, religion, ethnic group or other socio-economic characteristics that affect their demand for recreation. Second, even if individuals' preferences for recreation at the policy and study sites were the same, the recreational opportunities (i.e., substitute sites and activities) and the nature of the change in the good valued might not be. Unit values for non-use values of e.g. ecosystem services from Contingent Valuation (CV) studies might be even more difficult to transfer than recreational (use) values for at least two reasons. First, the unit of transfer is more difficult to define. While the obvious choice of unit for use values is recreational value per activity day, there is greater variability in reporting non-use values from CV surveys, both in terms of WTP for whom, and for what time period. WTP might be reported both per household or per individual, and as a one-time payment, an annual payment for a limited time period, an annual payment for an indefinite time, or even as

2. However, as the unit value often originates from a study conducted some years ago, it is often adjusted for inflation to account for changes in value over time and implicitly assuming that people's valuation of environmental and health effects increase with the same rate as the domestic Consumers Price Index. This is a strict assumption as changes in peoples valuation of health and environmental impacts might be very different from changes in the CPI (see e.g. Zandersen et al., 2007a,b), and also the change in their real income over time could influence their valuation. The latter effect is typically corrected for in the adjusted unit value transfer approach discussed below.

monthly payments. Second, the WTP is reported for one or more specified discrete changes in ecosystem services, and not on a marginal (e.g. per ha) basis.

Whereas one considers the household as the unit for value transfer of environmental impacts, the individual is the natural unit for value transfer of health impacts. Morbidity impacts are often transferred in terms of unit values for a symptom day or an illness episode for acute illnesses, and per case for chronic illnesses.

For valuing changes in mortality risk, the Value of a Statistical Life (VSL) is most frequently used, but the Value of a Life Year (VOLY) is used in sensitivity analyses by the European Commission (Holland et al., 2005; Brouwer and Navrud, 2015). Unit value transfer in terms of transfer of a mean value from one national (best practice) primary valuation study, or a mean of many studies nationally (and sometimes internationally), have been used (and is still in use in many countries) to establish the Value of Statistical Life (VSL) for use in cost-benefit analyses both in the United States, Canada, the European Union and in individual European countries. Simple unit value transfer is currently also used in many countries, especially for morbidity impacts (Ready and Navrud, 2007), whereas meta-analysis is increasingly used to determine unit values for VSL (Viscusi and Aldy, 2003; OECD, 2012; Lindhjem et al., 2011; US EPA, 2016).

For environmental goods, WTP per household per year would be the preferred transfer unit, and then the unit value would be aggregated over the total number of affected households (i.e. households whose utility is affected by the impact) in order to get an estimate of total benefits (or costs). Using WTP per individual per year for environmental goods might lead to overestimation of total benefits when aggregated over individuals, or underestimation if individuals underestimate other household members' WTP (see e.g. Lindhjem and Navrud, 2009). WTP as a one-time amount might lead to underestimation of annual WTP, as the reported WTP will be the present value of a flow of annual WTP amounts and will be constrained by their income in the year they report their one-time amount WTP. Aggregating WTP per household, for a specified change in ecosystem services in the same size area, over the number of affected households avoids the procedure of scaling up and down reported WTP to the size of the area at the policy site. Such scaling assumes a constant value per ha (and no non-linearities in valuation); which does not seem to be the case in practice (See e.g. Lindhjem, 2007; Lindhjem and Navrud, 2008; and Kaul et al., 2013).

Thus, simple value transfer for environmental and health impacts could be based on estimates from only one primary study or the mean estimate from many valuation studies considered being close to the policy site; both geographically and culturally, and in terms of similarity of the characteristics of the public good valued.

The simple unit value transfer approach should not be used for transfer between countries with very different income levels and costs of living (or between regions with very different income levels within a country). Then, *unit transfer with income adjustments* should be applied if one is conducting unit value transfer. The adjusted WTP estimate $B_p{}'$ at the policy site can be calculated as

$$WTP_p{}' = WTP_S\,(Y_p / Y_S)^{ß} \qquad (2.1)$$

where WTP_S is the original WTP estimate from the study site, Y_S and Y_p are the income levels at the study and policy site, respectively, and *ß* is the income elasticity of WTP for the environmental good in question. This income elasticity of WTP (*ß*) for different environmental, morbidity and mortality impacts is typically smaller than 1, and often in the 0.3-0.7 range (Kriström and Riera, 1996; Hökby and Söderqvist, 2003; Desaigues et al., 2006; OECD, 2012; Lindhjem et al., 2013). When one lacks data on the income levels of the affected populations at the policy and study sites, gross domestic product (GDP) per capita has been

used as a proxy for income in international benefit transfers. However, this approach could give wrong results in international benefit transfers when income levels at the local study and/or policy site deviates from the average income level at the national level.

Using the official exchange rates to convert transferred estimates in e.g. US dollars to the national currencies does not reflect the true purchasing power of currencies, since the official exchange rates reflect political and macroeconomic risk factors. If a currency is weak on the international market (partly because it is not fully convertible), people tend to buy domestically produced goods and services that are readily available locally. This enhances the purchasing powers of such currencies on local markets. To reflect the true underlying purchasing power of international currencies, the International Comparison Program (ICP) has developed measures of real GDP on an internationally comparable scale. These Purchasing Power Parities (PPPs) should be used to correct for differences in purchasing power between countries.[3]

Even if PPP-adjusted GDP figures and exchange rates can be used to adjust for differences in income and cost-of-living in different countries, doing so will not correct for differences in individual preferences, baseline levels of environmental quality, ecosystem services and public health nor international differences in cultural and institutional conditions (or even within different parts of a country). Thus, population characteristics should be as similar as possible between the study and policy sites.

2.3 Value function transfer

Transferring the entire *value function* is conceptually and theoretically more appealing than just transferring unit values because more information is effectively taken into account in the transfer. The evidence is mixed with regards to whether function transfers perform better than unit value transfer for environmental and health impacts. For example, Ready et al. (1997) found unit value transfer (with income adjustment) to perform better than function transfer for respiratory illnesses while Bateman et al. (2011) found the opposite result for water quality improvements. In many instances, value function transfer does not seem to reduce transfer errors significantly compared to simple unit value transfer (with income adjustment). Bateman et al. (2011) also note that their result that value function transfer outperforms unit value transfer in their international value transfer validity test of WTP for water quality improvements in five European countries is based on transfer of highly similar sites and commodities.

The value relationship to be transferred from the study site(s) to the policy site could be estimated using either revealed preference (RP) approaches like the Travel Cost (TC) and Hedonic Price (HP) methods or stated preferences (SP) approaches like the Contingent Valuation (CV) method and Choice experiments (CE); see Alberini (2017) for a review of the valuation methods in this context. For a CV study, the value function can be written as:

$$WTP_{ij} = b_0 + b_1 G_j + b_2 H_{ij} + e \qquad (2.2)$$

where WTP_{ij} = the willingness-to-pay of household (or individual if one is valuing health impacts) i at site j, G_j = the set of characteristics of the environmental good or health impact at site j, and H_{ij} = the set of characteristics of household/individual i at site j, and b_0, b_1 and b_2 are sets of parameters, and e is the random error.

To implement this approach, the analyst would have to find a study in the existing literature with estimates of the constant b_0 and the sets of parameters, b_1 and b_2. Then the analyst would have to collect data on the two groups of independent variables, G and H, at the policy site, insert them in equation (2.2), and calculate households' WTP at the policy site. One challenge in value function transfer is the lack of

3. For ICP, see http://siteresources.worldbank.org/ICPEXT/Resources/ICP_2011.html.

data at the policy site for the independent variables explaining most of the variation in WTP at the study site. Thus, instead of using the "best" value function (in terms of significant variables and highest explanatory power), one normally has to resort to the second-best option in terms of a value function containing variables for which there are data at the policy site.

Another problem with the value function approach is due to the exclusion of relevant variables in the WTP (or bid) function estimated in a single study. When the value function is based on observations from a single study of a recreational site, a specific change in environmental quality or a specific illness episode, a lack of variation in some of the potentially important independent variables usually prohibits inclusion of these variables. For domestic value transfers, researchers tackle this problem by choosing the study site to be as similar as possible to the policy site with regards to socioeconomic and health characteristics of the population and/or the environmental quality or ecosystem service change in question.

2.4 Meta-analysis

Instead of transferring the value function from one selected valuation study, results from several valuation studies could be combined in a *meta-analysis* to estimate one common value function. Meta-analysis has been used to synthesise research findings and improve the quality of literature reviews of valuation studies in order to come up with adjusted unit values. In a meta-analysis, several primary valuation studies are analysed as a group, where the estimates from each study is treated as a single observation in a regression analysis. If multiple estimates from each study are used, various meta-regression specifications can be used to account for such panel effects.

Meta-analysis allows one to evaluate the influence of a wider range in characteristics of the environmental good or health impacts, the features of the samples used in each analysis (including characteristics of the population affected by the change), and the modelling assumptions. In practice, however, detailed characteristics of the good or study site and the population are often not reported in the primary studies (in particular, if they are published journal papers, which often focus on methodological tests of valuation methods rather than reporting monetary estimates and the data needed in a meta regression analysis), and it requires a large effort to find them (if at all possible). The resulting regression equations explaining variations in unit values can then be used together with data collected on the independent variables in the model that describes the policy site in order to construct an adjusted unit value. The regression from a meta-analysis would look similar to equation (2.2), but a set of variables reflecting differences in the valuation methods applied need to be added; i.e. C_s = characteristics of the methodology applied in study *s*; as meta-analyses typically find that differences in valuation methods account for a significant part of the variation in mean WTP across studies.

3. Value transfer guidelines for environmental and health impacts of chemicals

Value transfer guidelines should be practical and simple to use, and show in a transparent and step-by-step manner how one can arrive at economic values for health and environmental impacts of chemicals. The guidelines below are based on other similar general guides for value transfer for environmental goods, like the Danish EPA Guidelines (Navrud, 2006) and the UK Defra Guidelines (Bateman et al., 2009).

Eight main steps for value transfer for environmental and health impacts of chemicals can be identified:

1. Identify the environmental and health endpoints or impacts to be valued at the policy site;
2. Identify the affected population (i.e. the population thought to experience welfare loss from the impact) at the policy site, and the characteristics likely to influence their values of the respective impact;

3. Conduct a literature review (from databases of primary studies and other sources) in order to identify relevant primary studies; preferably of a population with similar characteristics as the population at the policy site;
4. Assess the relevance and quality of study site values for transfer;
5. Select and summarise the data available from the study site(s);
6. Transfer value estimate from study site(s) to policy site;
7. Calculate total social benefits or costs; aggregated over the affected population and geographical area if WTP/household is expressed per unit of area for environmental goods and over time, in terms of their Present Value (PV); and
8. Assess the uncertainty and transfer errors.

3.1. Identify the environmental and health endpoints to be valued at the policy site

Identify and quantify the environmental and health impacts to be valued at the policy site, in terms of the endpoint from a damage function impact pathway approach involving dose-response functions (DRFs) for environmental impacts and exposure-response functions (ERFs) for health impacts. However, one often lacks specific DRFs and ERFs, and therefore these endpoints are often assessed by experts in separate assessments or as part of Environmental Impacts Assessments (EIAs), or Health Impact assessments (HIAs), or in risk assessments.

This step involves:

1. Identifying the types of health or environmental impact, and the direction of impact (positive or negative) and unit of measurement and transfer; and
2. Describing the baseline and magnitude of the health or environmental impact.

3.2 Identify the affected population at the policy site and the characteristics influencing their values of the respective impact

Early value transfer guidelines, like Desvousges et al. (1998), had this step as their last step in their value transfer guide. However, it is important to identify the size of the affected population at the policy site before one reviews the valuation literature and evaluate the relevance of selected studies, and more recent value transfer guidelines (Navrud, 2007; and Bateman et al., 2009) have this as one of the initial steps. The transferred value should come from the same type of affected population, in terms of whether one looks at users and/or non-users, and whether the good in question is of importance to the population at the local, regional or national (or even international or global) level. Population characteristics also need to be similar in order to ensure they share the same type and level of welfare determinants.

This is particularly important for environmental impacts. If one just wants to establish the recreational use value of an environmental good, the relevant, affected population is the recreationists. If one would like to estimate both use and non-use values, and the policy site is only of local importance (e.g. a small forest area with many substitute sites regionally), one should use only the population of the municipality. If there are few substitutes for the sites at the regional level, the population in several communities, or even the county population, should be used. If the good is of national importance, e.g. a national park, or the single site of an endangered species in the country, the national population should be used. For use values, the number of individual recreationists should be estimated (before and after the change), while for non-use values (or use and non-use values combined), the number of households should be the unit of aggregation at the relevant geographical scale (community, regional or county or national level).

3.3 Conduct a literature review to identify relevant primary studies

This step involves conducting a literature search to identify relevant primary studies; preferably based on a database (e.g. EVRI, www.evri.ca), but supplemented by journal and general web search, in order to identify *similar* studies from the same country or other closely located countries (which share the same type institutional and cultural context). This recommendation is based on value transfer validity tests showing that studies closer spatially tend to have lower transfer errors (see Section 4). Studies closest in time should be selected for the same reason. The current practice of using the Consumer Price Index (CPI) of the country where the policy site is located is at best a crude approximation of how people's preferences and values for environmental and health impacts change over time (as these goods are not fully represented in the basket of goods for which the CPI is calculated). While there are several studies testing transferability in space, only a few studies test transferability over time; especially over long time periods typical of time horizons for CBAs. Studies of environmental goods, like non-timber forest externalities, show that values are stable over periods of one year or less, but could change drastically over longer time spans, of e.g. 20 years (Zandersen et al., 2007 a, b). This could be due both to changes in users' and non-users' preferences, recreational use patterns, availability and quality of substitute recreational sites and activities, and changes in the quantity and quality (i.e. scarcity) of biodiversity and ecosystem services.

3.4 Assess the relevance and quality of study site values for transfer

Here, the quality of the relevant valuation studies is assessed in terms of scientific soundness and richness of information. Desvousges et al. (1998) identify the following main criteria for assessing the quality and relevance of candidate studies for transfer:

1. Scientific soundness; the transfer estimates are only as good as the methodology and assumptions employed in the primary studies. Assess quality of the primary valuation study using state-of-the-art guidelines and checklists for conducting revealed and stated preference studies.
2. Relevance; primary studies should be similar and applicable to the "new" context.
3. Richness in detail; primary valuation studies should provide a detailed dataset and accompanying information.

All three criteria are equally important for assessing the relevance and quality of the primary study for transfer, and form the basis for checklists for judging the similarity of characteristics of the good and population at the study sites versus policy site described in step 8 (see Section 3.8).

3.5 Select and summarise the data available from the study site(s)

Several parallel approaches should be applied, and the results from these should be used to present a range of values:

- Search the studies to provide low and high estimates, which can define a lower and upper bound for the transferred estimate, respectively. Collect data on the mean estimate and standard error of each study, and specific spatial transfer errors if available.
- Consult relevant meta-analyses to see if the scope of these is narrow enough to provide relevant information about the estimate to be transferred; as a check on the unit value transfer performed. The scope of the meta-analysis could be too wide to produce reliable estimates if the meta-analysis consists of studies which vary a lot in terms of methodology and the environmental or health impact considered.
- Compare the magnitude of the value from the meta-analyses, when methodological parameters in the meta-function are set according to the best practice guidelines and a context corresponding to the policy site.

3.6 Transfer value estimate from study site(s) to policy site

1. *Determine transfer unit* (e.g. Value of Statistical Life, willingness-to-pay (WTP) per household per year for a specified water quality decrement)
2. Determine value transfer method for *spatial transfer*
3. The value transfer methods for spatial transfer include: i) unit value transfer (with income adjustments for international transfers), ii) value function transfer (from one primary valuation study) and iii) meta-analytic function transfer (from a meta-analysis of several primary valuation studies). For value transfer between countries, Purchasing Power Parity (PPP)-corrected exchange rates should be used to translate into the relevant currency for all these transfer methods.
4. Determine value transfer method for *temporal transfer.*

The value transfer method for temporal transfer is usually the Consumer Price Index (CPI) (i.e. assuming for simplicity that the valuation of environmental impacts and resource use increase at the same rate over time as the bundle of goods that are used to calculate CPI) to get the values to a fixed, common price level. Expected increase in GDP per capita and an income elasticity range can be used to estimate the value in future years. While this approach will suffice for use in CBAs with time horizons of 20-40 years, changes in preferences of future generations should also be accounted for in CBAs with longer time horizons. Skourtos et al. (2016) provides a framework for taking into account "greening" of preferences of future generations due to increased awareness and scarcity of environmental goods due to environmental degradation and depletion of natural resources. This additional relative increase in the value of environmental goods would counteract (some of) the effects of discounting.

As the primary study reflects the values held by people at the time of the study, spatial VT should be undertaken first. Then, the temporal VT should be performed by using the CPI of the policy site country to reflect changes in preferences over time of the population at the policy site.

3.7 Calculate **total** *social benefits or costs*

Annual benefits B_t are equal to aggregated WTP over the affected population (WTP_{tot}), which can be calculated as:

$$WTP_{tot} = \sum_{i=1}^{n} WTP_i \qquad (3.0)$$

where n = number of affected households (for environmental goods) or individuals (for health impacts), and WTP_i = mean willingness-to-pay for household or individual *i*. Since WTP per household varies between different parts of the affected population (e.g. with distance from the site, whether users or non-users are considered, population characteristics, baseline mortality risks, etc.), the estimates from the study site(s) should be based on the same type of affected population as at the policy site. If this is not possible, distance decay in WTP for environmental goods (e.g. percentage reduction in WTP per km increased distance from the environmental good) could be assumed, based on empirical evidence from relevant study sites (if such evidence does exist and suggests this). For health impacts, one could perform similar type adjustments for spatial differences in population characteristics, baseline mortality risks, etc. (if such evidence does exist and suggests this).

Thus, aggregate benefits would usually be calculated as the mean value per household (or individual) times the number of households (individuals) in the affected population for each environmental (health) impact; and aggregated over health impacts for each chemical (and over chemicals), and over the relevant time horizon to estimate the present value of the environmental and health impacts. Examples of these calculations include:

1. Value of Statistical Life multiplied by the number of cases of premature deaths (in terms incidence rate times the size of the "affected population");
2. Mean WTP for avoiding a case of a chronic illness multiplied by the number of cases of chronic illnesses (in terms of the incidence rate times the size of the "affected population");
3. Mean WTP per household for avoiding a specified environmental impact as a one-time amount multiplied by the number of affected households;
4. Mean WTP per household per year for avoiding a specified environmental impact or number of acute morbidity episodes per year; multiplied by the number of affected households to get the aggregate value, and then added up over the time horizon of the impacts, by calculating the present value, using the social discount rate of the country or a region (e.g. the European Union).

When aggregating damages and costs of environmental goods, one also needs to consider whether these goods are independent (meaning one can just add them up), or if they are substitutes or complements. If they are substitutes, one would overestimate aggregated damage or benefits, while if they are complements, we would underestimate.

3.8 Assess the uncertainty and transfer errors

In order to account for the added uncertainty of value transfer instead of conducting a primary valuation study of the environmental or health impact in question, a generalised mean transfer error could be added using Table 1. The transfer exercise should be placed in one of the four categories of "similarity" between the study site and the policy.

Table 1. Four categories of how similar the study site is to the policy site and the corresponding transfer errors for unit value transfers

Category	Level of fit between study and policy sites	Percentage transfer error
1	Perfect fit	20
2	Acceptable fit	50
3	Poor fit	100
4	No fit	Discard study for this VT exercise

Note: These indicative transfer errors are based on a review of transfer errors from the benefit transfer validity test literature. The judgment of similarity should be based on the checklist of site and population characteristics described below this table.

The checklist below for two broad sets of criteria can serve as a guide to issues that need to be considered when judging the level of "similarity" between values from primary valuation studies at study sites and values needed at the policy site for assessing environmental or health benefits of chemical regulations.

Characteristics of the good

- Similar good? (i.e. similar type of health impact, environmental quality impact, similar use and non-use value components; similar recreational activities, similar ecosystem services (ES)?)
- Similar *size*, *baseline* and *direction* of change in the good valued?

- One should avoid scaling values up and down according to the *size* of change in quality or quantity of the public good in terms of the area of the ecosystem service, environmental quality level (e.g. water quality level), morbidity impact and mortality risk; as this usually implicitly assumes a constant unit value (e.g. constant value per ha of forest area for the use and non-use values). This assumption often does not hold, see e.g. Lindhjem (2007), and especially not for large differences in the scale of the change. The general recommendation is to choose a domestic primary study as close as possible geographically, and thus culturally, (and in time) to the policy site. However, in the case of no domestic primary study valuing an approximately similar size change in the public good, one should rather consider foreign studies with nearly similar size changes (instead of trying to adjust the values from the domestic primary study). The same recommendation applies to the judgment of similarity in terms of the *baseline* quality or quantity of the public good and the *direction* of the change
- For value transfer of environmental impacts it is beneficial if there is similar *availability of substitute* sites at the policy site as at the study site. Substitute sites for use values are other recreational sites and/or activities; whereas for substitute sites for non-use values one would have to consider national parks and other preserved areas (and the ecosystem services they contain)

Population characteristics

- Similar average *income* level (and income distribution)? If not, income adjustments (using GDP per capita as a proxy) should be made when performing the value transfer.
- Similar *gender, age and educational* composition, and other population characteristics significantly affecting people's valuation?
- Similar *size* of affected population? Expected similar *distance decay* in value, if any, in non-use values?
- Similar "property rights", e.g. the same right to use forest areas for recreation?
- Similar attitudinal and cultural factors, which affects people's valuation?

The transfer errors in Table 1 above refer to the *mean* WTP estimate, and would come *in addition* to the inherent uncertainty of the valuation methods applied in the primary valuation study at the study site. The uncertainty about the size of the affected population should also be taken into account.

As the size of the affected population is equally important for calculating the total benefit estimate, sensitivity analyses should also be conducted for the *size* of the affected population. If there is evidence of distance decay in WTP in the primary study that one think could be transferred to the policy site, sensitivity analysis with WTP and population estimates for each distance zone should be performed (see Bateman et al., 2009; Box 21 for an example).

When performing a cost-benefit analysis of a new project or policy, the discount rate used should be explicitly stated, and the estimated present value (PV) of benefits (costs) should be compared with the corresponding PV of costs (benefits). Adding the expected transfer error (from Table 1.) could widen the range of the expected PV of benefits (increases the costs) to include the critical level where the PV of net benefits becomes negative. If this is the case, the transfer errors are large enough to change the outcome of our CBA, and one should try to increase the accuracy of the transferred estimate (either by conducting a full primary study or calibrating the transferred value by conducting a small-scale primary study)

When there is a need for estimates of environmental goods for policy purposes, a CBA of conducting a new environmental valuation study should be performed in order to determine whether the costs of a new primary study is worth the benefits in terms of lower probability of making the wrong decision.

These decision rules could be used as a rough test of whether value transfer has acceptable transfer errors.

4. Validity and transfer errors of value transfer -- Empirical evidence

4.1 Mortality impacts

Mortality risk reductions, in terms of reducing the number of premature deaths due to illnesses such as cancer, could constitute a major part of the benefits in CBAs of regulatory frameworks for chemicals. Recently, Hammitt and Robinson (2015) concluded in their review of US revealed preference (i.e. hedonic wage-risk) studies and stated preference studies that despite the increasing number of illness-related mortality risk studies the last few years, few US studies of illness-related risks meet their criteria for quality. The studies that did, found similar VSL estimates to studies of injury-related risks. They combine the findings of these few studies with the findings of the larger literature on injury-related VSL in order to provide a reasonable range of estimates for application in regulatory CBAs in the United States, of USD 4.2 million to USD 13.7 million with a mid-point of USD 9.0 million (2013 dollars). They note that although the identified studies differ from those that underlie the VSLs currently used by US Federal agencies, the resulting estimates are remarkably similar. Although this study does not constitute a formal value transfer test, it indicates that injury-related mortality risk studies could be used to approximate the VSL to use for all illness-related mortality risks, including those stemming from chemicals.

However, this transferability can be questioned based on the results from stated preference studies, which have the advantage of eliciting the preferences of the overall population and not only at the preferences of the workforce (in the hedonic wage risk studies). Alberini and Ščasný (2013) find that VSL increases with dread, exposure to risk, and the respondents' assessments of the baseline risks; and it is higher when the risk reduction is delivered by a public programme, and increases with the effectiveness rating assigned by the respondent to the mode of the risk reduction. Even when they control explicitly for all of these factors, the cause of death per se accounts for a large portion of the VSL. They did not find that people discounted risk when there was latency. They found, however, that, all else the same, the fact that the cause of the death was "cancer" resulted in a VSL about one million euro *higher* than the amount predicted by dread, exposure and other risk perception variables. The VSL in the road safety context was about one million euro *less* than what is predicted by dread, exposure, or beliefs compared to VSL for the respiratory risk context. Thus, the effect of cause of death was found to be as large as the effect of other sources of VSL heterogeneity. Even if this recent European choice experiment study is concerned with respiratory illness mortality risks, there could well be a "cancer premium" also for cancer-related mortality risk from chemicals.

Based on a meta-analysis of stated preferences studies of VSL worldwide in the environment, transport and health sectors (Lindhjem et al., 2011; OECD, 2012), Lindhjem and Navrud (2015) show how quality screening of studies in meta-analytic value transfer of VSL can yield lower transfer errors (compared to no quality screening), and achieve accuracy gains over the use of the unit value transfer method. They conclude that transfer accuracy may in some contexts depend as much on the quality of the underlying data (i.e. measurement errors) as on the value transfer technique itself (i.e. generalisations error).

4.2 Morbidity impacts

In their meta-analysis of value transfer validity studies of public goods, Kaul et al. (2013) found that only 2 out of 31 transfer validity studies concerned health impacts (while the rest concerned environmental goods). Rosenberger (2015) in his more recent meta-analysis of 38 transfer validity studies of public goods identified the same two studies. Both CV studies tested the validity of morbidity impacts in terms of

respiratory illnesses (Ready et al., 2004) and skin cancer (Brouwer and Bateman, 2005). Both Ready et al. (2004) and Brouwer and Bateman (2005) value endpoints from exposure-response functions for air pollution and UV radiation, respectively.

Ready et al. (2004) valued five acute respiratory symptoms in five European countries (Spain, Portugal, Norway, the Netherlands and the United Kingdom), and performed value transfer validity tests. They found transfer errors of international value transfer for these respiratory symptoms (ranging from light symptoms of "coughing days" to hospital admission) to be 38% on average. They found little difference in transfer error between simple (naïve) unit value transfer (without any adjustment), unit value transfer with adjustment for income, and value function transfer.

Brouwer and Bateman (2005) conducted the same CV study in New Zealand, Scotland, England and Portugal and undertook transfers between these countries. The results show that when transferring between similar contexts, simple unit value transfers outperform more complex value function transfers; having a mean transfer error of 0.4 and 18.7%, respectively. However, this result was reversed when transfers were undertaken across dissimilar contexts where value functions partially adjust for these differences. The mean transfer errors were then 31 and 9% for simple unit value transfer and value function transfer, respectively.

Thus, these international value transfer tests indicate that the mean transfer errors for acute morbidity are acceptable for use in CBAs. While there are many stated preference studies valuing *acute* morbidity impacts, there are few valuing *chronic* illnesses, and few valuing health impacts from chemicals. A notable exception is the multi-country (Czech Republic, UK, Italy and the Netherlands) study recently commissioned by the European Chemicals Agency (ECHA) valuing chronic illnesses (as well as acute) of five key endpoints caused by chemicals exposure: (1) sensitisation, (2) dose toxicity, (3) effects on fertility, (4) developmental toxicity, and (5) carcinogens (Ščasný et al., 2014). Ščasný et al. (2014) do not perform value transfer tests between the four countries where their valuation studies are performed, but they use the overall sample to establish central European estimates for use in CBAs of the European chemicals regulatory framework, REACH.

4.3 Environmental impacts

Brouwer (2000) suggests that if non-use values of environmental goods are motivated by overall commitment to environmental causes, they may be relatively constant across populations and contexts. In a contingent valuation survey of the national populations in all Nordic countries, Kristofersson and Navrud (2007) found that transfer errors are consistently smaller for non-use values of a preservation plan for Nordic freshwater fish stocks (which is a resources that could typically be affected by emissions of chemicals) than recreational use value of the same fish stocks. For example, for value function transfers between Sweden and Norway, transfer errors for recreational use values ranged from 16% to 34% while transfer errors for non-use values were only 8–11%. It may be that non-use values in these two countries are motivated by similar factors and are relatively context independent, while recreational use values are more context-specific. Clearly, this result could vary with different resources, particularly if the good has cultural significance in one country.

Bartczak, Lindhjem and Stenger (2008) provide an overview of the studies in Europe and the United States which have tested the validity and transfer errors of transferring use and non-use values of forests. They found 12 studies dealing with benefit transfer between forest sites and a few others in which forest sites were among other analysed environmental resources. The majority of these studies transferred recreation benefits, using a benefit function based either on contingent valuation or travel costs estimations. They mainly focused on four areas: physical attributes of forests, time aspects, and methodological improvements to increase the estimated accuracy and reduce surveys costs. This can serve

as an example of characteristics it is important to control for when conducting value transfer to value environmental impacts.

Kaul et al. (2013) and Rosenberger (2015) in their reviews of value transfer validity tests found that the majority of these tests concerns outdoor recreation, environmental quality and ecosystem services. Rolfe et al. (2015) note that the findings of these two and other such evaluations of transfer errors are broadly similar and mostly intuitive, and well represented by Kaul et al. (2013). Their results suggest that: (i) value function transfers tend to outperform unit value transfers, (ii) transfer of values for quantities are more accurate than those for qualities, (iii) geographic site similarity influences transfer errors, (iv) contingent valuation estimates are associated with systematically lower transfer errors than other stated and revealed preference techniques, and (v) the combination of data from multiple studies can reduce transfer errors.

While there are many studies of value transfer for environmental goods across sites and locations, both nationally and internationally, there are few studies testing the validity of value transfer of ecosystem services across habitats. However, recently Interis and Petrolia (2016) used a choice experiment to value ecosystem services and perform transfer across three habitats (oyster reef, salt marsh and black mangrove) in two US Gulf Coast locations. They found unit value transfer of ecosystem services across habitats that provide overlapping ecosystem services to perform well, whereas there were mixed results for transfers of values for same type habitats across locations. While the results are not conclusive, this indicates that it is possible to transfer ecosystem services values from habitats that differ from the policy habitat, and that this could be better than transferring habitat values across locations. Further research is needed to establish whether an ecosystem service approach to value transfers can broaden the basis for value transfer, avoid transfer errors due to site-specific issues, and reduce overall transfer errors. Also, one needs to test whether these results for national transfers also hold for international transfers of values of ecosystem services. However, for environmental impacts from chemicals, reporting impacts in terms of effects on specific ecosystem services should be explored as a way of improving value transfer.

Ready and Navrud (2005) concluded their review of validity tests of international value transfer that mean transfer errors for international value transfers tended to be in the range of 20% to 40%, but some transfers have errors as high as 100-200%. These results were similar, both in the size of the mean transfer error and the error range to those found in intra-country transfers at that time. 10 years after, Johnstone et al. (2015) reports the state-of-the-art for environmental value transfer, and their conclusions supports the conclusion of Ready and Navrud (2005) that both domestic and international environmental value transfers have acceptable transfer errors for use in CBAs of projects, measures and policies.

5. Possibilities and challenges

Value transfer for estimating the benefits of regulatory frameworks for chemicals in terms of avoided or reduced impacts on environmental quality and ecosystem services as well as morbidity and mortality risks, face much of the same possibilities and challenges as in value transfer in other areas. However, as opposed to e.g. CBAs of regulatory frameworks for e.g. air pollution and water quality improvements, there seems to be few primary valuation studies of endpoints related to chemicals through exposure-response functions or risk assessments. This poses the additional challenge of value transfer across different contexts; as illustrated by the added uncertainty of transferring VSL from injury-related mortality risks to an illness-related VSL, and especially a cancer-related illness (which also begs the question of whether peoples' preferences vary across cancer types).

To summarise, there are six main challenges in value transfer for estimating the benefits of chemicals regulation:

1. The ability to translate risk assessments to health and environmental endpoints for valuation and value transfer.
2. The availability of primary valuation studies for value transfer of each of the identified endpoints and impacts.
3. The frequent need for international value transfer in a situation with a limited number of primary studies internationally (and the need to account for international differences in cultural and institutional contexts, in addition to the uncertainty in national, spatial value transfer).
4. Addressing the "scaling" issue. When there are few primary studies, it is difficult to find primary studies valuing the same level of impact, and there is a need to scale the result from the primary study up or down. This increases the uncertainty, as the often implicit assumption of constant marginal values does not hold; because individual values change with the size of impact, and the size of impact could influence the size of the affected population (and thus the aggregate value).
5. Temporal transfer errors, both in terms of transferring values over time from existing primary studies, but also when predicting future values in CBAs with a time horizon of many decades.
6. Addressing the "adding-up"-issue. Moving from benefit assessment of regulating one chemical to also address a larger groups of chemicals covered by regulations like REACH, one need to take account of possible interactions between these chemicals in all stages of the damage function and impact pathway approach used. Both the impacts and people's valuation of these impacts might not be independent, and this needs to be accounted for when aggregating benefits over different chemicals when performing CBAs of regulatory frameworks affecting many chemicals.

6. Conclusions and recommendations

6.1 Suitability of value transfer to assess social benefits of chemicals regulations

In theory, value transfer can be just as well fit for chemicals regulations as for other environmental regulations. The wide range of chemicals and resulting ranges of environmental and health impacts pose a challenge in the evaluation of overall regulatory frameworks for chemicals such as REACH. Theoretically, one should map each type of environmental and health impact from each (group of) chemicals, identify all relevant primary valuation studies of these impacts, perform value transfer; and aggregate across chemicals. However, crucial information on both dose-response relationships, and thus the impacts of individual or groups of chemicals, as well as economic estimates of their environmental and health impacts is lacking.

An obvious question is then whether it is possible to extrapolate from existing assessments of the environmental and health costs of one or a few chemicals in order to assess all chemicals covered by regulatory assessments such as REACH. This, however, currently seems like an impossible task due to the multitude of chemicals and the large knowledge gaps in all steps of the damage function impact pathway approach for most chemicals.

A tentative approach would be to apply the value transfer methodology outlined in this paper to improve the existing assessments of individual chemicals, which usually covers selected impacts and selected cost components of these impacts. Existing economic assessments typically cover only health (not environmental) effects, and are sometimes based on costs-of-illness and productivity losses rather than the overall welfare loss (including loss in well-being and other costs components); see e.g. Trasande and Liu (2011) and Barlett and Trasande (2013). Improved assessments of individual chemicals would then provide better estimates of the order of magnitude costs avoided by regulating these specific chemicals, and contribute to identify which parts of the damage function approach and which cost components future research efforts should be directed towards in order to close the knowledge gaps for economic assessments

of other chemicals. Utilising risk assessments to establish causational relationships between more chemicals and environmental and health effects would be an obvious step in the right direction; see Chiu (2017).

A second-best tentative approach could be to utilise the extensive literature on lost DALY (and QALY) for many chemicals in Global Burden of Disease Assessments; and combine the aggregate impacts in terms of DALYs multiplied by Value of a Life Year (VOLY).[4] However, the methodology for estimating VOLY is less developed than VSL, and VSL avoids the ethical issues of summarising life years (and implicitly valuing older adults less than younger adults, if a constant VOLY is assumed). If rough order of magnitude estimates of the avoided costs and social benefits of chemical regulations overall are needed fast, this second-best alternative could still be explored. Note, however, that DALYs are based on expert judgements/preferences, whereas QALYs are based on individual preferences just as the welfare theoretic basis for VOLY. Thus, QALYs rather than DALYs should preferably be combined with VOLY; see also a recent attempt to value QALYs directly by Hammitt and Haninger (2017).

6.2. Methodological and informational requirements for valid assessment

Although there seem to be a sufficient number of primary studies internationally to conduct value transfer exercises of mortality impacts using VSL meta-analyses (Lindhjem et al., 2011; OECD, 2012), the OECD database of Stated Preference (SP) studies of VSL should be continuously updated with new valuation studies, The empirical evidence on morbidity endpoints, however, especially with regards to all costs components of chronic illnesses, seems to be scarce. The same is true for chemical-related environmental impacts (especially related to ecosystem services) for the multitude of chemicals. Thus, the main methodological and informational requirement for valid value transfer of environmental and health impacts from chemical regulations seems to be new primary valuation studies of environmental and morbidity impacts scenarios specifically related to (groups of) chemicals.

These primary studies should be designed with value transfer in mind, and cover several countries, in order to extrapolate and generalise the values to evaluate international chemical regulations. These new primary studies should ideally also cover all relevant scales of the impacts, in order to develop generalised adjustment factors for differences in scale of the impacts between the study site and policy site. This would improve the spatial transfer of values. The same is true for the combination Geographical Information System (GIS) data with existing primary studies of impacts at different scales; see e.g. Brander et al. (2012) for an application to valuing all European wetlands based on a set of primary valuation studies. Furthermore, these new primary studies should be repeated over time in order to get more information about how values for the relevant impacts change over time, as both preferences, scarcity of the public good and the real income of the affected population change. This would improve temporal transfer.

4. As most assessments of health impacts of chemicals reports premature acute deaths and DALYs (e.g. Weinhold, 2011), a possible second-best solution in a transition period, could be to combine Value of a Life Year (VOLY) estimates from e.g. Desaigues et al. (2011)'s 9-country European study of VOLY with existing estimates of impacts in terms of saved Quality Adjusted Life Years (QALYs) or Disability-Adjusted Life years (DALY) estimates (e.g. from Global Burden of Disease) that might exist for chemicals-induced acute and chronic illnesses (and their related mortality risks). However, this raises the question of the theoretical foundation, as well as reliability and relevance of combing QALY and DALY estimates with VOLY, and the reliability and relevance of VOLY estimates themselves in valuing changes in mortality risk.

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