

Economic assessments of the benefits of regulating mercury: A review

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Foreword

This paper on economic assessments of the benefits of regulating mercury was prepared for the SACAME workshop in Ottawa, Canada of 30-31 August 2017 by Richard Dubourg of The Economics Interface Limited.

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

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

Abstract

This paper gives an overview of economic assessments of the benefits of the control of emissions of mercury compounds, discusses their completeness from a social cost point of view, and discusses the relative magnitudes of the values attached to mercury compounds in different contexts. The majority of the assessments have been conducted in the context of coal-fired electricity generation and the valuation of human health impacts linked to ingestion of methylmercury. The focus on foetal exposure to methylmercury through maternal fish consumption leading to neurological developmental impacts results in the primary measure of economic impact being the effect of IQ change on i.a. labour market performance. A small number of studies have also considered the possibility of impacts of methylmercury ingestion on cardiovascular health in the general population. The decision to include such impacts or not is important since general population changes in cardiovascular risks can have high value when measured in terms of willingness-to-pay for mortality risk reductions. Gaps in the assessments regarding the coverage of endpoints are primarily due to gaps in the underlying science, not to gaps in the evidence concerning the economic value of changes in the outcomes.

JEL codes: Q51, Q53, Q58

Keywords: Mercury, impacts, benefits, valuation, policy

Résumé

Ce document propose un tour d'horizon des évaluations économiques réalisées sur les avantages du contrôle des émissions de composés mercuriels, une analyse de l'exhaustivité de ces évaluations sous l'angle des coûts sociaux, ainsi qu'un examen des ordres de grandeur relatifs des valeurs applicables aux composés mercuriels dans différents contextes. La majorité des évaluations ont pour toile de fond la production d'électricité à partir de charbon et portent sur les incidences de l'ingestion de méthylmercure sur la santé humaine. Elles se concentrent sur les impacts que peut avoir sur le développement neurologique du fœtus l'exposition au méthylmercure due à la consommation de poisson contaminé par la femme enceinte, et l'impact économique est donc mesuré principalement à l'aune de l'effet de la variation du QI sur, par exemple, la performance sur le marché du travail. Un petit nombre d'études s'intéresse aussi aux possibles incidences de l'ingestion de méthylmercure sur la santé cardiovasculaire de la population générale. La décision de faire entrer en ligne de compte ces incidences ou non est importante, car des variations des risques cardiovasculaires encourus par la population générale peuvent avoir une grande valeur lorsqu'elles sont mesurées en termes de consentement à payer pour une réduction du risque de mortalité. Les lacunes des évaluations en ce qui concerne les effets pris en compte sont principalement imputables à l'insuffisance des données scientifiques, et non des données économiques sur les effets

Codes JEL : Q51, Q53, Q58

Mots-clés : Mercure, impacts, avantages, valorisation, action publique

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Abbreviations and acronyms

AMI	Acute myocardial infarction
ASGM	Artisanal small-scale gold-mining
AUD	Australian dollars
CAD	Canadian dollars
CAIR	Clean Air Interstate Rule
CAMR	Clean Air Mercury Rule
CMAQ	Community Multiscale Air Quality model
DALY	Disability-adjusted life-years
ECHA	European Chemicals Agency
EUR	Euro
FAO	Food and Agriculture Organization
FGD	Flue gas desulphurisation
GBD	Global Burden of Disease
GDP	Gross domestic product
IQ	Intelligence quotient
MATS	Mercury and Air Toxics Standards
MR	Mental retardation
NESCAUM	Northeast States for Coordinated Air Use Management
NHANES	National Health and Nutrition Examination Survey
OECD	Organisation for Economic Co-operation and Development
PPP	Purchasing power parity
QALY	Quality-adjusted life-year
REACH	Registration, Evaluation, Authorisation and restriction of Chemicals
RfD	Reference dose
RGM	Reactive gaseous mercury
RIA	Regulatory impact assessment
UNEP	United Nations Environment Programme
USD	United States dollars
US EPA	United States Environmental Protection Agency
VSL	Value-of-a-statistical-life
WHO	World Health Organization

Executive summary

The objectives of this paper are to give an overview of the available economic assessments of the benefits of the control of emissions of mercury compounds, to discuss their completeness from a social cost point of view, and to discuss the relative magnitudes of the values attached to mercury compounds in different contexts.

The majority of detailed socio-economic analyses that have been undertaken so far have been in the context of coal-fired electricity generation, almost exclusively in the United States. Regulatory focus in the developed world is now shifting towards other sources of mercury emissions, particularly mercury in products and used in dentistry. A future need for analysis of mercury emissions in artisanal small-scale gold mining (ASGM) and more generally in a developing world context is apparent.

The principal routes of human health impacts from mercury exposure are through direct inhalation of mercury vapour, and through ingestion of methylmercury. The former is the primary route for those working in the ASGM sector. Studies of the economic effects of these health impacts are few. The main source of ingestion of methylmercury (the second route) is the consumption of fish. Methylmercury is a form of mercury which is created in the environment from other types of mercury, and which bioaccumulates in fish, particularly in longer-lived species near the top of the food chain. This fish consumption route to mercury exposure has a number of important implications, as follows:

1. The benefits of mercury emissions reductions are subject to potentially complex pathways, from the source of the emissions to deposition into marine environments, take-up by fish and subsequent human consumption. Some impact-pathway analysis studies set out highly detailed models of this link comprising several steps.
2. Ultimately, who is affected by mercury emissions and emission reductions depends on where the emissions are generated and deposited, how much they affect methylmercury concentrations in fish, which types of fish and by whom those fish are consumed.
3. The different types of mercury are generated in varying proportions (“speciation”) dependent on the source, and have different deposition properties. This means that different sources can have different near-field and long-distance deposition implications, and hence affect different fish and, thereby, human populations. Reducing emissions at a given source might entail few benefits for individuals close to that source, or even for the country in which the source is located.
4. The fact that human exposure through fish consumption is mediated via atmospheric deposition and bioaccumulation processes means that the time delays between “cause” (emission) and “effect” (human health response) can be considerable (estimated often to be decades or even longer).

Studies of the health impacts of mercury vary in the extent to which they take account of these factors.

The neurological development impacts of foetal exposure to methylmercury through maternal fish consumption is the primary focus (directly or indirectly) of all studies considered in this review. The primary source of evidence for those impacts is three epidemiological studies conducted in the Faroe Islands, New Zealand and the Seychelles. Studies in this review make use of these results directly, or of results of integrated analyses of the combined datasets. The focus on neurological development means that the primary measure of economic impact has been the effect of IQ changes on labour market performance, particularly earnings. Other potential outcomes associated with neurological development impacts have received comparatively little (or no) consideration.

A small number of studies have also considered the possibility of impacts of methylmercury ingestion on cardiovascular health in the general population, based on a limited number of small epidemiological studies. This outcome has been excluded by some authors who consider this evidence to be too weak. The decision to include them or not is important since, even if small, general population changes in cardiovascular risks can have high value when measured in terms of willingness-to-pay for mortality risk reductions. Studies that have included cardiovascular risks have found they dominate the value of IQ-related health impacts.

Finally, although other human health impacts of mercury have been suggested – such as genotoxic effects and carcinogenicity – the evidence is not considered strong and no study currently has included them as endpoints. The same is true of environmental impacts, which are included (qualitatively) by only one study considered in this review.

A brief summary of the main findings of this review would be as follows.

First, the general context of existing studies is undeniably coal-fired electricity generation in the United States. This is the context in which all of the most sophisticated analyses have been undertaken. (In fact, no other OECD member country has undertaken a study even remotely close in analytical rigour or sophistication as those undertaken in the United States.) As already discussed, this has important implications in terms of the parameters and relationships governing speciation, deposition, fish consumption patterns and so on, all of which this review has demonstrated to be relevant to the valuation of mercury emission reductions. Because no similarly sophisticated studies have been conducted in other contexts and countries, it is not clear just how transferable to other contexts the values obtained from these US studies actually is. Although mercury is a global pollutant, the value of emission reductions does vary, and possibly significantly, across countries.

Second, there is an important related issue concerning the geographical and temporal coverage of analyses. Depending on speciation and other factors, a relatively high proportion of the benefits of mercury emission reductions could be international (“external”). In addition, there is disagreement in the literature about the extent to which marine fish should be a pathway to exposure affected by individual country emissions. Evidence suggests that the time taken for emission reductions to show up in ecosystem and thereafter health impacts could be extended – decades or longer. This implies a need for long time horizons for analysis and also underlines the importance of discounting. However, values are being transferred from detailed impact-pathway analyses for use in simpler studies (including governmental regulatory impact assessments) without a clear recognition of these (and other) important factors which affect their transferability. As a result, it is not clear that the transferred values provide an accurate indicator of the true values of the benefits of emission reductions in the contexts to which they are transferred.

Third, the focus in terms of economic outcomes so far has been on IQ-related earnings loss. Other health and environmental impacts of mercury are possible, but the evidence underpinning them is comparatively weak (or worse) and generally considered insufficient to support quantitative analysis. Some studies have included cardiovascular effects which dominate values when they are included, but their existence is very uncertain. Estimating willingness-to-pay for general mercury impacts would be one way of addressing this scientific uncertainty but would itself be subject to significant uncertainty, which would affect the reliability of resulting values. The conclusion is that the current state of scientific knowledge about mercury's effects means there are not significant practical gaps in the valuation of endpoints. Gaps in the coverage of endpoints are due to gaps in the underlying science, not to gaps in the economic evidence.

Fourth, some significant variation in values has been observed as a result of studies using different dose-response assumptions (particular the slope and functional form, and whether a threshold is employed). Different assumptions can cause values to increase by an order of magnitude (although variation within impact-pathway studies specifically is smaller). There is a more general issue about the strength of the evidence covering the mercury-IQ-earnings relationship, in particular in terms of establishing the continuation of observed childhood impacts into adulthood, which is currently based largely on assumption. It has been suggested that new evidence on the mercury-IQ (and general cognitive development) connection is likely to be forthcoming over the next few years.

Finally, it is not considered currently possible to make generalisations about the "best values" to be used in future socio-economic analyses. Useful future analysis would undertake a systematic, quantitative assessment of how the various value-relevant parameters affect transferability, and indicate what adjustments might be appropriate to make transfers more accurate.

Synthèse

Ce document propose un tour d'horizon des évaluations économiques réalisées sur les avantages du contrôle des émissions de composés mercuriels, une analyse de l'exhaustivité de ces évaluations sous l'angle des coûts sociaux, ainsi qu'un examen des ordres de grandeur relatifs des valeurs applicables aux composés mercuriels dans différents contextes.

La majorité des analyses socio-économiques détaillées conduites à ce jour l'ont été dans le contexte de la production d'électricité au charbon, et ce presque exclusivement aux États-Unis. Toutefois, dans les pays développés, l'attention de la réglementation se tourne désormais vers d'autres sources d'émission de mercure, en particulier le mercure présent dans les produits ou utilisé dans le secteur dentaire. Il est également devenu nécessaire d'analyser les rejets de mercure dans le secteur de l'extraction artisanale et à petite échelle de l'or (orpaillage) et, plus généralement, dans les pays en développement.

Les principales voies d'exposition au mercure qui présentent un danger pour la santé humaine sont l'inhalation directe de vapeur de mercure et l'ingestion de méthylmercure. La première est la voie d'exposition majeure pour les orpailleurs. Peu d'études ont été consacrées aux effets économiques de ces impacts sur la santé. S'agissant de la deuxième voie d'exposition, la principale source d'ingestion de méthylmercure est la consommation de poisson. Produit dans l'environnement à partir d'autres formes du mercure, le méthylmercure se concentre par bioaccumulation dans les poissons, en particulier les espèces du sommet de la chaîne alimentaire, qui vivent plus longtemps. Cette voie d'exposition au mercure par la consommation de poisson a plusieurs conséquences importantes :

1. Les bénéfices de la réduction des émissions de mercure dépendent de voies potentiellement complexes, qui vont de la source des émissions jusqu'au dépôt dans les environnements marins, à l'absorption par les poissons puis à la consommation humaine. Certaines analyses du cheminement des impacts définissent des modèles très détaillés de cette relation à plusieurs étapes.
2. Les facteurs qui déterminent en définitive qui sera touché par les émissions de mercure et par leur réduction sont le lieu où le mercure est rejeté et déposé, l'ampleur des effets de ces dépôts sur la concentration en méthylmercure dans les poissons, le type de poisson consommé et les consommateurs de ce poisson.
3. Les différentes formes de mercure sont produites en différentes proportions (spéciation), en fonction de la source, et leur dépôt présente des caractéristiques variables. Autrement dit, chaque source a des conséquences différentes sur le dépôt, tant au niveau local qu'à de plus longues distances, donc touche des poissons différents et, par conséquent, des populations humaines différentes. Réduire les émissions d'une source donnée pourrait donc générer peu d'avantages pour les personnes proches de la source, voire pour la population du pays où se trouve la source.

4. Le fait que l'exposition humaine au mercure par la consommation de poisson mette en jeu des processus comme le dépôt par voie aérienne et la bioaccumulation signifie que le délai entre la « cause » (émission) et l'« effet » (impact sur la santé humaine) peut être considérable (on l'estime souvent à des décennies, voire plus).

Les études des impacts du mercure sur la santé diffèrent principalement parce qu'elles n'accordent pas le même poids à chacun de ces facteurs.

Toutes les études passées en revue dans la présente recension portent avant tout (directement ou indirectement) sur les impacts que peut avoir sur le développement neurologique du fœtus l'exposition au méthylmercure due à la consommation de poisson contaminé par la femme enceinte. La première source d'information concernant ces impacts est un ensemble de trois études épidémiologiques conduites dans les Îles Féroé, en Nouvelle-Zélande et aux Seychelles. Les études de cette recension utilisent directement ces résultats, ou les résultats d'analyses globales des ensembles de données combinés. L'attention portée au développement neurologique signifie que la première mesure de l'impact économique est l'effet de la variation de QI sur la performance sur le marché du travail, en particulier le niveau de revenu. Les autres conséquences possibles des impacts sur le développement neurologique ont été très peu (voire pas du tout) examinées.

Un petit nombre d'études s'intéressent également aux impacts possibles de l'ingestion de méthylmercure sur la santé cardiovasculaire de la population générale, sur la base d'un nombre limité de petites études épidémiologiques. Mais ce résultat a été exclu par certains auteurs qui considèrent que les éléments de preuve sont insuffisants. La décision de les inclure ou non est importante puisque, même lorsqu'elles sont faibles, des variations des risques cardiovasculaires encourus par la population générale peuvent avoir une grande valeur lorsqu'elles sont mesurées en termes de consentement à payer pour une réduction du risque de mortalité. Les études qui ont pris en compte les risques cardiovasculaires montrent que ces risques dominent la valeur des effets sur la santé associés au QI.

Enfin, bien qu'il ait été suggéré que le mercure peut avoir d'autres impacts sur la santé humaine – par exemple des effets génotoxiques et cancérigènes – les éléments de preuve ne sont pas jugés solides et ne figurent dans aucune étude en tant que paramètres. Il en est de même des impacts environnementaux, qui ne sont inclus (qualitativement) que dans une seule des études examinées ici.

Les principales conclusions de cette recension peuvent être résumées comme suit.

Premièrement, le contexte général des études existantes est indéniablement la production d'électricité au charbon aux États-Unis. C'est dans ce contexte que s'inscrivent toutes les analyses les plus élaborées. (De fait, aucun autre pays membre de l'OCDE n'a conduit d'étude dont la rigueur ou la finesse de l'analyse soit comparable à celles des études américaines.) Comme indiqué plus haut, cet état de fait a d'importantes répercussions pour ce qui est des paramètres et des relations régissant la spéciation, le dépôt de mercure ou les profils de consommation de poisson, autant de paramètres que cette recension a jugés pertinents pour l'évaluation de la réduction des émissions de mercure. Dans la mesure où aucune étude aussi élaborée n'a été conduite dans d'autres contextes ou d'autres pays, il est difficile de savoir dans quelle mesure les valeurs obtenues par ces études américaines sont effectivement transférables à d'autres contextes. Si le mercure est

un polluant partout sur la planète, la valeur de la réduction des émissions varie, sans doute beaucoup, d'un pays à l'autre.

Deuxièmement, la couverture géographique et temporelle des analyses soulève des questions importantes. En fonction de la spéciation et d'autres facteurs, une proportion relativement élevée des avantages de la réduction des émissions de mercure pourrait être internationale (« externe »). De plus, les avis diffèrent sur la pertinence de l'étude des poissons de mer pour l'évaluation de l'exposition aux émissions de mercure des différents pays. Certains éléments probants semblent indiquer que le temps nécessaire pour qu'une baisse des émissions se répercute dans l'écosystème puis sur les impacts sur la santé pourrait être long – de l'ordre de plusieurs décennies ou plus. Cela signifie qu'il faut conduire des analyses à long terme, mais aussi qu'il est important de tenir compte de l'actualisation. Or, on transfère des valeurs provenant d'analyses détaillées du cheminement des impacts pour les utiliser dans des études plus simples (notamment des analyses d'impact de la réglementation destinées aux pouvoirs publics) sans clairement établir les facteurs importants (et d'autres) qui influent sur la transférabilité. De ce fait, il n'est pas sûr que les valeurs transférées fournissent un indicateur précis de la valeur réelle des avantages d'une réduction des émissions dans les contextes dans lesquels elles sont transférées.

Troisièmement, les évaluations économiques ont jusqu'à présent mis l'accent sur la baisse du revenu liée à la baisse du QI. Le mercure peut avoir d'autres effets sur la santé et l'environnement, mais les éléments de preuve en la matière sont comparativement faibles (ou très faibles) et généralement considérés comme insuffisants pour soutenir l'analyse quantitative. Certaines études incluent les effets cardiovasculaires, qui dominent alors les valeurs, bien que leur existence soit tout à fait incertaine. Pour faire face à cette incertitude scientifique, on pourrait estimer le consentement à payer pour les effets généraux du mercure, mais cette estimation comporterait elle aussi une grande incertitude qui influencerait sur la fiabilité des valeurs obtenues. On peut conclure que, compte tenu de l'état actuel des connaissances scientifiques sur les effets du mercure, l'évaluation des paramètres ne présente pas en pratique de lacunes importantes. Les lacunes de la couverture des paramètres sont dues à l'insuffisance des données scientifiques, et non des données économiques.

Quatrièmement, on observe d'importantes variations des valeurs qui résultent de l'utilisation dans les études d'hypothèses différentes concernant la dose-effet (en particulier sur la pente et la forme fonctionnelle, et la prise en compte ou non d'un seuil). Les différences entre les hypothèses peuvent faire monter les valeurs d'un ordre de grandeur (bien que les variations soient moins marquées en ce qui concerne les études du cheminement des impacts). Une question plus générale se pose au sujet de la solidité des éléments de preuve concernant la relation mercure-QI-niveau de revenu : en particulier, l'idée que les effets observés dans l'enfance existent toujours chez l'adulte se fonde largement sur des hypothèses à l'heure actuelle. Il semble que des données nouvelles sur la relation entre mercure et QI (et développement cognitif général) pourraient être disponibles dans les années à venir.

Enfin, il ne paraît pas possible pour l'instant de procéder à des généralisations sur les « meilleures valeurs » à utiliser dans les analyses socio-économiques à venir. Pour être utile, l'analyse devrait comporter à l'avenir une évaluation quantitative systématique de l'influence des différents paramètres des valeurs sur la transférabilité, et indiquer les ajustements à opérer pour que les transferts aboutissent à des résultats plus exacts.

Economic assessments of the benefits of regulating mercury: A review

1. Background and objectives

The OECD is carrying out a project (SACAME)¹ aimed at supporting the socio-economic analysis of chemicals, through enabling better quantification and monetisation of morbidity and environmental impacts of their production, use and disposal. One element of this project was a workshop held in Helsinki, Finland in July 2016, on the methodologies and information requirements for quantification of these social costs. As background for that workshop, the OECD commissioned four background papers that discussed the types of information available in a typical chemical risk assessment and their translation into economic assessments; the methodologies and information requirements for estimating economic values; the possibilities for transferring benefit estimates from one assessment to others; and the need for ex post analyses of chemicals regulations. These papers are available at <http://oe.cd/sacame>.

Building on the outcome of that workshop, the SACAME project has continued with case studies of the economic valuations which have been carried out on five selected substances or groups of substances – phthalates, mercury compounds, PFOA and its salts, 2-NMP (1-Methyl-2-pyrrolidone) and formaldehyde – and related health and environmental endpoints. The aim is to provide “best estimates” of the social costs related to impacts caused by these substances, and to share experiences regarding the use of economic valuations related to countries’ risk management of chemicals. This paper represents the outcome of the mercury case study.

Mercury was chosen to be part of the project because it was judged to provide an opportunity for comparative analysis of valuation approaches between jurisdictions within a more data-rich environment than other chemicals. For example, Canada has regulated mercury for several uses and has developed an economic valuation to support regulations under the Canadian Environmental Protection Act (Canada Gazette, 2014_[1]). There are also socio-economic analyses in support of mercury restrictions under the EU’s REACH regulation for measuring devices and in the production of polyurethane coatings (ECHA (2011_[2]) and (2011_[3])). Mercury compounds have also been regulated in other jurisdictions under the UN Minamata convention (see e.g. Marsden Jacob Associates (2015_[4])). The most extensive literature relates particularly to the context of limiting air pollution emissions from coal-fired power stations in the United States. However, these

¹ Socio-economic Analysis of Chemicals by Allowing a better quantification and monetisation of Morbidity and Environmental impacts.

studies vary in the extent to which they capture all relevant social costs, including people's willingness-to-pay to avoid the endpoints of concern.

The objectives of the paper are to give an overview of the available economic assessments regarding mercury compounds, to discuss their completeness from a social cost point of view, and to discuss the relative magnitudes of the values attached to mercury compounds in different contexts. However, as shall be made clear in the review, the ability to make this comparison is limited by the significant variations in the complexity of analysis across different studies. A relatively small number of highly complex studies tend to provide results which are used by other, simpler ones. These complex studies are reviewed in some detail in an attempt to understand which factors are the most important drivers of their results. In so doing, the paper tries to identify weaknesses and gaps in studies and suggest ways in which they could be improved.

The paper outline is as follows. First, a brief description of the sources and impacts of mercury is provided, identifying some of the key features of the "system" which are important for socio-economic analysis. This provides a context in which the existing socio-economic analyses of mercury can be characterised. A number of these studies are then described in greater or lesser detail, to identify their main methodological features, results and results drivers, as well as sources of variation across studies. An assessment of the important features and drivers is then provided. The paper concludes with some thoughts on improving future socio-economic analyses on mercury.

2. The impacts of mercury on health and the environment

The science of the human and environmental health impacts of mercury have been extensively covered elsewhere and it is not the purpose of this paper to provide a detailed consideration here. However, it is worth drawing attention to a number of aspects which are of relevance to socio-economic analysis of policies to regulate the use and emission of mercury and its compounds.

First, coal-burning continues to be the largest source of mercury emissions in the developed world, although it has been declining in volume over time with a shift away from coal as the basis of electricity generation. Coal-burning in the developing world has been increasing with recent economic development. UNEP (2013_[5]) estimates that coal-burning is now the second biggest source of anthropogenic mercury emissions globally (24%). The largest source (37%) is from artisanal small-scale gold mining (ASGM), where mercury has traditionally been used to separate the gold from impurities. ASGM occurs particularly in south-east Asia, South America and sub-Saharan Africa, and together with coal-burning, these areas are now responsible for 40%, 12% and 16% of global anthropogenic emissions. The EU and North America are now responsible for 4.5% and 3.1% of these emissions respectively. Mercury emissions have been addressed (directly or indirectly) in the developed world over the last two decades through the implementation of regulations on air pollution, and there is now a move towards addressing other sources (such as the use of mercury in products), related to drivers such as the UNEP Minamata Convention (www.mercuryconvention.org). This Convention was

agreed in July 2013, came into force in August 2017 and, as of May 2018, had 128 signatories and 91 parties. It includes a ban on new mercury mines, the phase-out of existing ones, the phase-out and phase-down of mercury use in a number of products and processes, control measures on emissions to air and on releases to land and water, and the regulation of the informal sector of artisanal and small-scale gold mining.

Against this backdrop, it is perhaps unsurprising that the majority of detailed socio-economic analyses that have been undertaken so far have been in the context of coal-fired electricity generation, almost exclusively in the United States (e.g. US EPA (2005_[6]) and (2011_[7])). The Minamata Convention effectively requires an uptake of air pollution-control technologies such as flue gas-desulphurisation (FGD), which has already largely been installed in the developed world. As a result, regulatory focus there is already shifting towards other sources of mercury emissions, particular mercury in products and used in dentistry (e.g. Canada Gazette (2014_[11]), ECHA (2011_[2]) and (2011_[3])). A future need for analysis of mercury emissions in ASGM and more generally in a developing world context is apparent.

Second, the primary routes of human health impacts from mercury exposure are through direct inhalation of mercury vapour, and through ingestion of methylmercury. The former is the primary route for those working in the ASGM sector, where sufferers can experience erethism (“mad hatter disease”), tremor, gingivitis, and other symptoms which have been likened to alcoholism (Steckling et al., 2014_[8]). Studies of the economic effects of these health impacts are few (e.g. Steckling et al. (2014_[8]) and (2017_[9])). The second route is the single major route for public health purposes, since the principal source of ingestion of methylmercury is the consumption of fish. Methylmercury is a form of mercury which is created in the environment from other types of mercury (elemental or metallic mercury, particulate mercury and oxidised or reactive mercury), and which bioaccumulates in fish, particularly in longer-lived species near the top of the food chain. This fish consumption route to mercury exposure has a number of important implications, as follows:

1. The benefits of mercury emissions reductions are subject to potentially complex pathways, from the source of the emissions to deposition into marine environments, take-up by fish and subsequent human consumption. Some impact-pathway analysis studies (e.g. US EPA (2005_[6]) and Rice and Hammitt (2005_[10])) set out highly detailed models of this link comprising several steps;
2. Ultimately, who is affected by mercury emissions (and who benefits from emission reductions) depends on where the emissions are generated and deposited, how much they affect methylmercury concentrations in fish, which types of fish and whom those fish are consumed by;
3. The different types of mercury are generated in varying proportions (“speciation”) dependent on the source and have different deposition properties. This means that different sources can have different near-field and long-distance deposition implications, and hence affect different fish and, thereby, human populations. Reducing emissions at a given source might entail few benefits for individuals close to that source, or even for the country in which the source is located;
4. The fact that human exposure through fish consumption is mediated via atmospheric deposition and bioaccumulation processes means that the time delays between “cause” (emission) and “effect” (human health response) can be considerable (estimated often to be decades or even longer (US EPA (2005_[6])).

Studies of the health impacts of mercury vary in the extent to which they take account of these factors. As already mentioned, some (e.g. US EPA (2005_[6])) go to considerable lengths to identify each step in detail. Others make a number of simplifying assumptions or take short-cuts between steps (e.g. Spadaro and Rabl (2008_[11])). Still other studies focus on the impacts of human exposure and do not attempt to explain how that exposure has been caused or would be affected by changes in emissions. These different types of study are characterised below and considered in more detail in the next section.

As stated above, workers in the ASGM sector have demonstrated symptoms associated with direct inhalation exposure to elemental (metallic) mercury vapours. The potential health impacts of mercury exposure through fish consumption came to prominence following industrial accidents. In particular, between 1953 and 1960, organic mercury was discharged into the Minamata Bay in Japan, contaminating fish and resulting in methylmercury poisoning of the local population through their fish consumption. Symptoms of exposure included paresthesia (burning sensation in the skin), ataxia (failure of muscle control), sensory problems (e.g. impaired vision, hearing and smell), tremors, irritability, and others, including death (National Research Council, 2000_[12]). Importantly, children of exposed women displayed a higher incidence of symptoms than exposed adults, with some victims born with a condition resembling cerebral palsy, with severe disturbances of nervous function and delays in development (US EPA (2005_[6])).

The neurological development impacts of foetal exposure to methylmercury through maternal fish consumption is the primary focus (directly or indirectly) of all studies considered in this review. The primary source of evidence for those impacts is three epidemiological studies conducted in the Faroe Islands (Grandjean et al., 1997_[13]), New Zealand (Kjellström et al., 1989_[14]), (Crump et al., 1998_[15]), and the Seychelles (Davidson et al., 1998_[16]), (Myers et al., 2003_[17]). These studies considered various measures of foetal mercury exposure (e.g. maternal hair concentrations, umbilical cord blood concentrations) and a range of neuro-developmental outcomes (tests of cognitive functioning such as IQ, problem solving, social behaviour, language, motor skills, attention, memory etc.), assessed at various ages of the mothers' children. Studies in this review make use of these results directly, or of those which have performed integrated analyses of the combined datasets (e.g. Ryan (2005_[18]), Axelrad et al. (2007_[19])). The focus on neurological development means that the primary measure of economic impact has been the effect of IQ changes on labour market performance, particularly earnings (via quite specific studies of the effect of IQ on earnings (Schwartz (1994_[20]); Salkever (1995_[21])). Other potential outcomes associated with neurological development impacts (e.g. non-earnings-related wellbeing, outcomes associated with very low IQ) have received comparatively little (or no) consideration.

A small number of studies have also considered the possibility of impacts of methylmercury ingestion (via fish consumption) on cardiovascular health in the general population, based on a limited number of small epidemiological studies. This outcome has been excluded by some authors on the basis of weak evidence. The decision to include them or not is important since, even if small, general population changes in cardiovascular risks can have high value when measured in terms of willingness-to-pay for mortality risk reductions ("the value of statistical life"). Studies which have included cardiovascular risks have found they dominate the value of IQ-related health impacts. Finally, although other human health impacts of mercury have been suggested – such as genotoxic effects and carcinogenicity (US EPA (2011_[7])) – the evidence is not considered strong and no study currently has included them as endpoints.

Lastly, regarding environmental impacts, US EPA (2011_[7]) provides a relatively recent summary. It states that, although numerous studies have been undertaken, many of the resulting data are anecdotal in nature, and although the study of the environmental effect of mercury exposures is growing, it is still incomplete. Much of the research has been carried out in laboratory settings rather than in natural ones, and hence conclusions about overarching ecosystem health and population effects are “difficult to make at this time”. US EPA (2011_[7]) cites references to reproductive deficits in fish, effects on birds, including reduced foraging and nest-sitting ability, physiological impacts and lower fertility and acute mortality in some fish-eating mammals like otters. Although, through resource-harvest models, adverse outcomes at the individual level could actually generate better outcomes at the population level (e.g. through reduced competition for food), US EPA (2011_[7]) report that, in relation to the common loon (a type of diver), for instance, “population-level effects negatively impacting population viability occur in parts of Maine and New Hampshire, and potentially in broad areas of the loon’s range.” However, US EPA (2011_[7]) concludes as follows:

“The studies cited here provide a glimpse of the scope of mercury effects on wildlife, particularly reproductive and survival effects, at current exposure levels. These effects range across species from fish to mammals and spatially across a wide area of the United States. The literature is far from complete however. Much more research is required to establish a link between the ecological effects on wildlife and the effect on ecosystem services (services that the environment provides to people) such as recreational fishing, bird watching and wildlife viewing. EPA is not, however, currently able to quantify or monetize the benefits of reducing mercury exposures affecting provision of ecosystem services adversely affected by mercury deposition.”

3. Existing assessments of the benefits of regulating mercury

The studies identified and reviewed for this paper can broadly be classified under four headings:

- ***Impact-pathway analysis***

These studies have a more or less explicit modelling of the processes affecting exposure, from emissions, through deposition, to uptake by and consumption of fish populations, generating methylmercury exposure in unborn children and potentially in the general population. These studies also tend to have a more explicit dynamic aspect, and treatment of the timing of changes in emissions, exposure and ultimately impacts. In this way, they can estimate the expected benefits of possible changes in emissions. The studies by the US EPA (2005_[6]), (2005_[22]), (2006_[23]) and (2011_[7]) are the most detailed but have a scope limited to the United States. The study by Rice and Hammitt (2005_[10]) was undertaken at around the same time but has some important differences, particularly in the inclusion of cardiovascular outcomes (updated by Rice et al. (2010_[24])). Gayer and Hahn (2006_[25]) is another early assessment which employs an unusual approach to valuing child IQ impacts. All of these studies consider the effects of reducing mercury

emissions from coal-fired power generation in the United States. Spadaro and Rabl (2008_[11]) employ some simplifications but produce apparently the first global impact-pathway assessment of the costs of mercury emissions. Giang and Selin (2016_[26]) also takes a wider perspective, considering the benefits to the United States of its own and the rest of the World's actions under the Minamata Convention. Shih and Tseng (2015_[27]) is one of the few impact-pathway studies undertaken outside of the United States context, and considers the benefits of a renewable energy policy in Chinese Taipei.

- ***Damage assessment***

These studies tend to be based on evidence of exposure at a particular point in time, without an explicit consideration of how that exposure was generated, although some ex post attribution to particular sources might be made. They also tend to lack an explicit consideration of timing issues, such that the damage estimates they produce can only be seen as describing the potential benefits of emission reductions. The “seminal” work of this type was by Trasande et al. (2005_[28]) and (2006_[29]), and their general approach was adopted in subsequent studies by Hylander and Goodsite (2006_[30]) for Greenland, Pichery et al. (2012_[31]) for France and Bellanger et al. (2013_[32]) for the European Union.

- ***Economic valuation***

Economic valuation studies focus on the specific impacts of exposure to mercury and the valuation of those impacts, and tend not to be “attached” to other steps in the impact pathway. Only a very small number of valuation studies have been identified in the context of mercury pollution and exposure specifically. These include Steckling et al. (2014_[8]) and (2017_[9]), who consider the quality-of-life impacts of mercury poisoning linked to ASGM, measured using the World Bank and World Health Organization concept of disability-adjusted life years (DALYs) (cf. Murray and Lopez (1996_[33])). Hagen et al. (1999_[34]) undertake possibly the only stated preference study of the value of the human health and environmental benefits of reducing mercury emissions (in Minnesota and the mid-west of the United States).

- ***Regulatory impact assessment***

Studies under this heading are assessment of the impacts of specific proposals to regulate the use of, emissions, of exposures to (etc.) mercury and mercury compounds. They tend to adapt existing estimates of damage costs for comparison against estimates of the costs of implementing the proposals. Identified regulatory impact assessments include those conducted by the European Chemicals Agency (ECHA) on proposed restrictions on the use of mercury in measuring devices (ECHA, 2011_[3]) and on five phenyl mercury compounds (ECHA, 2011_[2]), a proposed regulation on mercury in various products in Canada (Canada Gazette, 2014_[1]), and the costs and benefits of complying with the Minamata Convention in Australia (Marsden Jacob Associates, 2015_[4]).

The classification of studies under these different headings is not exact, and there are overlaps. For instance, the Steckling et al. (2014_[8]) and (2017_[9]) valuation studies include estimations of the prevalence of mercury poisoning to permit a calculation of the damage costs associated with this group of illnesses. However, their primary interest here is in terms of the value of individual health impacts and how those values are generated, so they are classified as valuation studies for this review. Further, the Rice and Hammitt (2005_[10]) study is classified here as an impact-pathway analysis, but does not (apparently) have an explicit treatment of the time it is likely to take for changes in emissions to translate into changes in health impacts. However, it does have a detailed consideration of

the pathways from emission to exposure, and hence is more appropriately treated under this heading than as a damage assessment.

The next section provides a more detailed description of the identified studies.

4. Review

This section provides a more detailed description of the studies outlined above. The intention is to convey an understanding of the main methodological features of the principal studies, the factors affecting the results they generate, and how they vary across studies. As already indicated, the literature is led by a small number of relatively detailed studies, which provide the basis for other, simpler pieces. Hence, more detail is provided on the former studies than the latter. The descriptions include a summary of the uses covered in the analysis (if applicable), the main endpoints of concern, and the approach to quantification and valuation. Significant issues, gaps, assumptions, etc., are highlighted where appropriate. In some cases, studies are compared against one another so that differences can be better understood. The results of the review and comparison are drawn together in the next section.

The review follows the natural development of the literature. The study by the US EPA (2005_[6]) was the first detailed impact-pathway study, and the one by Trasande et al. (2005_[28]) was the first important damage assessment study and these are considered first. The Trasande et al. (2005_[28]) study also inspired a number of applications, which are briefly described alongside, as well as an extension of the US EPA (2005_[22]) and (2006_[23]). Rice and Hammitt (2005_[10]) is the other “seminal” paper which appeared at around the same time and which is also considered in some detail. Spadaro and Rabl (2008_[11]) is the next important study and the first to take a “global” perspective. The Shih and Tseng (2015_[27]) study of Chinese Taipei has features of both Rice and Hammitt (2005_[10]) and Spadaro and Rabl (2008_[11]). The Giang and Selin (2016_[26]) paper also has a global scope from the perspective solely of the United States, and is briefly compared with the updated US EPA (2011_[7]) study. The Gayer and Hahn (2006_[25]) and Hagen et al. (1999_[34]) studies are considered together because they both take a “willingness-to-pay” approach to valuation (although the former is an impact-pathway study, and the latter is a valuation study). The papers by Steckling et al. (2014_[8]) and (2017_[9]) are the only studies of the impacts of chronic mercury poisoning in the context of ASGM. The review finishes with a consideration of four impact assessments of mercury regulations.

4.1. US EPA (2005)

The US EPA (2005_[6]), (2005_[22]) and (2006_[23]) provided a highly detailed analysis of the impacts of reducing mercury emissions from power stations under the Clean Air Mercury Rule (CAMR). The first report (US EPA, 2005_[6]) limited its scope to the consumption of self-caught freshwater fish by recreational anglers. The second report (US EPA, 2005_[22]) extended this initial analysis to cover all additional sources of fish (marine, estuarine and aquaculture) as well as the general population. The final report in the series (US EPA,

2006_[23]) provided a minor update to the dose-response function used to generate the US EPA (2005_[22]) general population figure. It is worth providing a reasonably detailed description of this work as it represents something of a benchmark against which other impact-pathway analyses can be judged, as well as other studies which do not use this general approach. It is also helpful to understand how the results obtained from a relatively sophisticated modelling approach compare with those of simpler ones.

The US EPA (2005_[6]) impact assessment comprised a number of steps. Geographical depositions were modelled for each of the emissions scenarios, and differences in depositions between scenarios were translated into changes in fish mercury concentrations. For each location, the resulting change in fish mercury concentration was translated into a change in mercury intake for the relevant human population at risk and thence into changes in maternal hair mercury. The resulting changes in IQ decrements and earnings losses were then calculated and aggregated over the population at risk. These values were discounted to reflect the adjustment time between changes in deposition and the mercury content of fish. The rest of this section provides a summary of each of these steps.

US EPA (2005_[6]) used the Community Multiscale Air Quality (CMAQ) model to predict the levels of mercury deposition for a 2001 base-year and a 2020 baseline reflecting co-control of mercury from implementation of the assumed baseline Clean Air Interstate Rule (CAIR) as well as two control options for CAMR. CMAQ accounts for spatial and temporal variations as well as differences in the reactivity of mercury emissions. Model simulations were performed based on plant-specific emissions of mercury by species.

2001 and 2020 emissions inventories were developed for input into the air quality modelling. Almost 115 (United States) tons of mercury were emitted across all United States sources in 2001, 48.6 tons from power plants (with almost 21 tons of that being of the most readily deposited form of mercury, reactive gaseous mercury, RGM). Total baseline mercury emissions in 2020 were forecast to be roughly 87 tons. CAIR co-control was estimated to lead to a reduction in power plant RGM emissions from 20.58 tons in 2001 to only 7.87 tons in 2020. Reductions in mercury emissions associated with the CAMR Control Option 1 in 2020 were forecast to be approximately 10 tons (14 tons under Option 2); RGM emissions were expected to fall by 1.3 tons and 2.16 tons with CAMR 1 and 2 respectively. Thus, CAIR was forecast to generate large decreases in RGM emissions from power plants (through the implementation of scrubber control technology). Most of the mercury emissions reductions from CAMR, however, were expected to be in the form of elemental mercury (Hg₀), which is not readily deposited, but enters the global pool of mercury, thus resulting in a reduction in mercury transport to the rest of the world.

US EPA (2005_[6]) reviewed available information and concluded that a large majority of the commercial fish consumed in the United States is either imported from foreign sources, or caught domestically 3-200 miles offshore. These sources were considered unlikely to be impacted by the control of utilities from the CAMR rule. There were also considered to be major limitations to the ability to model how changes in mercury deposition would affect fish tissue concentrations for the marine and commercial fish consumption pathways. However, methylmercury concentrations from freshwater sources were considered likely to be affected by the control of US electric utilities. Therefore the quantified benefit analysis evaluated the benefits of improved health from reduced exposure to methylmercury from recreational freshwater fishing activities only. This

limitation of scope was subsequently relaxed in US EPA (2005_[22]) and also differs from other impact-pathway and damage valuation studies.

Forecast changes in mercury deposition were translated into changes in fish mercury concentrations using the US EPA's Mercury Maps (MMaps) approach, and mercury fish concentration data from two national surveys. Following MMaps, it is assumed that, over the long term, fish mercury concentrations decline proportionally to declines in atmospheric loading to a waterbody. MMaps is essentially a static (steady-state) approach which does not incorporate time lags between reductions in mercury deposition and reduction in methylmercury concentrations in fish. US EPA (2005_[6]) case study modelling of ecosystem response times suggested that response times between five and 30 years were "most likely appropriate", although some systems could take more than 50-100 years to reach steady state. As a result, US EPA (2005_[6]) assumed a lag of either 10 or 20 years between changes in depositions and changes in methylmercury concentrations, which meant that estimated health benefits were discounted by between 10 or 20 years at 3% per year to generate present value figures. Other lag assumptions were used for sensitivity.

The two sources of fish mercury concentrations used by US EPA (2005_[6]) were as follows. First, the National Listing of Fish and Wildlife Advisories (NLFA) included (at the time) more than 91 500 samples of fish tissue contaminant data collected from over 10 700 locations nationwide from 1967 until 2003. Second, the EPA's National Lake Fish Tissue Survey (NLFTS), conducted in 1999-2003, sampled fish tissue from 500 randomly selected lakes and reservoirs across the United States (from the estimated 270 000 lakes and reservoirs in the lower 48 states). Despite their apparent detail, when combined, the two datasets gave, on average, one sample location per 1 000 km². In addition, both surveys had weaknesses. For instance, NLFA samples showed variations in sampling over time and across water bodies, which meant that it was not considered possible to identify temporal trends in the dataset.

The next step in the US EPA (2005_[6]) impact-pathway covered the consumption of fish caught in the course of recreational angling, and hence the ingestion of and exposure to methylmercury. Two models of freshwater angling behaviour were used, based on two sources of angling activity data, the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (NSFHWR), and the National Survey of Recreation and the Environment (NSRE). The first represented a "push" model in that it first identified where recreational anglers lived and then modelled their fishing trips out to different distance rings from there. Fish tissue samples were averaged within each ring to provide exposure levels for anglers in each ring. The size of the exposed population was estimated from the number of women of child-bearing age in each geographical "block", and state-level fertility and freshwater angling participation rates. The second model represented a "pull" model in that it identified where anglers fished, rather than where they lived. The total number of days' angling at each site was estimated, and the number of "angler-year equivalents" calculated on the basis of NSFHWR-reported fishing days. These "angler equivalents" were allowed to be males in child-bearing households or child-bearing women themselves, thereby resulting in a larger estimate (586 000) of the population exposed than the "push" model (434 000). These population estimates were then attached to estimated mean fish mercury concentrations for each fishing location or residential "ring" and combined with fish consumption data to produce average daily mercury ingestion.

As with all other studies considered in this review which assess mercury impacts in terms of lost IQ, the dose-response function used by US EPA (2005_[6]) was based on the epidemiological studies undertaken in New Zealand, the Faroe Islands and the Seychelles. The version used in this study was that of Ryan (2005_[18]), who undertook the first integration of the data from the three studies to produce a single estimate of the IQ dose-response relationship, in this case -0.13 IQ points per ppm mercury in maternal hair. The subsequent analysis was based on a linear functional form with no effect threshold assumed.

The monetary value of IQ losses was assessed in terms of foregone future earnings. The results of Salkever (1995_[21]), developed in the context of childhood lead exposure, were used to estimate the effects of IQ loss on expected future earnings (mean 2.38% reduction per IQ point, being 1.93% for men, and 3.23% for women.) and years of education (0.01 year decrease per IQ point). Expected future earnings were estimated from the 1992 US Current Population Survey at just under USD 370 000 (discounted at 3%). The cost of a year's schooling was estimated in terms of school expenditure per student and one year's average income (foregone through being at school). After updating to 1999 prices, the cost per IQ point lost comes to USD 8 800.

Regarding cardiovascular disease, US EPA (2005_[6]) conducted a critical review of the available literature at the time and determined that, while some studies showed that the effect might exist, the evidence was insufficient to support including cardiovascular effects in the benefit analysis.

After implementing the steps just described, the damage associated with the 2001 base-case was estimated at between USD 239 million ("push" model) and USD 354 million ("pull" model). With the "push" model, and relative to the base-case in 2020 (the 2001 base-case with forecast 2020 demographics), the introduction of CAIR was estimated to generate annual benefits of between USD 10.5 million (50-year lag) and USD 28.1 million (5-year lag), with a best-estimate of USD 20.5 million based on a 20-year lag. With the "pull" model, the introduction of CAIR was estimated to generate annual benefits 70-80% higher. Mercury emissions reductions under CAIR were estimated at around 28 tons relative to 2001, implying per kg values of USD 817-1 429 ("push" vs. "pull", 20-year lag).

Reductions in mercury emissions associated with the CAMR Control Option 1 were forecast to be approximately 10 tons from total emissions in 2020 with CAIR. Given estimates of benefits of between USD 1.7 million and USD 2.5 million per year, this implies benefits of USD 198-288 per kg of mercury emitted. Under CAMR Control Option 2, additional reductions of just under 4 tons and benefits relative to 2020 CAIR of USD 2.5-3.8 million imply unit benefits of USD 206-307 per kg of mercury emitted.

These figures appear low compared with other studies (see below), and clearly show a disparity between CAIR and CAMR benefits. One explanation for this could relate to speciation. The CMAQ modelling indicated that CAIR would generate a very large decrease in readily-deposited RGM emissions, whereas most of the mercury emissions reductions from CAMR were expected to be in the form of not-readily deposited elemental mercury. Thus, CAIR emission reductions would be expected to generate relatively more benefits domestically, compared with CAMR which would tend to generate relatively more benefits for the rest of the World. This suggests that it is important to take account of the variation in speciation when considering the impact of mercury emissions from different sources (see the discussion of Spadaro and Rabl (2008_[11]) below).

The subsequent reports by US EPA, (2005_[221]) and (2006_[231]), were produced at least partly in response to work by other commentators, particularly Trasande et al. (2005_[281]) and (2006_[291]). It therefore makes sense to consider these studies first.

4.2. Trasande et al. (2005) and (2006), Hylander and Goodsite (2006), Pichery et al. (2012) and Bellanger et al. (2013)

Trasande et al. (2005_[281]) provided one of the earliest estimates of damage resulting from IQ loss attributed to emissions of anthropogenic mercury. Trasande et al. (2006_[291]) modified the original analysis slightly by adding in the effects on the prevalence of “mental retardation” (defined as IQ below 70) as well as correcting the value used for the slope of the mercury-IQ dose-response function. The studies were undertaken in the context of a proposed slowing of the introduction of limits on mercury emissions from coal-fired power stations in the United States – the so-called *Clear Skies Act* and the subsequent “Utility Mercury Reductions Rule”. The analyses of both proposals had failed, according to Trasande et al. (2005_[281]), to recognise and quantify the health impacts of the proposals. Their study was an attempt to estimate the health costs associated with these emissions and thereby the potential costs associated with these new proposals. However, unlike US EPA (2005_[6]), they did not attempt to establish a formal impact-pathway between emissions and health (and economic) impacts, but rather estimated current damage on the basis of current total exposure to methylmercury, and then attributed a portion of that damage to mercury emissions from coal-fired power stations. As such, therefore, the study does not trace out “what will happen” if mercury emissions change, but could be said to give an indication of the “potential” or “long-run” benefits of emissions control.

The principal components of Trasande et al.’s (2005_[281]) analysis were as follows. The exposed population was based on data from the National Health and Nutrition Examination Survey (NHANES) (Mahaffey et al. (2004_[351])), in particular women of child-bearing age with mercury blood concentrations greater than 3.5 µg per litre. A no-adverse effect threshold of 3.41 µg per litre had been proposed by the (National Research Council, 2000_[12]), but to be conservative, Trasande et al. (2005_[281]) assumed no adverse effects below 4.84 µg per litre. The dose-response function used was based on the study from the Faroe Islands by Grandjean et al. (1999_[361]) and Budtz-Jorgensen et al. (2002_[371]), whose analysis (according to Trasande et al. (2005_[281])) suggested a logarithmic functional form in which a doubling of mercury concentration produced a decrement of approximately 10% of a standard deviation, or 1.5 IQ points. (Different slopes and a linear functional form were used as sensitivity analyses.) As with US EPA (2005_[6]), (2005_[221]) and (2006_[231]), the work of Salkever (1995_[211]) was used to estimate the effect of one IQ point decrement on earnings. The values for lifetime earnings were drawn from the work of Max et al. (2004_[381]), and included an adjustment for wage supplements (e.g. fringe benefits and employer contribution to insurance benefits) and the imputed value of household production. This resulted in lifetime earnings estimates of just over USD 1 million for males and USD 0.76 million for females (significantly higher than the value used by US EPA (2005_[6]), which was based on wage earnings only). The implied costs per lost IQ point were just under USD 20 000 for males and just under USD 24 700 for females.

Combining all of these data together and applying them to the 2000 American birth cohort, and using their base-case assumptions, Trasande et al. (2005_[281]) calculated that 405 881 children suffered IQ losses as a result of foetal methylmercury exposure of

between 0.76 and 3.21 points each. At the estimated value of an IQ point, this produced the estimate of the total cost of anthropogenic exposure to methylmercury of USD 8.7 billion. The range of estimates obtained by varying assumptions was USD 2.2 billion to USD 13.9 billion for the logarithmic model, and as wide as USD 7 billion to 43.8 billion for the linear model, although this latter range was subject to a one-order-of-magnitude downward correction as a result of Trasande et al.'s (2006_[29] update).

Trasande et al. (2005_[28]) then used the concept of the “environmental attributable fraction” to estimate the proportion of this cost which might have been caused by emissions from US coal-fired power stations. Because this is the first study reviewed which adopted this approach, it is worth explaining in some detail how this attribution was performed. Based on UNEP (2002_[39]), they first assumed that 70% of mercury released into the atmosphere is from anthropogenic sources. Then, based on various US EPA sources, they estimated that domestic sources were responsible for 60% of “anthropogenic mercury” deposited in the United States, which was then available to bioaccumulate in American fish. They further estimated that 42% of fish consumed in the United States was imported, and that 2% of the mercury in this fish was of US origin; the remaining 58% of fish consumed in the United States was therefore of domestic origin with a mercury content which was 60% domestic. They therefore applied a 36% factor (the weighted average of American sources of mercury content in fish, or $(0.6 \times 0.58) + (0.02 \times 0.42)$) to identify the proportion of economic costs of anthropogenic methylmercury exposure attributable to American sources. Finally, US EPA data were used which indicated that 41% of US anthropogenic emissions were attributable to the electricity generation sector.

The results were, therefore, that out of the total USD 8.7 billion cost in 2000 attributable to anthropogenic methylmercury, USD 3.1 billion were attributed to United States sources of mercury, of which USD 1.3 billion were linked to emissions from coal-fired power stations.

The subsequent paper by Trasande et al. (2006_[29]) added in a valuation of the costs associated with so-called “mental retardation” (MR), defined as having an IQ lower than 70 points. MR has been proposed to be associated with additional costs (such as schooling, medical care and social care (Grosse et al., (2002_[40]); Honeycutt et al. (2000_[41])), and because exposure to methylmercury is said to shift downwards the distribution of IQ in an exposed population, it could increase the number of children in the population with MR, and hence increases the associated social costs.² However, after adjusting for uncertainties and attribution, Trasande et al. (2006_[29]) estimated that US power plants accounted for only 231 MR cases per year (with a range of 9-394), at a cost of USD 2-43 million per year, which was not high enough to change the headline cost estimates reported by Trasande et al. (2005_[28]).

A number of authors use the Trasande et al. (2005_[28]) and (2006_[29]) methodology as the basis for their own work. Hylander and Goodsite (2006_[30]) provided an estimate of the costs of IQ loss due to foetal mercury exposure in Greenland. Using a sample of mercury cord blood measurements obtained from 178 new-born children in western Greenland (Bjerregaard and Hansen, 2000_[42]), and applying a modelled distribution to the

² IQ is a relative concept, of course, with the median always set at 100 by definition, so the IQ distribution cannot be “shifted” as such – a more correct way of expressing it might be to say that the IQ level which is used to define MR is raised as a result of methylmercury exposure.

967 children born in 1998-1999, they estimated that only 27% of children were born with mercury cord blood below the US EPA RfD threshold. The 703 affected children were estimated to be associated with lifetime costs of USD 59 million, or around USD 84 000 each (valued according to US earnings due to an absence of Greenland-specific data).

Pichery et al. (2012_[31]) evaluated the economic impacts of neurotoxicity associated with prenatal methylmercury exposure in France. Their estimations were carried out assuming a linear and a logarithmic relationship between mercury exposure and IQ losses, for three possible thresholds, based on the Joint FAO/WHO Expert Committee on Food Additives (JECFA, 2006_[43]) (2.5 µg/g mercury in maternal hair), the US EPA methylmercury reference dose (1µg/g), and the US National Research Council (2000_[12]) (0.58µg/g). They used data on hair mercury concentrations in French women of child-bearing age from a national sample of 126 women, a sample from Brittany and a sample from the Loire-Atlantique coastal district, and applied them to the 2008 national birth cohort. The annual economic impact (or benefits of mercury reduction) were estimated on the basis of the value of an IQ point of EUR 17 363, estimated for France in 2008 by Pichery et al. (2011_[44]), which in turn was an update (for prices levels but not apparently for purchasing power parity) of Gould (2009_[45]) (who estimated a 2006 United States figure based on Schwartz (1994_[20]), Salkever (1995_[21]) and Nevin et al. (2008_[46])).

The Pichery et al. (2012_[31]) results clearly depend on the assumed distribution of mercury exposure (which in turn depends on which sample is used) and the assumed threshold, as well as the functional form of the mercury-IQ relationship. The authors reported different threshold figures only for the linear model, with the 0.58µg/g hair threshold only being reported for the log model. Comparing the linear and log models produced the following estimates:

Table 4.1. Estimated costs of mercury exposure in France

Sample and assumed mercury distribution

Model	National	Brittany	Loire-Atlantique
Linear	EUR 1.62bn	EUR 3.02bn	EUR 2.51bn
Log	EUR 5.46bn	EUR 9.13bn	EUR 8.17bn

Thus it can be seen that the log model produces much higher values than the linear model (as demonstrated in the Trasande et al. (2005_[28]) and (2006_[29]) estimates and the Griffiths et al. (2007_[47]) review (see below)). The higher values based on the Brittany and Loire-Atlantique samples reflect the higher mean mercury concentrations found in these regions, where more fish is consumed.

Finally, Bellanger et al. (2013_[32]) essentially repeated the Pichery et al. (2012_[31]) analysis with mercury hair data relating to the EU. Distributions of hair-mercury concentrations among women of reproductive age were obtained from the EU DEMOCOPHES project (1 875 subjects in 17 countries) and data obtained from literature search (6 820 subjects from eight countries). For missing EU member states, methylmercury exposures were assumed to be the same as a neighbouring country. IQ losses and economic costs associated with methylmercury exposures above the threshold and using the linear model were found to be greatest in the largest countries with the highest proportions of subjects with exposures above the threshold. The total cost of methylmercury exposure was the highest for Spain and the lowest for Hungary. On a per capita basis, the calculated costs were highest in the Faroe Islands and the southern countries, Spain, Greece, Portugal,

Italy and Croatia. With an average cost of EUR 13 579 per IQ point, the total economic costs within the EU were estimated to exceed EUR 9 billion per year. (Bellanger et al. (2013_[32]) also made what they called an adjustment for productivity which was equivalent to undertaking a willingness-to-pay benefits transfer based on GDP ratios. The result was costs somewhat lower for several countries, and an EU total slightly less than EUR 8 billion per year.) For comparison, they also estimated IQ losses and economic losses using the log dose-response function. Due to the steeper curve shape at exposures close to the threshold, the estimated costs were about four times higher, at about EUR 39 billion for the EU (EUR 33 billion after income adjustment).

4.3. US EPA (2005, 2006), Griffiths et al. (2007)

Partly in response to Trasande et al. (2005_[28]), US EPA (2005_[22]) reviewed various data and analyses regarding the likely contribution of US power station emissions to overall mercury consumption via exposure routes other than non-recreational freshwater fishing. The conclusion was that these other routes did not constitute a significant source of mercury exposure and that their original results (US EPA (2005_[6])) were reasonable. Nevertheless, they undertook a “bounding analysis” including these other routes, for both recreational anglers and the general public, of total exposures from US power plants. The scope was therefore comparable with the power station attributable fraction calculation by Trasande et al. (2005_[28]) and (2006_[29]). The dose-response function and economic valuation components of the analysis were largely the same as before. For freshwater systems, an ecosystem response lag of 15 years was assumed, and for marine systems 30 years. The population at risk was assumed to be all births (around four million per year), although those to recreational angler households (about 830 000 per year) were treated separately.

Using assumptions to produce “a clear upper bound estimate”, US EPA (2005_[22]) calculated aggregate economic loss of USD 41.5 million from the consumption of self-caught freshwater fish, and USD 8 million from marine fish consumption (USD 1.6 million to recreational anglers, the remainder to the general public). To avoid underestimation, freshwater, estuarine and farmed fish consumption were assumed to be similar to (i.e. as contaminated as) self-caught recreational freshwater fish consumption. The resulting total value for the costs of IQ losses across the entire population from all US power station emissions in 2020 was USD 168 million, subsequently increased by US EPA (2006_[23]) to USD 210 million following a correction to the dose-response calculations by Ryan (2005_[18]).

US EPA (2005_[22]) and (2006_[23]) was at pains to emphasise that both of these values represented unrealistic upper bounds of the costs of power station mercury emissions, calculated solely to demonstrate that the conclusion that CAMR did not pass a benefit-cost test was robust to the most severe stress-testing. Nevertheless, translating the USD 210 million figure for reducing 2020 power plant emissions (34 tons) to zero into a per kg value gives USD 6 808, which, according to the US EPA (2005_[22]), must be an upper bound for the benefits of mercury reduction (p. 37).

Griffiths et al. (2007_[47]) provided a “reconciliation” of the estimates produced by Trasande et al. (2005_[28]) and (2006_[29]) and the USD 210 million value from US EPA (2006_[23]). They used Trasande et al.’s (2005_[28]) linear model for comparison purposes (although this model provided only sensitivity bounds in the Trasande et al. (2005_[28]) study), with a cord-maternal blood ratio of 1.7 and an effects threshold at a blood mercury level of ≥ 4.84 $\mu\text{g/l}$. Starting from Trasande et al.’s (2006_[29]) linear model, Griffiths et al.

(2007_[47]) generated estimates of the United States IQ costs of all anthropogenic mercury emissions of USD 33 billion (based on the original dose-response function slope) and USD 3 billion (using the (2006_[23]) corrected slope). The amounts due to US power station emissions were calculated at USD 5 billion and USD 480 million respectively. Using the US EPA (2006_[23]) assumptions and approaches, they estimated a figure of USD 370 million for all anthropogenic emissions, and USD 10 million for emissions from United States power stations. They highlighted the following principal differences between the Trasande et al. (2005_[28]) and US EPA (2006_[23]) approaches in generating these different values:

- Trasande et al.'s (2005_[28]) “environmentally attributable fraction” approach to allocating costs to emissions from US power plants, versus US EPA’s (2006_[23]) use of spatially detailed emissions and deposition models. This was estimated to result in the US EPA’s (2006_[23]) estimate being 72% lower than Trasande et al.’s (2005_[28]);
- Trasande et al. (2006_[29]) used a dose-response function with a slope of -0.093 IQ points per ppb increase of mercury in cord blood (corrected from -0.93), significantly higher than the US EPA’s (2006_[23]) -0.032 (equivalent to -0.016 ppm in maternal hair). This was responsible for a 66% difference in the estimates;
- The value of lifetime earnings assumed by US EPA (2006_[23]) of around USD 0.47 million (2000 prices), compared with Trasande et al.’s (2005_[28]) figures of USD 0.76 million for women and over USD 1 million for men. The latter incorporated (via Max et al. (2004_[38])) wage supplements, which US EPA (2006_[23]) did not. Excluding them reduced the US EPA (2006_[23]) estimate by 46% compared with Trasande et al. (2006_[29]);
- Trasande et al. (2005_[28]) assumed that 58% of United States fish consumption was affected by domestic mercury deposition, and 60% of United States mercury deposition was due to domestic sources. The comparable figures in US EPA (2006_[23]) were 30% and 16% respectively, generating a 46% reduction;
- Trasande et al. (2005_[28]) assumed instantaneous changes in fish mercury concentrations following changes in mercury emissions and depositions, whereas US EPA (2006_[23]) assumed ecosystem response lags of 30 (marine) and 15 (all others) years, meaning the latter’s estimates were subject to 3% per year discounting, and hence were 36% lower.

These findings are summarised in the table below.

Table 4.2. Griffiths et al. reconciliation of US EPA and Trasande estimates of the US costs of mercury emissions

Assumption/approach	Percentage change in Trasande results from adopting an US EPA assumption or approach
Detailed emissions and deposition modelling	-72%
Dose-response function	-66%
Value of life-time earnings	+46%
Domestic mercury emissions effect on US mercury intake	-46%
Ecosystem lag	-36%

The additional factor which Griffiths et al. (2007_[47]) did not account for (apparently since to do so would complicate the analysis) was the functional form of the dose-response function. As seen above, the logarithmic functional form used by Trasande et al. (2005_[28]) has generated much higher values in studies than the linear one, due to the slope

it implies around the threshold exposure level. The choice between the two seems largely a matter of whether the author prefers an estimation which better fits the data (Trasande et al. (2005_[28]) claim that the logarithmic model provides a better fit for the data analysed by Grandjean et al. (1999_[36])) or which better fits with biological expectation (Nedellec and Rabl (2016_[48]) argue that mercury is a toxin with no beneficial effects (unlike some metals), making the “ceiling” to adverse effects which the logarithmic form implies implausible).

This particular choice could be seen as a matter of “academic judgment”, as could the choice of the slope of the dose-response function. The differences due to modelling of deposition, fish mercury consumption (exposure) and life-time earnings seem less a matter of judgement as analytical rigour. For instance, it would be expected that Trasande et al. (2005_[28]) would have used a disaggregated emissions-deposition model like the US EPA’s CMAQ if they had access to one, but in its absence, they used the comparative “rule of thumb” which was the attributable fraction approach, accepting the errors that this was likely to introduce.

The question of whether a discount factor should be introduced to account for ecosystem lags is rather different. The evidence reviewed by US EPA (2005_[22]) and others suggests that no ecosystem would respond instantaneously to changes in mercury depositions, so it is not as much whether there should be an ecosystem lag but rather how long it should be. However, when calculating the damage costs of existing exposures to mercury, ecosystem lags are not relevant since the “deposition” steps of the impact-pathway are assumed already to have happened. It is only when making an attribution to a cause (previous mercury emissions), or a forecast of the benefits of a proposed policy (reducing emissions in future), that how long “causes” take to become “effects” becomes important.

Therefore, Trasande et al.’s (2005_[28]) calculation of the health costs of mercury exposures is not incorrect as such (ignoring any other errors already discussed), but attributing those costs to the most recent year for which data were available, as Trasande et al. (2005_[28]) did, most likely would be, since currently observed health impacts are actually the result of emissions and depositions many years previously.³ Similarly, damage estimates give only an estimate of the potential, or even theoretical, benefits of emissions reductions, because they effectively represent a current state (current health impacts from previous exposures), not the effects of changing states (changes in health impacts from changing exposures). Most obviously for the purposes of cost-benefit analysis, they do not say how long changes in health impacts will take to manifest themselves – assuming they would – if mercury emissions were to be changed. Trasande et al. (2005_[28]) did not present an estimate of the damage costs per kg of mercury emitted, but it would be of questionable value anyway. A rough estimate can be calculated on the basis of emissions figures presented by US EPA (2005_[6]) for 2001. As already seen, US EPA (2005_[6]) estimated that almost 115 (US) tons of mercury were emitted across all United States sources in 2001, 48.6 tons from power plants. Using Trasande et al. (2005_[28]) estimates of USD 3.1 billion attributable to United States sources of mercury, and USD 1.3 billion linked to emissions from coal-fired power stations, this gives estimates of cost per kg of mercury emitted of around USD 29 486, between one and two orders of magnitude higher

³ Trasande et al. (2005_[28]) used deposition data from 1995 (“the most recent year for which federal data on the relative deposition of mercury from American and other global sources are available” and 1999 (“the most recent year for which data on American mercury emissions are available”).

than estimated from the US EPA (2005_[6]) results. This difference is similar to that found by Griffiths et al. (2007_[47]) in their comparison of the two studies.

4.4. Rice and Hammitt (2005)

Rice and Hammitt (2005_[10]) is another complex impact-pathway study which is important in the literature since it has been cited in several other studies which have used it as a source for assumptions and/or results (e.g. Spadaro and Rabl (2008_[11]), Canada Gazette (2014_[1]), ECHA (2011_[2]) and (2011_[3])). As with the US EPA (2005_[6]) and Trasande et al. (2005_[28]) studies, this study estimated the health benefits of reducing mercury emissions from coal-fired power plants in the United States, as part of the Clear Skies Initiative (CSI). It was prepared for the Northeast States for Coordinated Air Use Management (NESCAUM). It is similar to the US EPA (2005_[6]) study in many respects, but differs in some important ways which will be briefly described here.

First, the model identified deposition changes (based on early US EPA modelling), across eight geographic regions around the North American continent (including three ocean regions), which were translated into changes in the methylmercury content of both freshwater and marine fish. Marine fish were excluded from the original US EPA (2005_[6]) work, and included later (US EPA, 2005_[22]) only as an upper-bound sensitivity analysis and to demonstrate that marine fish were unlikely to be a significant source of changes in mercury exposure for US fish consumers. As with the US EPA's (2005_[6]) use of the MMaps model, changes in deposition were assumed to result in a proportional change in methylmercury concentrations with no formal modelling of ecosystem response times (i.e., changes in deposition effectively result in instantaneous changes in fish mercury). However, US EPA (2005_[6]) and (2005_[22]) made an ex post allowance for ecosystem response times by discounting health benefits by 15 years (freshwater systems) or 30 years (marine systems), i.e. it was assumed ecosystems would take 15 or 30 years to respond to changes in deposition, and hence that health benefits would only accrue 15 or 30 years in the future; no such adjustment was made by Rice and Hammitt (2005_[10]).

Second, Rice and Hammitt (2005_[10]) used national surveys to estimate the numbers of marine and freshwater anglers, and obtained estimates of the recreational fish-consuming population by multiplying by a factor to account for the fact that anglers tend to share their catch. Recreational fish-consumers were assumed to consume a mix of fish from the region in which they were resident. This compares with the comparatively detailed spatial analysis undertaken by US EPA (2005_[6]) which modelled the relationship between recreational fishing at particular watersheds and the fish mercury concentrations in those watersheds.

Third, Rice and Hammitt (2005_[10]) used a dose-response function for IQ losses with a slope of -0.6 IQ points per 1 ppm increase in hair mercury. This was based on unpublished work by Cohen et al. (not referenced by Rice and Hammitt (2005_[10]) but apparently Cohen et al. (2005_[49])). This was much higher than the figure used by US EPA (2005_[6]) of -0.13 (later corrected to -0.16 (US EPA, 2006_[23]), from Ryan (2005_[18])). Rice and Hammitt (2005_[10]) also calculated IQ losses assuming, first, no threshold, and, second, a threshold equal to the US EPA reference dose (RfD) of 0.1 µg/kg-day. As with similar studies (including US EPA (2005_[6])), Rice and Hammitt (2005_[10]) used the work of Salkever (1995_[21]) to estimate the impact of IQ on earnings. They applied the estimated impact of one IQ point (2.4% – as used by US EPA (2005_[6])) to the value of lifetime earnings calculated by Grosse (2003_[50]) (excluding the value of household production on the grounds that they considered there to be no evidence that

IQ decrements would affect this value significantly) of USD 0.69 million (2000 prices) – significantly higher than the USD 0.37 million used by US EPA (2005_[6]) – resulting in a cost per IQ point of USD 16 500. Like Trasande et al. (2006_[29]) but unlike US EPA (2005_[6]), Rice and Hammitt (2005_[10]) also included an estimate of costs associated with having a higher proportion of the population being “mentally handicapped” (IQ below 70), although (again like Trasande et al. (2006_[29])) this did not have a significant impact on the IQ point unit cost.

Unusually, Rice and Hammitt (2005_[10]) cited the work of Torrance et al. (1996_[51]) and Feeny et al. (2002_[52]) who developed a series of utility weights for cognitive decrements, but at the same time considered these weights too high to match the rather minor impacts reported in the New Zealand and Faroe Islands epidemiological studies. Hence, they assumed that the utility weight for the neurological effects associated with in utero methylmercury exposures was equal to an arbitrary 0.01 per year, implying an estimated lifetime loss of 0.77 quality-adjusted life-years (QALYs) for each child born to a mother with exposure above the effect threshold. QALY impacts were not assigned a monetary value in the final results.

Fourth, and most importantly, Rice and Hammitt (2005_[10]) included cardio-vascular effects as a health impact of mercury exposure. US EPA (2005_[6]) explicitly excluded these due to a claimed lack of consistent evidence, an issue acknowledged by Rice and Hammitt (2005_[10]).

Nevertheless, they calculated risk coefficients for acute myocardial infarction and all-cause mortality based on the relative risks estimated by Salonen et al. (1995_[53]). Applying these to estimated mercury ingestion and baseline risks of these events for those aged over 39 gave estimated total numbers of additional deaths and non-fatal heart attacks per year. Excess deaths were valued according to the US EPA’s (1997) value of statistical life of USD 5.9 million (updated to 2000 prices). Non-fatal heart attacks were valued in terms of lost earnings Grosse (2003_[50]) and additional health service (etc.) costs (Blake et al. (2003_[54]), as well as by applying a utility weight of 0.9 for each year of assumed remaining life (Blake et al. (2003_[54])).

Rice and Hammitt (2005_[10]) generated results for current emissions, projected 2010 and 2020 emissions, and two policy scenarios – 2010 projections including a 47% reduction in emissions from coal-fired power stations (Scenario 1), and 2020 projections including a 60% reduction in emissions from coal-fired power stations (Scenario 2). Results indicated that, for current (2000) and projected (non-policy) emissions, costs of IQ reductions were in the region of USD 19-20 billion per year assuming no effect threshold, or around USD 3 billion per year if a threshold was assumed. Existing and projected cardiovascular impacts (for which no threshold was assumed) were not presented by Rice and Hammitt (2005_[10]), and cannot readily be estimated from the information presented in their report.

In terms of the impacts of the two policy scenarios, Rice and Hammitt (2005_[10]) estimated IQ-related benefits for Scenario 1 of USD 193 million (assuming no threshold) or USD 75 million (with threshold). Scenario 2 was estimated to have IQ-related benefits of USD 288 million (without threshold) or USD 119 million (with). Regarding cardiovascular impacts, the benefits were estimated at USD 3.3 billion (Scenario 1) or USD 4.9 billion (Scenario 2). Translating these figures into USD per kg of mercury emissions gives the following:

Table 4.3. Rice and Hammitt (2005) estimates of the benefits of mercury emissions control

Impacts considered	Policy scenario 1	Policy scenario 2
IQ benefits only, threshold	USD 4 346 per kg	USD 4 954 per kg
IQ benefits only, no threshold	USD 11 193 per kg	USD 11 900 per kg
Cardiovascular benefits only, all consumers over 39	USD 189 644 per kg	USD 202 586 per kg
IQ benefits, no threshold, cardiovascular benefits, all consumers over 39	USD 200 836 per kg	USD 214 486 per kg

This demonstrate the extent to which cardiovascular effects can dominate, relative to IQ benefits, the results of studies which include them, in this case representing around 95% of the total benefits of mercury emissions reduction. Recall that no discounting for ecosystem lag effects was included in this study (estimated by Griffiths et al. (2007_[47]) to increase figures by about 50%), and that both the IQ dose-response function (four-times) and the lifetimes earnings (two-times) were higher than used by US EPA (2005_[6]).

4.5. Spadaro and Rabl (2008), Rice et al. (2010) and Nedellec and Rabl (2016)

The final “seminal” study considered in detail in this review is the one by Spadaro and Rabl (2008_[11]), which is notable in a number of respects. First, it is one of the few papers to provide an estimate of the costs of mercury exposure at a global level. Second, it used a relatively simple methodology to model the relationship between changes in emissions and changes in exposure. Third, the assumptions and results are used by a number of other studies considered in this review. Because the methodology is relatively simple, it can be explained here quite easily.

The basic starting point of the study was that mercury is a global pollutant and therefore that the benefits of emissions control (and other regulatory actions) should be evaluated at the global scale. In particular, the gaseous metallic component of mercury emissions is a global pollutant because its effective residence time in the atmosphere is long enough to allow uniform distribution around the globe. The other principal components of mercury emissions (reactive gaseous mercury (RGM) and particulate mercury), on the other hand, deposit locally and regionally. The speciation of mercury (the relative composition of the different forms) can vary across emission sources, which means that the extent to which those emissions represent “local” or “global” pollution also varies. However, Spadaro and Rabl (2008_[11]) argued that, even though the actual distribution of ambient total mercury might not be very uniform, the ingestion dose of methylmercury is much more uniform because of extensive international trade in fish. (They did not provide evidence to support this claim, however.)

Therefore, as an alternative to detailed modelling of the pathways determining global exposure to methylmercury as a result of anthropogenic mercury emissions, Spadaro and Rabl (2008_[11]) defined what they termed a “comprehensive transfer factor”, T_{av} :

$$T_{av} = D_{av}/E$$

where D_{av} is the global average ingestion dose of methylmercury per person (measured in terms of μg per year) and E is the global emission rate of mercury, measured in kg/year . Thus, Spadaro and Rabl (2008_[11]) assumed that, as emissions of mercury vary across the globe, so the ingestion of (exposure to) methylmercury varies in direct proportion. D_{av} was set at $2.4 \mu\text{g}$ per year, and global mercury emissions, E , were set at 6 000 tonnes per year, both figures sourced from UNEP (2002_[39]).

The exposure-response function used by Spadaro and Rabl (2008_[11]) was that developed by Axelrad et al. (2007_[19]) based on an integration of the New Zealand, Seychelles and Faroe Islands epidemiological studies. This assumed a linear functional form with a central estimate of the slope of 0.18 IQ points per ppm increase in mercury in maternal hair, translating into 0.036 IQ points per µg/day, once adjustment is made for the relationship between hair concentration, blood concentration and methylmercury ingestion. Spadaro and Rabl (2008_[11]) took no threshold as their base case, with the US EPA RfD as a threshold sensitivity. They used data for the United States from the US National Center for Health Statistics (2005_[55]) to estimate the proportion of the world population of mothers exposed to methylmercury above this threshold. This in turn permitted an estimate of the mean number of IQ points lost as a result of current estimated methylmercury exposures, equal to 0.020 IQ points per person (threshold) or 0.087 IQ points per person (no threshold).

For the cost associated with the loss of an IQ point, Spadaro and Rabl (2008_[11]) cited a number of North American studies (Rice and Hammitt (2005_[10]), Trasande et al. (2005_[28]), Griffiths et al. (2007_[47]), Muir and Zegarac (2001_[56]) and Grosse et al., (2002_[40])) to generate a simple mean figure of USD 18 000 per IQ point (2005 prices). Transfers across countries were made using the relevant GDP ratio in purchasing-power parity (PPP) terms.

Applying this figure to the estimates of the number of lost IQ points per capita gave a cost per person of current levels of methylmercury exposure of USD 78 (threshold) and USD 344 (no threshold). The marginal damage cost used the same assumptions and data with estimates of the number of births in a year, and applied a discount factor to reflect an assumed 15-year delay (at 3% per year) between a change in emissions and a change in impacts (Griffiths et al., (2007_[47])). The result was a mean global cost per kg mercury emitted of USD 1 500 (threshold) or USD 3 400 (no threshold).

Spadaro and Rabl (2008_[11]) usefully provided a comparative assessment of their results with those of Rice and Hammitt (2005_[10]). They described the two sets of estimates as “not very comparable” since Rice and Hammitt (2005_[10]) considered “only impacts within the United States”, whereas the Spadaro and Rabl (2008_[11]) estimate was a “simple global average”. They argued that, “due to local and regional variations in mercury speciation, dispersion, population, dietary habits, etc.”, geographically specific estimates such as those by Rice and Hammitt (2005_[10]) would be likely to generate very different results from those from more global approaches such as their own. In particular, they argued that the speciation for the power plants considered by Rice and Hammitt (2005_[10]) had a higher fraction of RGM and particulate mercury than the global emissions they assumed, and thus the Rice and Hammitt (2005_[10]) study models a higher fraction of mercury deposited in the United States where the cost of a lost IQ point is higher than the global average.

Spadaro and Rabl (2008_[11]) attempted to provide a reconciliation of the results of the two studies, by adjusting their global result to account for the scope and assumptions made by Rice and Hammitt (2005_[10]). First, they reduced their global result by 80% to account for the fact that Rice and Hammitt (2005_[10]) excluded the global impacts of emissions from US power plants. They then adjusted for differences in the assumed value of an IQ point (USD 16 500 in Rice and Hammitt (2005_[10]) compared with USD 18 000 in their own study), and the slopes of the exposure-response functions (0.6 IQ points per ppm mercury in maternal hair in Rice and Hammitt (2005_[10]), which translates into 0.12 IQ points per µg/day, compared with their own 0.0362 IQ points per µg/day (from Axelrad et al.

(2007_[19]). Spadaro and Rabl (2008_[11]) then multiplied their estimate by a factor of three, reflecting the larger than assumed fraction of RGM in the speciation of power plant emissions in the United States (although it is not clear how this adjustment factor was arrived at). They also stripped out from their figure the discount factor which reflects their assumed delay (15 years) between a change in emissions and a change in impacts. The result was estimates for the United States of USD 4 380 per kg (with threshold) and US 9 993 per kg (without threshold), which they suggested compared remarkably closely with the corresponding numbers from Rice and Hammitt (2005_[10]) (USD 4 300 per kg and US 11 200 per kg), given the radically different estimation approaches.

Rice et al. (2010_[24]) effectively provided an update of the earlier work by Rice and Hammitt (2005_[10]), which revised some assumptions, parameter values, etc. and attached probability weights to impacts depending on the strength of the evidence supporting their calculation. They adopted the Axelrad et al. (2007_[19]) value for the slope of the IQ-mercury dose-response function, adjusted to control for confounding factors (e.g. fish fatty acid impacts on health). They assigned probabilities to the existence of an IQ effect threshold and to the link between mercury intake and cardiovascular impacts (but not to the size of this effect). They further assumed that the impact of reduced mercury intake on heart attack risk takes between two and 10 years to materialise. They also switched the source of the relationship between IQ and earnings to the work of Heckman et al. (2006_[57]), assuming a direct impact on hourly earnings of between 0.6% and 1.2%. The value of statistical life was given a mean value USD 5.5 million.

Unfortunately, Rice et al. (2010_[24]) present their results in terms of median present value of the benefits of reducing per capita daily methylmercury exposures by 0.1µg, which is not simply translated into figures comparable with others considered in this review. However, they do provide a translation of the results for the United States of Spadaro and Rabl (2008_[11]), Gayer and Hahn (2006_[25]) and a version of the US EPA CAMR studies, as follows:⁴

Table 4.4. Present value of the benefits of reducing per capita daily methylmercury exposures by 0.1µg

	5 th percentile	USD 1.60 per 0.1g per capita
Rice et al. (2010 _[24])	50 th percentile	USD 7.30 per 0.1g per capita
	95 th percentile	USD 116 per 0.1g per capita
Gayer and Hahn (2006 _[25])	“Low”	USD 1.07 per 0.1g per capita
	“High”	USD 1.57 per 0.1g per capita
US EPA (2005 _[22]) and (2006 _[23])	5 th percentile	USD 1.10 per 0.1g per capita
	50 th percentile	USD 4.67 per 0.1g per capita
	95 th percentile	USD 7.67 per 0.1g per capita
Spadaro and Rabl (2008 _[11])	5 th percentile	USD 1.07 per 0.1g per capita
	50 th percentile	USD 5.33 per 0.1g per capita
	95 th percentile	USD 26.33 per 0.1g per capita

⁴ The link provided in the Rice et al. (2010_[24]) paper to the US EPA study in question no longer works, so it is not clear exactly which version is being referred to. The values presented for the US EPA study suggest a scope which must include fish consumption other than from recreational freshwater angling, which US EPA (2005_[22]) and (2006_[23]) described as an absolute upper bound.

None of these other studies includes a valuation of cardiovascular impacts, which presumably is the principal reason why the Rice et al. (2010_[24]) estimate is the highest of these four. The Gayer and Hahn (2006_[25]) figures are much lower than the others' median results, as this study uses a much lower value of IQ loss (USD 1 200 (“low”), USD 2 000 (“high”) than the others (see below).

Rice et al. (2010_[24]) noted that the value of cardiovascular benefits was likely to exceed the value of IQ benefits as long as it was judged that the probability that the estimate of the cardiovascular effect of mercury is real was greater than 10%. This reflects the relatively large size and value of the cardiovascular impact compared with IQ loss, and occurs despite what Rice et al. (2010_[24]) term “greater confidence in the scientific evidence” underlying the latter effect and the general “scientific consensus about the interpretation” of the mercury-IQ epidemiology. Note, however, that this conclusion relates only to the existence of the cardiovascular effect, not its magnitude.

In subsequent work, Nedellec and Rabl (2016_[48]) provided what is basically a simple update of the Spadaro and Rabl (2008_[11]) study, with an extension to include the costs of cardiovascular effects of mercury exposure. The updated figure for IQ loss is given as USD 5 626 per kg (2013), with no threshold assumed (USD 2 475 with a threshold).⁵ For cardiovascular effects, they followed Rice et al.'s (2010_[24]) probability-weighted approach which drew on the study by Salonen et al. (1995_[53]). However, rather than value premature deaths via a value of statistical life, they assumed that each premature mortality would result in the loss of 10.34 years of life, and cited Quinet (2013_[58]) in selecting a figure of EUR 126 000 (2013) for the value of a lost life-year. They then assumed a 10-year latency period between exposure and impact on cardiovascular health (risk) – reflecting the biological processes determining when disease impacts are felt after human exposure (Nedellec and Rabl, 2016_[59]).⁶ Combined with a 3% discount rate, Nedellec and Rabl (2016_[48]) obtain a figure for cardiovascular effects of USD 55 702 (2013) per kg of mercury (no threshold), and USD 24 509 (2013) (threshold).⁷ The total damage cost per kilo of mercury was therefore USD 61 328 (no threshold) or USD 26 985 (threshold), consistent with other studies which include cardiovascular effects (e.g. Rice and Hammitt (2005_[10]) and Giang and Selin (2016_[26])), representing about 90% of total costs.⁸

4.6. Shih and Tseng (2015)

The study by Shih and Tseng (2015_[27]) is one of the few estimations of mercury health costs outside of the United States, and adopts aspects of the studies by Rice and Hammitt (2005_[10]) and Spadaro and Rabl (2008_[11]). It used a co-benefits model to consider the benefits of mercury reductions associated with a policy of clean energy development

⁵ An exchange rate of 0.85 (Autumn 2017) is used to convert euros into US dollars in this section.

⁶ Nedellec and Rabl (2016_[59]) and (2016_[48]) apparently exclude the 15 year ecosystem lag which Spadaro and Rabl (2008_[11]) included to reflect how long it takes for changes in mercury deposition to show up in methylmercury concentrations in fish, but do not provide an explanation for this.

⁷ Estimates of cardiovascular effects by other authors do not include a threshold.

⁸ Rather strangely, Nedellec and Rabl (2016_[59]) describe these results as pertaining to “industrial emissions in the European Union” (Table 5 in that paper), whereas Nedellec and Rabl (2016_[48]) describe the same figures as “worldwide damage costs”. The accompanying calculations spreadsheet suggests the latter is the correct interpretation.

involving natural gas and renewable energy in Chinese Taipei. Mercury emissions reductions were estimated based on assumed mercury emissions factors for coal, and assuming zero mercury emissions for renewables and gas. Mercury depositions were estimated based on Hope (2006_[60]), and the US EPA's AER MOD model was used to simulate mercury deposition from anthropogenic sources. Changes in methylmercury exposure from fish consumption were estimated on the basis of AER MOD and Chinese Taipei data on fish origin, consumption and baseline mercury content. Changes in mortality risk associated with acute myocardial infarction (AMI) were calculated applying the exposure-response function developed by Virtanen et al. (2007_[61]). The mercury-associated IQ loss was estimated using the 0.6 IQ points per ppm mercury in hair figure used by Rice and Hammitt (2005_[10]).

Mortality valuation was achieved through the application of a value of statistical life (VSL) for Chinese Taipei of USD 1.75 million (Shih et al., 2012_[62]). The cost of reduced IQ was calculated by applying the USD 18 000 per IQ point value used by Spadaro and Rabl (2008_[11]), adjusted for purchasing power parity to USD 7 900 per IQ point in Chinese Taipei (2011). Citing the US EPA (2005_[22]), Shih and Tseng (2015_[27]) used an ecosystem response time of 15 years for the time assumed to be needed for emissions reductions to show up in reduced methylmercury content in fish and thereby, through fish consumption, on foetal IQ. They also assumed that 15 years is the latency period between changes in exposure and changes in risk of AMI mortality. (It is not clear whether this 15 years' latency was assumed in addition to the 15 year ecosystem lag, as would seem appropriate.)

As with previous studies such as Rice and Hammitt (2005_[10]) and Nedellec and Rabl (2016_[48]), Shih and Tseng (2015_[27]) found that the benefits of mercury emission reductions were dominated by AMI mortality compared with IQ benefits. AMI mortality benefits were almost five-times higher than IQ benefits, equal to USD 1.87 million per kg mercury with AMI mortality included, USD 0.325 million per kg when excluded. The latter IQ-only figure is two orders of magnitude higher than the comparable results reported by Spadaro and Rabl (2008_[11]) (USD 1 500 and USD 3 400 per kg depending on threshold assumption), a fact the authors suggested might be because their study area encompassed the densely populated Taipei metropolitan area.⁹ Shih and Tseng (2015_[27]) also assumed a steeper slope to the dose-response function linking methylmercury exposure and IQ loss (0.102 IQ points/ug/day compared with 0.036 IQ points/ug/day used by Spadaro and Rabl (2008_[11])), although this is a relatively minor difference compared with the relativities of the benefits estimates.

4.7. Giang and Selin (2016) and US EPA (2011)

The study by Giang and Selin (2016_[26]) is notable in a number of respects. In particular, it is a global-scale study which still uses relatively sophisticated global and regional atmospheric modelling of emissions and deposition – something which was rejected by, e.g., Spadaro and Rabl (2008_[11]) as too complicated. The study is undertaken in the context of the UN Minamata Convention on Mercury and the United States Mercury and Air Toxics Standards (MATS) policy, and the focus is on benefits experienced within the United States associated with these two policies over the period to 2050. MATS is interpreted as sufficient for the United States to comply with the requirements of the

⁹ The population density of Taipei itself is given by Wikipedia as 7 500 per km², compared with 470 per km² for the most densely populated state in the United States, New Jersey.

Minamata Convention, and hence any Minamata benefits estimated in the study are external benefits enjoyed by the United States as a result of mercury emissions reductions implemented in the rest of the world. This is categorically different from results estimated in other studies.

Giang and Selin (2016_[26]) draw on Streets et al. (2009_[63]) to make their emissions projections. Under the baseline scenario, emissions are projected to more than double between 2005 and 2050, largely as a result of growth in coal use in Asia. Since air quality abatement technologies often capture mercury as a co-benefit, the main differences in projections between the baseline and other scenarios depend on assumptions about what emission controls are implemented for coal use. The Minamata Convention requires the application of best available technologies, and hence Giang and Selin (2016_[26]) assume the application of flue gas desulphurisation or similar technology in the rest of the world under this scenario.

For the MATS case, which affects only US emissions, Giang and Selin (2016_[26]) model deposition to the United States to be 20% lower in 2050 than under the baseline, while deposition to the global oceans is 6% lower. Under the Minamata case, mercury deposition to the United States and to global oceans is respectively 19% and 57% lower in 2050 than under baseline. Thus, overall United States deposition reductions are similar under the MATS and Minamata scenarios, but the distributions of the reductions vary, with avoided deposition under MATS more highly concentrated in the north east United States, where there are significant coal-fired emission sources, and following precipitation patterns under Minamata, since this policy avoids increases in global background mercury concentrations. It can be expected that the benefits of the emissions reductions under these two scenarios will reflect these different deposition patterns.

Giang and Selin (2016_[26]) use the United States National Health and Nutrition Examination Survey (NHANES)¹⁰ to specify current regional blood methylmercury concentrations, and then scale these concentrations according to the predicted change in the intake of methylmercury via fish consumption. They take account of consumption of domestic freshwater fish and fish imported from global fisheries, using data from United States seafood market studies and data compiled by the US EPA. They also reference several studies to specify an ecosystem response time of 10 years between changes in depositions and changes in methylmercury concentrations in fish. Giang and Selin (2016_[26]) calculate the average US mercury intake in 2050 to be 32% lower under the MATS scenario compared with the baseline, and 91% lower than baseline under the Minamata scenario. Although the deposition decreases over the United States are modelled to be approximately the same between the two policy scenarios, the Minamata scenario has a greater impact on the mercury content of marine fish which makes up more than 90% of the US commercial fish market. However, the relative benefits under the MATS and Minamata scenarios are found to depend on the mix of fish type which is consumed in each region of the United States.

To translate these modelled changes in methylmercury intake into health impacts, Giang and Selin (2016_[26]) take the Axelrad et al. (2007_[19]) dose-response function for the relationship between IQ loss and maternal hair mercury concentration (0.18 IQ points lost per µg mercury per gram maternal hair), adjusted to control for confounding (Rice et al. (2010_[24])). They assume a linear functional form with no threshold. They also use the work of Virtanen et al. (2005_[64]) on the link between mercury exposure and heart attacks

¹⁰ See <https://www.cdc.gov/nchs/nhanes/index.htm>.

and calculate a risk coefficient of 0.1 per ppm mercury in hair. This compares with the values used by Rice and Hammitt (2005_[10]) and Rice et al. (2010_[24]) of 0.06 for heart attacks and 0.086 for all-cause mortality.

For the value of lost IQ points, Giang and Selin (2016_[26]) use the midpoint of the range of values used by the US EPA in their RIA for the MATS policy (US EPA, 2011_[7]) – USD 9 936. For heart attack mortality, they use a value of statistical life of USD 6.3 million, the mean of 26 study values used in the US EPA’s Environmental Benefits Mapping and Analysis Program (BenMAP) (RTI International, 2015_[65]). Finally, for non-fatal heart attacks, they use a value of USD 120 953 which covers direct medical costs and lost earnings over a five-year period, and which was used by the US EPA (2011_[66]).¹¹

The results of the Giang and Selin (2016_[26]) study can be broadly summarised as follows. The present value of the benefits to the United States of the MATS scenario, over the period to 2050, is estimated at USD 147 billion. These are benefits to the United States which accrue entirely as a result of reductions in domestic emissions of mercury. The present value of the benefits to the United States of the Minamata scenario over the same period is estimated at USD 339 billion. These are benefits to the United States which accrue entirely as a result of reductions in emissions of mercury in the rest of the world. If MATS is interpreted as the US policy to meet its own Minamata obligations, it can be seen therefore that the United States gains more (more than twice as much) from the actions of all other parties to the convention than it does from its own. As with other studies which include cardiovascular effects, the heart attack benefits of reduced mercury emissions dominate the avoided IQ losses (more than 90% of the total benefits).

Giang and Selin (2016_[26]) also report results in terms of benefits per unit of avoided emissions (USD 1.1 billion per tonne for MATS and USD 150 million per tonne for Minamata, far in excess of values estimated in other studies reviewed here), but the basis for these calculated figures but the basis for these calculated figures is not consistent with the standard economic approach adopted generally elsewhere.¹² Giang (personal correspondence) provided recalculated values of USD 47 000 per kg for MATS, and USD 6 900 per kg for Minamata. Recall that these figures include cardiovascular effects, said to account for over 90% of the total benefits. The IQ component of benefit could then be said to be in the region of USD 4 700 per kg for MATS, which is much more comparable with the figures reported elsewhere by (e.g.) Spadaro and Rabl (2008_[11]). For Minamata, the IQ benefits would be around USD 700 per kg, but note that these are the United States share in the total benefits of emission reductions in the rest of the world, which do not have a comparable figure elsewhere in the literature but which one would not expect to be as significant as “own generated” benefits.

¹¹ Giang and Selin (2016_[26]) also use what they call a human capital approach, which does not appear consistent with any other study considered in this review, and does not appear to add anything of particular interest compared with the more standard methodology. Hence it is not considered further.

¹² Giang (personal correspondence) confirmed that the reduction in mercury emissions in 2050 was used in the paper to provide a common comparator for results across different scenarios. This was instead of the more usual (and more meaningful in economics terms) total emission reductions over the whole estimation period used in other papers considered in this review. For the current paper, Giang (personal correspondence) provided revised figures based on this latter, comparable basis.

Giang and Selin (2016_[26]) conduct a sensitivity analysis which provides a check on their figures but also provides a useful insight into what might be more significant and less significant cost drivers in benefits assessment work of this type.¹³ First, as emphasised by Spadaro and Rabl (2008_[11]), the extent to which benefits of emissions control vary regionally is likely to be affected by speciation, that is, the relative proportions of the different forms of mercury in emissions. Giang and Selin (2016_[26]) find that, if 90% of global Minamata mercury reductions were to occur as elemental mercury, benefits would be USD 405 billion, or 20% higher, which is perhaps surprisingly low for a “global pollutant” (Spadaro and Rabl (2008_[11])).¹⁴ Nevertheless, the results indicate that, if policy prevents primarily metallic mercury emissions, there is greater long-range benefit to the United States and to global oceans from avoided emissions occurring elsewhere.

Second, Giang and Selin (2016_[26]) consider the influence of their assumption of a 10 years response time between a change in depositions and a change in the mercury content of fish, and find cumulative benefits to 2050 from Minamata projected to be USD 575 billion (70% higher than the base case) if the ecosystem lag is zero, whereas a slower (50-year) response reduces projected benefits to USD 60 billion (over 80% lower).

Giang and Selin (2016_[26]) also test the sensitivity of their results to assumptions about the make-up of the “basket” of fish consumed by the US population. As previously mentioned, they estimated that more than 90% of the US commercial fish market, and therefore the majority of US mercury intake, comes from estuaries and oceans, particularly the Pacific and Atlantic, both regions heavily influenced by emissions from non-US sources. For low and high sensitivity bounds, respectively, Giang and Selin (2016_[26]) therefore assumed that diets were 100% from either local freshwater or global sources. This resulted in a low estimate of the benefits of Minamata of USD 56 billion (100% freshwater), and USD 418 billion (100% global). This result indicates that – at least relatively – the United States gets little benefit from foreign mercury reductions if its diet consists only of domestic freshwater fish, which contains mercury primarily due to local and regional deposition.

Giang and Selin (2016_[26]) also provide (in the technical annex) a comparison of their results with those generated by the US EPA (2011_[7]), which estimated the benefits of MATS to be between USD 4.2 million and USD 6.2 million for one year’s birth cohort of around 250 000 children. With a 10-year time lag and 3% discount rate, Giang and Selin (2016_[26]) estimated a figure of around USD 150 billion for the present value of MATS benefits over the period 2005-2050. From inspection of their Figure 2, the present value of MATS benefits in 2015 appears to be around USD 2 billion. The large difference between these figures and their own was ascribed to three factors:

- The baseline policy scenario – Giang and Selin (2016_[26]) suggest that many of the measures they include in their definition of the MATS scenario were assumed by the US EPA (2011_[7]) to occur under the baseline;

¹³ Their sensitivity analyses relating to dose-response and economic valuation factors are less interesting than others so are not considered here.

¹⁴ If 90% were assumed to occur as oxidised or particular mercury, benefits would be largely unchanged since the base case emissions projections assume control technologies (e.g. FGD) which capture oxidised mercury (Streets et al., 2009).

- Health endpoints – Giang and Selin (2016_[26]) included estimates of cardiovascular effects and impacts on mortality, which were excluded by the US EPA (2011_[7]);
- The population at risk – the US EPA (2011_[7]) analysis examined exposure changes only for a subset of the US recreational angler population which consumes self-caught fish from freshwater bodies, whereas Giang and Selin (2016_[26]) estimated benefits for the entire US population. The number of affected births considered by the US EPA (2011_[7]) was around 250 000 per year, compared with the total US annual birth cohort of around 4 million.

Starting from the Giang and Selin (2016_[26]) estimate of USD 2 billion in 2015, limiting to IQ benefits alone would reduce the value to around USD 200 million, and reducing the population at risk from 4m to 250 000 would reduce the value down to around USD 12.5 million. It is not clear how much the difference in assumed baseline emissions reductions might reduce this value even further. Against this is the fact that US EPA (2011_[7]) also assumed no lag between deposition changes and fish concentrations, which might increase estimates by around two-thirds. Nevertheless, these adjustments do suggest that the apparently quite dramatic differences between the Giang and Selin (2016_[26]) and US EPA (2011_[7]) results can be reduced so that the studies produce results of a similar order of magnitude.

4.8. Gayer and Hahn (2006) and Hagen et al. (1999)

All of the studies considered in this review so far take essentially the same approach to the valuation of the human health impacts of mercury – quantifying IQ impacts in terms (primarily) of changes in earnings (productivity), and cardiovascular impacts generally in terms of willingness-to-pay for mortality risk reductions (the value of statistical life). No studies take a more “holistic” approach to the valuation of the welfare effects of mercury’s health – and environment – impacts. However, the studies by Gayer and Hahn (2006_[25]) and Hagen et al. (1999_[34]) are notable departures from this general trend.

Gayer and Hahn (2006_[25]) undertook a cost-benefit analysis of two proposals to regulate mercury emissions from US coal-fired power stations, one based on “command-and-control” and the other on a “cap-and-trade” approach. For benefits, they estimated the effect of the complete elimination of all power plant mercury emissions (48 tons), and then multiplied this estimate by the proportional decrease in annual emissions under each policy scenario. Their study is notable in that it is the only one to take a strict willingness-to-pay approach to the valuation of children’s IQ losses and gains, based on the results of Agee and Crocker (1996_[67]), who examined parental preferences for the purchase of lead-reducing chelation therapy for their children. Lutter (2000_[68]) linked the resulting estimate of the willingness-to-pay for lead reduction in children to an estimate of the relationship between lead and IQ scores, and estimated that parental choices over chelation therapy indicated a willingness-to-pay between USD 1 295 and USD 2 236 per IQ point for their children (2004 prices).

Gayer and Hahn (2006_[25]) did identify some potential problems with using chelation therapy to infer the value of an IQ point. For instance, they stated that chelation therapy can be associated with negative treatment side effects (e.g. pain) which would make willingness-to-pay for it an underestimate of the value of any IQ benefits. Further, the long-term effect of chelation therapy on IQ is apparently unclear, and hence it would be difficult for parents to estimate what their willingness-to-pay should be – even assuming that they are able and willing to express their own willingness-to pay-in terms of the

(distant) future increase in earnings of their offspring, and any other known or unknown impacts.

Interestingly and importantly, Gayer and Hahn (2006_[25]) claimed that their assumptions about the value of an IQ point were consistent with the estimates in the literature of the effect of IQ on earnings (whilst also concluding that there is no clear consensus on the relationship). They cited Bound et al. (1986_[69]) who claimed that there is no significant impact of IQ on earnings, and Zax and Rees (2002_[70]) whose estimates implied a value of around USD 4 000 per IQ if lifetime earnings are around USD 1 million (Max et al., 2004_[38]). They further cited Cameron and Heckman's (1993_[71]) estimate that a standard deviation increase in IQ score leads to a 7-10% increase in earnings (around USD 6 000 per point for boys, USD 5 000 for girls). Finally, on what they call "the high end", they referenced the study by Neal and Johnson (1996_[72]), which found a 17% increase from a standard deviation increase in IQ (USD 13 700 for boys, USD 9 500 for girls).

Note, however, that even the "high end" estimate of Neal and Johnson (1996_[72]) is lower than values used elsewhere in the mercury (and lead) IQ literature. As summarised by Spadaro and Rabl (2008_[11]), most of these use the work of Schwartz (1994_[20]) and Salkever (1995_[21]) and an assumed impact on earnings of around 2.4% per IQ point, or 36% per standard deviation. The result is values around USD 15 000 – USD 20 000 per IQ point, with the EPA (2005_[6]) work using a relatively low value of USD 9 000. It is beyond the scope of this review to provide a thorough evaluation of the IQ-earnings literature, but it is worth noting this range of values for the impact of IQ, and the fact that most mercury valuation studies have used a study which appears to find estimates much higher than other studies on the topic.

In other respects, Gayer and Hahn's (2006_[25]) study was less unusual. They used relatively simple, high-level relationships between mercury emissions, depositions, mercury-fish concentrations and fish consumption, and assumed no lag between changes in deposition and changes in exposure. They assumed a linear dose-response function with no effect threshold, and took the simple mean of the IQ decrement estimates from applying each of the functions from the Seychelles, Faroe Islands and New Zealand studies. For the emission reductions expected under each policy scenario over the 2005-2020 period, they estimated total benefits of the command-and-control scenario at USD 82 million – USD 142 million (low and high IQ-earnings cost), and USD 86 million – USD 149 million for the cap-and-trade policy (both calculated with a 3% discount rate). Gayer and Hahn (2006_[25]) did not give exact figures for emissions reduction under their assumed scenarios, but inspection of their Figure 2 suggests reductions of 73 tons over 2005-2020 under the command-and-control policy, and 96 tons under the cap-and-trade policy. This would translate into benefits per kg of emissions reduced of USD 1 238-2 144 for command-and-control, and USD 987-1 711 for cap-and-trade. The difference between the two policies is most likely down to the different timings of emission reductions over the 2005-2020 study period.

The study by Hagen et al. (1999_[34]) is apparently the only stated preference valuation study of the total economic value of mercury emissions reductions. It was undertaken at a time when the state of Minnesota was said to be considering the development of policies which would place limits on the emissions of mercury into the environment. For this study, a mail survey was sent to 2 500 Minnesota households and personal interviews were conducted with 250 others. The responses to the survey were used to estimate monetary values of the benefits respondents associated with the described policy. A

statistical valuation function was estimated, and from this information the average annual willingness of Minnesota households to pay for the policy was computed.

This study evaluated the benefits respondents derived from four different policy scenarios affecting anthropogenic mercury emissions, but focused on the “baseline” 50% reduction in regional Midwest emissions, which respondents were told would reduce deposition in Minnesota by 12.5%.

In valuation studies such as this, a key element is how much and what type of information is given to those completing the survey about the policy they are being asked about and the impacts it could have. In this case, participants were given information about the health impacts of mercury and how they are generated, the environmental impacts of mercury, and the policy proposal(s) to combat them. They were then asked questions designed to elicit how much they might be willing to pay for the policy and hence for the associated benefits.

Regarding human health effects, participants were told about the fish consumption pathway to mercury exposure, and that the Minnesota Department of Health issues advisories recommending limits on the amount of some fish eaten from some lakes and rivers, particularly for women of child-bearing age because of the susceptibility of unborn children to mercury’s effects on neurological development. They were also told that “up to five percent” of Minnesotans (primarily anglers and their families) could be eating enough mercury-contaminated fish to cause health impacts, although nothing “easily observed and quantified”. Similarly regarding environmental impacts, participants were told that possible effects on birds and mammals could affect the muscles and nervous system, as well as possible decreases in the ability to bear and raise offspring. Individual birds and local populations of these birds were said to be at possible risk on some lakes when fish contain mercury above the level which scientists consider safe, but below the level where wildlife damages are easily observed and quantified.

Regarding policies, participants were told that the State of Minnesota and neighbouring states were considering a regional mercury reduction programme, which would cut future mercury emissions by half. This would result in about a 12% reduction in the amount of mercury deposited in the region, leading to a gradual reduction in the mercury content in fish, reaching 12% after about 20 years. The most restrictive fishing advisories would decrease from roughly 70% of lakes down to about 60% (with some minor benefit to rivers). The percentage of lakes which posed risks to wildlife was also said to change (e.g. 50% of lakes were said to pose a current risk to kingfishers, falling to 35%). Participants were told that they would pay for this policy through general price increases due to higher production and administrative costs, and were then asked whether they would be prepared to pay a stated amount which varied across the sample.

For the baseline policy scenario, a “best estimate” of annual household willingness-to-pay of USD 119 was obtained from the mail survey, and of USD 198 from the personal interviews, a difference ascribed by the authors to “yea-saying” behaviour as a result of the personal nature of the interview procedure. Hagen et al. (1999_[34]) stated that a substantial majority of respondents had expressed the belief that they would pay as much or more than the stated cost (with nearly half suggesting that it would cost more than was stated). Although this was seen as evidence by the authors that participants had taken the valuation exercise seriously, it could also be seen to imply that their responses reflected an expectation of the cost of the proposed policy, not a valuation of its expected benefits. More generally, it is worth noting that, although respondents were provided with a significant amount of information about the proposed policy and its effects, that

information was quite general. Impacts were generally described in terms of rather vague risks to health and wildlife, and specifically the number of state fishing advisories; prenatal neurological effects were specified in terms of “delayed walking and talking” rather than, for instance, reductions in IQ leading to reductions in expected lifetime earnings.¹⁵ This could explain the lack of clear sensitivity to scope shown by respondents’ valuations of policies with different emission reductions.

Using the more conservative mail-survey estimate, a state-wide willingness-to-pay was computed by multiplying the per-household figure by the number of households (1.786 million in 1996) to yield an annual state willingness-to-pay of USD 212 million per year. As a thought experiment, these estimates can be translated into illustrative cost per tonne figures. The baseline policy was described as reducing Minnesota and Midwest emissions by 50%, giving a total reduction in depositions of 12.5%. Hagen et al. (1999_[34]) stated that reducing Minnesota emissions alone by 50% would generate deposition reduction of 5%, implying that emissions in the rest of the Midwest were about 50% higher than Minnesota’s five tons (Jackson et al., 2000_[73]). Benefits were said to take 20 years to be fully realised. With no discounting of future benefits, an annual valuation of around USD 200 million to secure a reduction of around 5 670 kg (50% of 12.5 tons or 11.34 tonnes) equals USD 35 273 per kg benefit. If one considers the benefits over a 20-year period (taking account of the stated ecosystem response time), the present value of annual benefits is equal to around USD 26 235 per kg. If the benefits are weighted according to the ecosystem response in each year (assuming a proportional linear response from Year 1 to full response by Year 20), the present value of annual benefits rises to just under USD 109 000 per kg. These values are clearly orders of magnitude higher than those estimated in most other studies considered in this review (the Giang and Selin (2016_[26]) and Shih and Tseng (2015_[27]) studies being the exceptions).

4.9. Steckling et al. (2014) and Steckling et al. (2017)

The studies by Steckling et al. (2014_[8]) and (2017_[9]) are unusual in that they are the only ones which have attempted to quantify the health impacts of mercury exposure in the context of artisanal small-scale gold-mining (ASGM). Also unusually, their assessment was based on the Disability-Adjusted Life-Year (DALY) concept, developed initially under the auspices of the World Bank and World Health Organization (Murray and Lopez (1996_[33])). Only Rice and Hammitt (2005_[10]) in this review have previously used DALYs, to quantify some of the impacts of neurological impairment and non-fatal heart attacks.

ASGM is an activity practiced in over 70 countries worldwide, and Zimbabwe is amongst the top ten countries using large quantities of mercury to extract gold from ore. ASGM is the main global source of *intentional* anthropogenic mercury emissions. The main route of mercury exposure in ASGM is inhalation of vaporised elemental mercury released during amalgam smelting. Mercury is also dumped in the environment during mining, leading to emissions to air and water. Chronic exposure to mercury vapours can cause

¹⁵ In fact, one of the few times the survey did appear to get specific was in describing the effects of mercury poisoning (“Exposure to much higher levels of mercury found in other parts of the world show that mercury can cause kidney problems, nervousness, insomnia, tremors, blurred vision, slurred speech, and fatigue”), which is not a health impact likely to be experienced from eating contaminated fish. It is not clear to what extent including this information might have influenced responses.

erethism (“mad hatter disease”), tremor, gingivitis, and other symptoms, but due to its slow onset and diffuse symptoms, it can be difficult to make an early diagnosis of chronic mercury intoxication. The Steckling et al. (2014_[8]) analysis was performed to derive a preliminary estimate of the DALY burden due to mercury use in ASGM in Zimbabwe.

The study used surveys of miners and controls to identify cases of chronic mercury intoxication, from which sample prevalence the number of cases in the Zimbabwe population was estimated. The WHO (2008_[74]) is the source of many disability weights but does not include a disability weight for chronic mercury intoxication. It was judged that the severity of this disease is comparable to alcoholism, and hence the disability weight for this condition (0.18) was used as proxy, applied to life expectancy estimates, using non-uniform age weights and a discount rate of 3%.¹⁶ Chronic mercury intoxication is not associated with elevated mortality and hence is not associated with lost life-years.

On the basis of these data and a range of necessary assumptions, Steckling et al. (2014_[8]) estimated that there were around 17 700 incident cases of chronic mercury intoxication in Zimbabwe in 2004, with an average duration of 35 years and a disability weight of 0.18 per year, giving a total (discounted) 95 400 DALYs in the population. This represents 5.4 (discounted) DALYs per case.

Steckling et al. (2017_[9]) extended the basic methodology the authors employed for Zimbabwe to cover additionally the countries of Indonesia, Tanzania, the Philippines and Ecuador. Original data from surveys of gold miners in these countries were pooled and analysed. A previously developed tool was used to diagnose incidents of chronic moderate and severe mercury intoxication, and the resulting estimate of prevalence was applied to estimates of the population of gold miners in 62 countries for which data could be found. The disability weight for chronic moderate mercury intoxication previously developed by Steckling et al. (2017_[75]) was applied to the resulting estimates of the number of cases of the disease. (Moderate severity was assumed on the grounds that any individual suffering from a severe case would not be able to work, and hence could not be present in a survey of gold miners.)

The study estimated that between 14 million and 18.9 million people could be employed miners in the ASGM sector in these 62 countries, the largest being the People’s Republic of China (2.7 million). The prevalence of moderate chronic mercury intoxication was estimated at between 24.2% and 29.6%. Applying the disability weight of 0.368 gave a disease burden range of 1.22 million DALYs (lower estimate of miner population and low prevalence estimate) to 2.39 million DALYs (higher population estimate, higher prevalence). If the earlier Steckling et al. (2014_[8]) disability weight (0.18) is employed, these figures would fall by just over a half.

¹⁶ In a more recent study, Steckling et al. (2017_[75]) reported the results of a survey of 118 mercury health and burden of disease experts to estimate specific disability weights for moderate and severe chronic mercury intoxication. The results were disability weights for moderate and severe chronic mercury intoxication of 0.368 and 0.588 respectively, and comparable DW for alcoholism of 0.312, suggesting that transferring an alcoholism disability from the GBD might be a reasonable approximation to the weight which would be obtained for chronic mercury intoxication if the same GBD methodology were followed.

4.10. Applications in regulatory impact assessment

The final part of this section of this review considers a number of applications of results of other studies in impact assessments of government regulations governing mercury.

The Products Containing Mercury Regulations (Canada Gazette, 2014_[11]) were introduced to prohibit the Canadian manufacture and import of products containing mercury or any of its compounds, with some exemptions for “essential” products for which there were no technically or economically viable alternatives (examples given being certain medical and research applications, and dental amalgam). In the case of lamps, rather than introducing a prohibition, the regulations limit the amount of mercury contained in fluorescent and other types of lamps, to reduce mercury emissions from lamps and to facilitate a gradual shift to higher energy performance lighting. The regulations also contain additional provisions relating to labelling, reporting and authorisation (through a permit system) of specific applications for which it can be shown that no suitable alternatives exist.

The Regulatory Impact Assessment (RIA) accompanying the publication of the regulations contained forecasts of the impacts of their introduction on the volumes of mercury being placed on the Canadian market, being released into the environment from products, and being emitted to air from products, over the period 2015-2032. Over this period, the regulations were expected to lead to a cumulative reduction of approximately 41 tonnes of mercury entering the Canadian marketplace, 21 tonnes of which would have been released to the environment, of which just under 17 tonnes would have gone to land, and around 4 tonnes would have been released to air.

When generating an estimate of the monetary value associated with this reduction in emissions, the RIA cited the results of Rice and Hammitt (2005_[10]) for the IQ-related benefits of mercury emissions reductions – around USD 10 500 per kg (2000 prices) if no threshold is assumed, and around USD 4 000 per kg with a threshold assumption. It then referred to Spadaro and Rabl’s (2008_[11]) “reconciliation” of their results and those of Rice and Hammitt (2005_[10]), which obtained very similar results after adjustment for differences in methodology and data. The RIA gave this as justification for using the lower bound Rice and Hammitt (2005_[10]) value of USD 3 900 per kg to value the possible benefits of the regulations (uprated to CAD 6 110 (2012) per kg). This lower-bound figure is based on IQ losses only and excludes any cardiovascular impacts. The justifications given for using a lower-bound estimate were that it would provide a higher degree of certainty than the higher-end values; and that the lower population and population density of Canada compared with the US would be expected to result in lower benefits of mercury emissions reductions anyway.

The valuation of reduced mercury emissions was restricted only to the avoided emissions to air, which represented approximately 19% of the total environmental releases forecast to be avoided under the regulations. Most (80%) of the forecast avoided releases would be releases to soil (at landfill sites) but these were not valued on the grounds of “a lack of systematic evidence and a high degree of uncertainty regarding the long-term behaviour and fate of mercury in landfill sites”. The possible environmental benefits associated with the regulations were discussed in the RIA only qualitatively due to a lack of suitable quantitative data and evidence. The regulations were expected to result in some benefits for Canadian wildlife and ecosystems, as mercury is “known to impair reproduction potential in some wild populations of fish and birds and to have neurological effects in fish-eating animals”. The RIA noted that some of these benefits would be expected to be

realised internationally as roughly two-thirds of Canada's mercury emissions to air deposit internationally.

The RIA stated that, from 2015 to 2032, the discounted benefits of mercury air emissions avoided under the regulatory scenario were estimated to be CAD 18 million (2012; present value base year of 2014; 3% discount rate). The discounted costs of the regulations were estimated to be CAD 9 million over the same period (CAD 5.5 million to importers and consumers due to increased prices, CAD 2.1 million to government, and administrative costs of CAD 1.4 million to manufacturers and importers as the result of additional labelling and reporting). The conclusion was that the estimated benefits would be twice as large as the discounted costs, and that, altogether, the discounted net benefits of the regulations would be CAD 9 million.

Although this would normally be considered a significant difference between benefits and costs, it should be recalled that the unit value used to monetise emissions reductions assumed no delay in the time taken for this to show up in fish concentrations, maternal intake through fish consumption and, thereby, foetal exposure to methylmercury. However, the discussion of the results of Giang and Selin (2016_[26]) also demonstrated the potential sensitivity of benefits estimates to this assumption. To examine the possible implications in the context of this RIA, simple straight-line interpolation was made of the emissions figures presented in the RIA under the baseline and policy scenarios to the environment and air, to permit a calculation of the emission reduction in each year over the period 2015-2032. Applying a CAD 6 100 per kg valuation to the air emission reductions and discounting at 3% gives a present value of CAD 16.8 million – close enough for the purposes of illustration to the CAD 18 million in the RIA. Assuming a 10-year ecosystem-response lag, as per Giang and Selin (2016_[26]), reduces this present value by almost 80% to CAD 3.6 million, well below the estimated cost of the regulations. Extending the planning horizon by 10 years generates a present value of CAD 9.8 million, but presumably costs would also be expected to accrue over this extended period too.

As noted, the RIA did not include any quantification of benefits of reducing emissions to landfill, which represent a much bigger proportion of total reductions. Introducing these into the calculation, using the same CAD 6 100 per kg valuation and 10-year response lag, gives a present value for these landfill benefits alone of over CAD 19 million. Increasing the response lag to 20 years (which still might be considered a short time for mercury to leach from products into soil, find its way into fish and then expose pre-natal children) and extending the planning horizon to 2015-2040 reduces this present value to just over CAD 8.5 million – but could still be sufficient (along with the air emissions benefits) to justify the regulations if additional costs over this period were not significant.

In this case, therefore, these regulations might be judged unjustified on cost-benefit grounds under less optimistic assumptions about the ecosystem response time for mercury emissions to air – assuming the lower-bound Rice and Hammitt (2005_[10]) value of mercury reductions is the most appropriate to use in this context. They could be justified on the same grounds depending on the likely ecosystem response time for mercury emissions to landfill.

Marsden Jacob Associates (MJA) (2015_[4]) undertook a regulatory impact assessment of various ways of implementing the Minamata Convention in Australia. Three options were considered:

1. “Minimum” Minamata implementation, which was estimated to reduce Australian mercury emissions by 4 tonnes;
2. The “minimum” policy (1.) plus reduction in emissions from waste dental amalgam (a reduction of 6 tonnes);
3. The “extended” policy (2) plus earlier introduction of a pesticides phase-out (also 6 tonnes).

In terms of the assessment of the benefits of these emission reductions, MJA (2015_[4]) listed the following simplifying assumptions:

1. Mercury that is likely to be disposed of through hazardous waste or recycling processes is unlikely to enter the environment;
2. All mercury emissions were considered equally toxic irrespective of medium (air, land or water) and the form of mercury (elemental mercury and mercury compounds); and,
3. The analysis focused only on Australian emissions impacted by the Convention and the resulting benefit to the Australian population and environment.

Regarding quantification of IQ loss, MJA initially started by combining the results of an Australian survey of maternal blood mercury levels with different dose-response assumptions from US EPA (2011_[7]) and Trasande et al. (2006_[29]). Assuming a no-effects level of 3.5 µg per litre, they estimated IQ losses of between 36 and 494 for the 2013 birth cohort in Australia. However, despite referencing a number of studies providing estimates of the costs of a lost IQ point, for valuation purposes they fell back on Spadaro and Rabl’s (2008_[11]) values for the costs of mercury emissions (USD 1 500 per kg as a global mean cost with a threshold (USD 3 400 per kg with no threshold) and USD 4 380 per kg as a cost for the United States (USD 9 993 per kg with no threshold). MJA performed a benefits transfer exercise on these US figures by applying a PPP-based GDP ratio to estimate values for Australia of AUD 4 862 per kg (threshold) and AUD 11 093 per kg (no threshold).

Note that the USD 4 380 per kg figure for the United States from Spadaro and Rabl (2008_[11]) was actually the result of those authors’ attempt to reconcile their figures with those of Rice and Hammitt (2005_[10]). This reconciliation involved (amongst other adjustments) stripping out the effects of the assumed 15-year ecosystem response lag, increasing the slope of the dose-response function, and recognising the higher unit cost (due to speciation) of United States emissions relative to global emissions. By assumption, MJA did not take account of speciation or pathway variations across mercury emissions sources. The threshold value for Australia of AUD 4 862 per kg was used to translate estimated reductions in emissions into economic values. Estimating over a 20-year time horizon using a 7% discount rate gave present values of benefits of AUD 167 million for the baseline scenario, USD 214 million for the scenario including dental amalgam, and USD 273 million for the scenario also including faster phase-out of pesticides. The “most likely” net present values for these three options were estimated to be AUD 145.4 million, AUD 148.6 million and AUD 207.0 million respectively, implying net costs of around AUD 20 million for the first, and around AUD 70 million for the other two. Unfortunately, the cost estimates are not presented in a way which facilitates scenario analysis. However, attaching a 10-year ecosystem delay to the presented benefits profiles results in benefit present values of AUD 38 million, AUD 51 million and AUD 81 million respectively, clearly sufficiently reduced to threaten the economic justification of these options.

ECHA (2011_[2]) and (2011_[3]) compiled two impact assessments of regulations to restrict the use of mercury and mercury compounds. ECHA (2011_[3]) undertook a review of an existing EU restriction on the use of mercury in measuring devices. This review had the objective of identifying whether the emergence of new technically and economically feasible alternatives to measuring devices containing mercury could permit the tightening of the existing restriction and phase out mercury in this use. The focus of the report was therefore the assessment of the technical and economic feasibility of alternatives to such devices. In addition, no quantitative release estimates were made, since it was considered that they would have to be expressed in “exceedingly broad ranges to take into account all accumulated uncertainties”. Instead, the total estimated amount of mercury placed on the market in measuring devices containing mercury (3.5-7.6 tonnes in the EU in 2010) was used to describe the maximum potential for mercury emissions to the environment which might ultimately occur as a result of this use.¹⁷ However, it was recognised that a quantitative estimate of actual emissions would require detailed information on current waste management practices and emissions resulting from the waste stage. It was suggested that the low separate collection rate and resulting poor waste treatment of a substantial part of the devices leads in the long term to a relatively high share of mercury used in these devices being released to the environment. This was despite the fact that, in the EU, measuring devices containing mercury are legally required to be collected separately from other (hazardous and non-hazardous) waste streams at the end of their service life. In practice, collection efficiencies of mercury in measuring devices in accordance with requirements set out in hazardous waste legislation have been estimated to be as low as 20%, and collection efficiencies above 50% are in general not expected.¹⁸

On the basis of the proposed scope of the restriction, it was assumed that the declining trend in the number of mercury devices placed on the market would continue, and with an average lifetime of mercury-containing devices of around 10 years in most applications, the restriction would have its full effect 10 years after adoption, when all the existing mercury containing devices would be replaced. By 2024, just under 3 tonnes of mercury would no longer be placed on the market in measuring devices, and just over 60 tonnes would be saved over the 2015-2034 period. The present value of costs of replacement over the same period was estimated at EUR 129 million, varying across device type. For instance, replacements for barometers and manometers using mercury were judged to be no more costly than the existing technology, and no market was found for pycnometers and metering devices using mercury; sphygmomanometers were estimated to cost EUR 29 million to replace, and thermometers EUR 97.4 million, although the analysis was unable to take account of the value of the improved performance of some electronic alternatives.

The resulting estimated costs per kg of mercury avoided therefore ranged between zero and over EUR 19 000, with a weighted average of EUR 4 100 per kg. To judge whether this cost might be justified by the expected health benefits, a brief review of the valuation literature was undertaken. The values estimated by Rice and Hammitt (2005_[10]) of

¹⁷ 134 tonnes of mercury was estimated to be present in the total stock of measuring devices containing mercury in the EU in 2010. 5-15 tonnes of mercury were estimated to be supplied annually for use in certain types of measuring devices, but this figure was not considered a good indicator of potential emissions, which depend on measurement practice, re-use activities and so on.

¹⁸ Even if these figures are low, they suggest that assuming that all mercury in measuring devices placed on the market will ultimately be released into the environment is an exaggeration.

between USD 3 900 and USD 194 500 per kg mercury reduced were cited, as were the figures generated by Spadaro and Rabl (2008_[11]) of USD 1 500 and USD 3 400 (with and without threshold) for the overall global benefit. The Rice and Hammitt (2005_[10]) values were preferred on the grounds that EU GDP per capita is relatively close to that of the United States, compared with global GDP per capita. No formal comparison of costs and benefits was undertaken on the basis of this review; health benefits of between EUR 5 000 and EUR 20 000 per kg of emitted mercury were cited, although it was acknowledged that these values relate to emissions (to air) and are not directly comparable with the cost-effectiveness of reducing the amount of mercury placed on the market that was estimated in ECHA (2011_[3]).

ECHA (2011_[2]) also undertook an impact assessment of proposals to restrict the use of five phenylmercury compounds in the EU. The 2005 EU mercury strategy Action 8 specifies that in the short term the European Commission will further study the few remaining products and applications in the EU that use small amounts of mercury, and in the medium to longer term, any remaining uses may be subject to limitations under the REACH regulation. The purpose of ECHA's (2011_[2]) report was to investigate uses of certain phenylmercury compounds as catalysts in polyurethane systems as a source of mercury emissions and the possibility for restrictions on the manufacture and use of these compounds.

ECHA (2011_[2]) estimated that 75-150 tonnes of phenylmercury compounds (i.e. 16-31.3 tonnes mercury, calculated from the mercury/phenylmercury-neodecanoate ratio) are manufactured annually for use in the production of phenylmercury catalysts in the EU, of which 40-85 tonnes are used in exported catalysts. These catalysts are used in the manufacture of polyurethanes (PUs), which have a number of diverse applications, including gaskets and seals, encapsulants for electronic assemblies, film and television props, vibration dampers, clear PU on labels, water resistant coatings and concrete sealants, marine repair and repair on conveyor belts, rollers on swivel chairs and roller skates, and in shoe soles. They have also been used in floorings, but current use was not confirmed. Use was assumed to be declining over time across all uses, consistent with historical trends (although use data were found to be poor).

It was considered that mercury could be released at all stages of the life cycles of the compounds concerned, and the products in which they were used – manufacture, use and waste. A detailed consideration of the environmental fate of the mercury compounds was undertaken, which took into account factors such as the half-life of mercury compounds in polyurethane and the service life of polyurethane products in determining emissions. Evidence was also presented on the possible forms of mercury which might be released at the different states. On this basis, from the quantity of phenylmercury compounds used in the EU in 2008, around 2.4 tonnes of mercury were estimated to be released into the environment during the formulation of polyurethane systems, and of the 31 tonnes of mercury used in articles, 3.2 tonnes were estimated to be released in the article use phase. In the waste stage, about 8 tonnes of mercury were estimated to be released to the environment from incineration, and a total of around 27 tonnes was estimated to end up in landfills – of which about 250 kg per year would be released to the environment. Combining assumptions about the baseline use of the compounds, the service life of products and waste practices, the estimated emissions avoided in each year following the introduction of a restriction on use were estimated. Most of these emissions were assumed to be to air in the form of elemental mercury.

Costs of implementing the restriction were estimated in terms of R&D costs of finding and introducing mercury-free catalysts, assuming some processes would be “easy” to replace and others would be “difficult”. Lost profits on exports of phenylmercury compounds manufactured in the EU were also included. Over the 10 years following the implementation of the restriction (after which time it was assumed all uses would have ceased anyway), the cost per kg of mercury emissions avoided was estimated at EUR 649 and EUR 802, depending on the length of the initial derogation period (representing total present value costs of just under EUR 10 million and EUR 15 million respectively). Emissions would actually be avoided over a longer period than this (although some mercury being disposed of to landfill was expected to remain potentially for decades) – for instance, over 2018-2027, one restriction option was estimated to save around 15 tonnes of mercury, but almost double this amount over the 2018-2040 period. As with ECHA (2011_[3]), a valuation of the health and environmental benefits of these emissions reductions was not undertaken, even by transferring values from existing studies, on the grounds that it had not been possible to establish a level of exposure. Instead, reference was made to the review undertaken by ECHA (2011_[2]), which was said to have concluded that “the majority of the studies reviewed have a benefit estimate between EUR 5 000 – 20 000 per kg mercury reduced”. No mention was made of the timing or location of the impacts of emissions reductions.

5. Summary evaluation

The review in the previous section covered a large number of diverse and often complex studies. They will be briefly summarised in terms of the criteria presented above, as follows.

5.1. Scope of uses and geographical coverage

In terms of the (direct or indirect) uses of mercury which were the subject of the studies, impact-pathway studies generally considered the impact of mercury emissions from the coal-fired electricity generation sector, while damage valuation studies (and the Spadaro and Rabl (2008_[11]) “hybrid” study) considered all emissions which contribute to mercury exposure through fish consumption (sometimes with an attribution to coal-fired electricity). Only the studies by Steckling et al. (2014_[8]) and (2017_[9]), gold-mining), and the impact assessments from Canada (Canada Gazette (2014_[11]), various products), Australia (Marsden Jacob Associates (2015), various uses such as dental amalgam) and the EU (ECHA (2011_[2]) and (2011_[3])), measuring devices and catalysts) explicitly considered emissions in other use contexts. This probably reflects policies in the 1990s and first decade of this century for improving general air quality and the drive to install flue gas desulphurisation, and since then a wider interest in mercury emissions reduction (linked to, e.g., the Minamata convention). Apart from these three impact assessments, most studies were undertaken in the United States, with additional assessments from Chinese Taipei, Greenland, France and Zimbabwe, as well as wider regional and global studies.

The United States electricity-focussed studies (e.g. US EPA (2005_[6])), Rice and Hammitt (2005_[10]) and the Spadaro and Rabl (2008_[11]) and Giang and Selin (2016_[26]) studies make an interesting comparison because of the way they treat the relationship between geographical and economic boundaries. The electricity studies are concerned solely with the value to the United States of reductions in domestic emissions of mercury, although these reductions would have benefits for other countries due to the global dispersion of the pollutant. Spadaro and Rabl (2008_[11]) estimate the global benefits of emissions reductions under the assumption of perfect mixing, so that effectively it does not matter where in the world those reductions occur. Giang and Selin (2016_[26]) consider the value to the United States of both global and United States emission reductions, thereby estimating the “internal” and “external” benefits of the Minamata convention. The distinction is important since estimating accurate location-specific values for emissions requires knowledge of the speciation of the emissions source – the higher the proportion of reactive gaseous mercury, the more local to the source is deposition. By taking a global perspective, Spadaro and Rabl (2008_[11]) effectively estimate the value of an “average” tonne of mercury, but (as demonstrated by their Rice and Hammitt (2005_[10]) comparison) estimating the value of a tonne emitted from a particular country requires adjustment for that country’s speciation.

The US EPA (2005_[6]) results demonstrate the potential importance of speciation. For the “base case” CAIR policy, implied values per kg of mercury emissions reduced were USD 817-1 429, depending on the fishing model used, but the two CAMR control options implied values of USD 198-307 per kg of mercury emitted, a difference which US EPA (2005_[6]) ascribed to the removal of a high proportion of RGM and particulate mercury emissions due to the initial policy, whereas the subsequent CAMR policies necessarily focussed more on metallic mercury which disperses more globally. This does not mean that CAMR necessarily had lower global benefits than CAIR, but it did have lower benefits to the United States.

Ultimately, a problem with the current mercury literature is that the only context which has received thorough and rigorous analysis is that of mercury emissions from coal-fired power generation, almost exclusively in the United States. More recent studies of policies in other contexts (e.g. mercury in products, dental amalgam, etc.) have elected to transfer values estimated in the power generation context rather than estimate their own values from a detailed impact-pathway analysis. As a result, they implicitly assume that the circumstances of the context of their policy are equivalent in value-relevant terms to the coal-fired power generation context. As the speciation example just given, and other examples discussed above and below, this is not necessarily the case, and hence, the values that are being used could be giving an inaccurate indicator of the value of those policies.

5.2. Health and environmental endpoints

The main endpoints of concern were even more focused in the studies reviewed than the policy contexts, being primarily on neurological development measured in terms of IQ and associated impacts on earnings (and to a lesser extent schooling and care costs). Gayer and Hahn (2006_[25]) is the only study to use parental willingness-to-pay for IQ improvements for their children (via lead-reduction treatment) as a measure of the value of health benefits of emissions reductions. Only Rice and Hammitt (2005_[10]) estimated non-IQ neurological effects on general wellbeing and quality of life (via QALYs), and this was based on ultimately arbitrary assumptions because of a lack of evidence of the

nature of these effects in the general population. A smaller number of studies (based on Rice and Hammitt (2005_[10])) also considered possible impacts on cardiovascular health and mortality (via willingness-to-pay, lost life-years and costs of illness), but other studies explicitly excluded these impacts on the grounds of weak evidence. Those studies which did value both IQ and cardiovascular impacts found that the latter dominated the former in terms of economic value, by a factor of at least 5:1. Only Steckling et al. (2014_[8]) and (2017_[9]) considered (using DALYs) the impacts of chronic mercury poisoning as a result of direct exposure to mercury, and only Hagen et al. (1999_[34]) considered (at least in principle) all health and environmental impacts of mercury emissions. The coverage of these endpoints will be examined in more detail below.

5.2.1. Neurological effects

Quantifying the IQ impacts of mercury exposure is based in all studies on some combination or variant of the epidemiological studies undertaken in the Faroe Islands, the Seychelles and New Zealand. These studies were arguably rather small (917, 779 and 237 children respectively), and focussed on high fish-consumption (and hence high mercury-exposure) populations. In addition, as pointed out by Gayer and Hahn (2006_[25]), the estimated effect coefficients were in general not highly significant at conventional statistical levels (which could be one reason why integrative analyses such as those by Ryan (2005_[18]) and Axelrad et al. (2007_[19]) were undertaken). They also included different measures of mercury exposure (e.g. cord blood and maternal hair concentrations) and numerous tests of neurological development. As a result, quantification of IQ effects tends to require a large number of supporting assumptions and “translations” between endpoints (even before converting into foregone earnings).

Two of the principal ways in which studies in this review generate different estimates of effect from basically the same data are through the assumptions they make about the functional form of the mercury-IQ exposure-response function and about the level of any threshold. Trasande et al. (2005_[28]) and related studies (some of which were undertaken by co-authors of the epidemiological studies) have assumed a logarithmic functional form as their primary model, preferred by Trasande et al. (2005_[28]) ostensibly on the grounds that it provides a better fit of the data. US EPA (2005_[6]) used a linear model on the recommendation of the National Research Council (2000_[12]), which considered the logarithmic model biologically implausible, as it implies very large marginal effects at very low doses, declining towards zero as dose increases. Using the logarithmic model generates IQ losses from mercury exposure (and hence economic costs) which are much higher than with the linear one, largely due to the effects it predicts for large numbers of individuals at low levels of exposures. Perhaps for this reason, many of the studies which use a logarithmic model also tend to employ effect threshold assumptions, which serve to “filter out” some of these lower-exposed people. Trasande et al. (2005_[28]) estimated a total cost of USD 8.7 billion per year from their “base case” logarithmic model without threshold but USD 3.29 billion (after correction of the effect coefficient (Trasande et al. (2006_[29])) with a threshold. Bellanger et al. (2013_[32]) estimates costs in the EU of methylmercury exposure of just over EUR 9 billion with a linear model, but EUR 39 billion with their logarithmic model. Numbers of births in Europe above their three thresholds ranged from nearly 2 million down to fewer than 250 000.

There also seems to be a tendency for the “damage valuation” studies to adopt dose-response functions from individual studies within the Faroe Islands/Seychelles/New Zealand literature (e.g. Budtz-Jorgensen et al. (2002_[37]), often via Trasande et al. (2005_[28])), whereas the “impact-pathway” studies favour integrations of that literature

(e.g. Axelrad et al. (2007_[19])), the former tending to be associated with exposure-response coefficients possibly three times as high as the latter.¹⁹ Gayer and Hahn (2006_[25]) clearly regard this epidemiological literature as weak, whereas the National Research Council (2000_[12]) described them as “well designed and carefully conducted”. Nevertheless, it is perhaps surprising that a more extensive epidemiological literature is not available to support the assumption that IQ deficits measured in young children extend into adult life and affect earnings and other economic outcomes. Indeed, in a follow-up to the Faroe Islands study, Debes et al. (2016_[76]) found no effects on school performance at 16 years, and only a small effect on educational achievements at age 22 (Ha et al., 2017_[77]). Regarding this shortage of evidence, Ha et al. (2017_[77]) state that:

“In the next few years, there will be an explosion of data generated by over 10 cohort studies being conducted around the world [...] We would expect to obtain more conclusive data on the dose-response relationship between pre-natal exposure and a better understanding of the confounding factors including the nutritional and genetic factors.”

Most studies in this review translated lost IQ estimates into foregone earnings estimates by applying the earnings-IQ relationships estimated by Schwartz (1994_[20]) and/or Salkever (1995_[21]), using some measure of lifetime earnings. This generates a “value per IQ point” which can be applied to the aggregate estimate of the number of IQ points lost. The general approach of valuing IQ point losses seems to imply that there are (extremely marginal) economic effects from even highly subclinical changes in IQ, which might seem unlikely or even unbelievable from a lay perspective, but which has not apparently been questioned in the literature. It is worth noting also that small mean effects at the population level could disguise what are actually larger distinct impacts for a smaller group of individuals in that population. As already mentioned, the approach adopted by Gayer and Hahn (2006_[25]) was different in that it was based on the Agee and Crocker (1996_[67]) analysis of parental decisions to purchase lead-chelation (reduction) therapy for their children, and Lutter’s (2000_[68]) subsequent analysis to translate their results into an implied IQ point value. This produced values quite a lot lower than via Schwartz/Salkever, although comparable with the list of IQ-earnings studies referenced by Gayer and Hahn (2006_[25]).²⁰ A recent review by Salkever (2014_[78]) has argued that many studies of the IQ-earnings relationship which find small effects are based on old data and focus only on wage rates, omitting the impact of IQ on participation and hours worked.

Due to the complexity described in this section and encountered in the reviews above, it is difficult to assess ultimately how well neurological impacts of mercury exposure have been valued in studies in practice. As mentioned, the mercury-IQ epidemiological studies generally used a variety of indicators of neurological development, including IQ but also memory, hand-eye coordination, mood and behaviour disorders, but these were only evaluated at early stages in the child’s life. Rice and Hammitt (2005_[10]) alone attempted

¹⁹ The exception in the impact-pathway literature is Rice and Hammitt (2005_[10]), who used a value generated by Cohen et al. (2005_[81]) on the basis of expert judgment. This was “corrected” in the Rice et al. (2010_[24]) update through the use of Axelrad et al.’s (2007_[19]) value over two thirds lower. Shih and Tseng (2015_[27]) is an “outlier” in continuing to use Rice and Hammitt’s (2005) approach despite those authors’ own revision of it.

²⁰ Perhaps rather surprisingly, Gayer and Hahn’s (2006_[25]) list did not include Schwartz (1994_[20]) or Salkever (1995_[21]).

to account for these non-IQ effects directly, and could only do that by assuming an arbitrary disability weight for these conditions, because no disability weight (or other valuation approach) is available in the literature for these types of conditions, either in childhood or adulthood. Hagen et al. (1999_[34]) valued these non-IQ effects implicitly, without seemingly making it clear to their respondents what exactly they might be, how likely they might be and how seriously they might affect an individual's wellbeing. The only relevant information provided on the human health impacts of mercury exposure was as follows:

“Some health experts are concerned that children may be delayed in walking and talking or suffer reductions in learning abilities similar to those caused by exposure to low levels of lead.”

Clearly, this suggests that participants were given quite incomplete information about the neurological effects of mercury exposure, which makes it difficult to have confidence that their survey responses represent a full and accurate valuation of mercury's health impacts.

Gayer and Hahn (2006_[25]) used the study by Agee and Crocker (1996_[67]) (via Lutter (2000_[68])) for their basis of their valuation. Agee and Crocker's (1996_[67]) study was conducted in the context of child lead exposure, so the accuracy of Gayer and Hahn's (2006_[25]) transfer is dependent on the comparability of the neurological effects of mercury to those of lead. Accepting this, however, Agee and Crocker's (1996_[67]) participants do appear to have been fully informed about lead health impacts, described as follows:

“While the parents filled out a comprehensive medical and social history, the child received a comprehensive battery of neuropsychologic tests. After each examination was completed, a child psychiatrist, Herbert Needleman, M.D., informed the parents of their child's lead status and the consequences of this status. The parents were then counseled at no cost about a medically appropriate course of action.”

The reason for this was at least partly that Agee and Crocker's (1996_[67]) participants were being asked to make real choices about lead reduction treatment for their children, so it was quite appropriate that they should receive full information (or, at least, as full as was reasonable and possible in 1977, when their study was undertaken) about the benefits (and costs) of that treatment. This also means that their responses did not suffer from the hypothetical biases commonly ascribed to responses to stated preference questionnaires (Loomis, 2011_[79]). This could be one reason why the implied value of an IQ point, derived by Lutter (2000_[68]), seems relatively low compared with many of those “constructed” from combining estimates of the IQ-earnings impact (e.g. Salkever (1995_[21])) with estimates of lifetime earnings (e.g. Max et al. (2004_[38])).²¹ Lutter (2000_[68]) provides a number of possible explanations for this, for instance, that parents value their children's welfare less than their own (“parental altruism towards their

²¹ It should be noted that, to make his inference of participants' valuation of neurological impacts, Lutter (2000_[68]) had to make a number of assumptions about parameter values in the link between lead reduction and IQ, and higher implied IQ values could have been obtained from different assumptions. Conversely, although Agee and Crocker's (1996_[67]) participants were effectively valuing all of the neurological impacts of lead exposure (or, all they were aware of and/or were described), Lutter (2000_[68]) assigns all of this value to IQ points specifically, which could lead to an overestimate of that value compared with studies which measure the value of IQ solely in terms of wage (and non-wage) earnings.

children is incomplete”), and parents decline to finance investments in their children’s human capital, even if those investments would be worthwhile from the child’s perspective. In combination with other factors – such as relatively small effect of lead on future earnings, the associated uncertainty of the value of investments in lead reduction, and the expectation that children will be better off anyway due to future economic growth – the result could be a relatively low valuation by parents of treatments of negative child IQ risk factors.

Lutter (2000_[68]) also points out that any policy to reduce current emissions of mercury to obtain future benefits for unborn children effectively represents an intergenerational transfer if the value placed on those benefits is measured from the perspective of those unborn children (as adults) rather than of current taxpayers (or other funders of the policy). This is, in fact, the implication of any regulatory impact assessment (e.g. US EPA (2005_[6]) and, by extension, Canada Gazette (2014_[11]), MJA (2015_[4]) and ECHA (2011_[2]) and (2011_[3])) which uses future earnings as its measure of policy benefits. This might seem a rather arcane point but does serve to illustrate that, although “willingness-to-pay” might routinely be seen as the theoretically correct measure of benefits (Gayer and Hahn (2006_[25])), there is still the question of “whose” willingness-to-pay should count for public policy decisions – the parents of the children affected by the policy, the children who benefit from it, or the taxpayers, consumers and workers who finance it. The issue is further complicated by the fact that, because of pollutant dispersion, many of those who benefit from mercury emission reduction policies might be located outside of the jurisdiction where the reductions are actually being made.

The non-IQ neurological effects of mercury exposure might be so practically insignificant that one might question whether they would have any real impact on subjective wellbeing at all, in which case Rice and Hammitt’s (2005_[10]) assumption of a very minor disability weight to reflect them might be reasonable. If so, IQ-related impacts on earnings might well be the major cost of neurological deficits for most people, and omitting non-IQ impacts might not cause a significant underestimation. On the other hand, if, as Ha et al. (2017_[77]) suggest could be possible, foetal mercury exposure could also be associated with low-birth weight and attention deficit disorder, values for preventing them could be significant, although the evidence to support their inclusion in impact assessments does not currently seem strong enough. From whose perspective earnings and non-earnings impacts should be measured, though, is a moot point.

A small number of studies also consider the possible effects of mercury causing more children to have severe learning difficulties and what they term “mental retardation”. This is apparently a “logical” rather than evidence-based prediction, based on the idea that, if mercury reduces IQ generally, then it also by extension increases the number of children past the threshold for this health state. Rice and Hammitt (2005_[10]) assumed that an IQ below 70 would result in costs of additional schooling with a present value of USD 77 000, but no further impacts, whereas Trasande et al. (2006_[29]) transferred values of around USD 250 000 and USD 1.25 million (present value) which covered life-time costs, including caregiving. Despite the significant difference in value, however, neither study estimated large costs of mental retardation at the population level, due to the low prevalence of this health state even after accounting for the effects of mercury.

5.2.2. Cardiovascular effects

Regarding cardiovascular impacts, the valuation of mortality events through the application of a value of statistical life (VSL) – the approach adopted by the relatively

few studies (except Nedellec and Rabl (2016_[48])) which value this impact – is standard practice, although the range of values available in the literature is broad. The studies by Rice and Hammitt (2005_[10]), Rice et al. (2010_[24]) and Giang and Selin (2016_[26]) all use a VSL which is some variant of the values for mortality risk used by the US EPA. Shih and Tseng (2015_[27]) use a Chinese Taipei-specific value. For the valuation of non-fatal heart attacks, lost productivity (earnings) and treatment costs were included by Giang and Selin (2016_[26]), Rice et al. (2010_[24]) and Rice and Hammitt (2005_[10]), who also included a quality-of-life effect using QALYs (although it was not given a monetary value). Non-fatal heart attacks were not covered by Shih and Tseng (2015_[27]). The approach adopted by Nedellec and Rabl (2016_[48]) was unusual in being based on estimates of the loss of life expectancy associated with cardiovascular impacts (valued using a monetary estimate of the value of a lost life-year). This approach effectively does not distinguish between fatal and non-fatal outcomes, and Nedellec and Rabl (2016_[48]) also exclude productivity and treatment costs associated with non-fatal events. Nevertheless, they find a difference between cardiovascular and IQ values which is comparable to those found in studies based on the VSL – around 80 to 95%.

Clearly, therefore, because of the high relative value of this effect, the primary issue in the treatment of cardiovascular effects is whether the authors in question regard the evidence in support of them as strong enough to warrant inclusion. Even Rice and Hammitt (2005_[10]) admit that they are subject to significant uncertainty (and subject them to probabilistic treatment in their subsequent study, Rice et al. (2010_[24])). It is beyond the scope of this paper to assess this evidence, but it is perhaps worth nothing that, in their 17 page review of the state of the evidence of mercury effect's on human health (Ha et al., 2017), members of the human health plenary panel at a recent international mercury conference devoted a single paragraph to cardiovascular effects, opening with:

“It has been over 10 years since the association of tissue mercury concentrations and cardiovascular outcomes were suggested, but inconsistent outcomes are still reported,”

and finishing with:

“It seems that the cause-effect relationship is far from conclusive and future prospective studies are warranted.”

This suggests that those valuation studies which elect to exclude these effects from their benefits estimations might be justified.

5.2.3. *Chronic mercury poisoning*

As indicated in Section 2 and elsewhere, ASGM is the major global source of anthropogenic mercury emissions, and occurs seemingly exclusively in developing countries. As a result, it is only recently starting to receive analytical attention, but with a commitment in the Minamata Convention to regulate ASGM activities better, a requirement for better evidence of the impacts of mercury emissions related specifically to ASGM is likely in future. Much of the existing evidence relating to IQ and other health effects is relevant, since much of the emissions from ASGM is in the elemental form and hence represents a global pollutant. A better characterisation of the direct health impacts of emissions from ASGM would enable preference-based valuations of those impacts to be estimated.

5.2.4. *Environmental effects*

As the discussion in Section 2 indicates, the environmental effects of mercury exposure are still poorly understood, and the evidence is certainly not yet sufficient to support a quantitative estimation of impacts. As a result, it is not surprising that no damage function-based studies considered in this review have included environmental endpoints. In the absence of strong evidence, one fall-back for valuation purposes is to describe environmental impacts in qualitative terms and let members of the public decide for themselves how much avoiding those impacts would be worth to them. This is effectively the approach adopted by Hagen et al. (1999_[34]). As already discussed, however, the problem with this approach is that reliable stated preference surveys depend on good information being provided to participants. If information provided to them is vague, then so their preferences and monetary valuations will tend to be. This could explain at least why Hagen et al.'s (1999_[34]) results exhibited scope insensitivity, and possibly also the rather high implied values per kg compared with other studies in this review.

The valuation of goods under uncertainty is often characterised by economists in terms of some form of “option value” (e.g. Schmalensee (1972_[80])). Essentially, if impacts on the environment (and other endpoints, such as cardiovascular health) are uncertain, policy analysts are faced with the choice of effectively giving them the value of zero by excluding them – the “conservative” approach generally favoured in socio-economic analysis – or including them, but recognising their uncertainty implicitly (through “vague” values) or explicitly (through probability or other means, cf. e.g. Rice et al. (2010_[24])).

5.2.5. *Impact-pathways*

The review above demonstrates clearly how complex the pathway from mercury emissions to health impacts is, and with it, how complex the majority of studies are which value these impacts. The study by US EPA (2005_[6]) is the most sophisticated in this regard, with detailed models of emissions speciation and deposition by watershed, of fishing behaviour and fish consumption, all of which together determine pre-natal mercury exposure. At the other extreme, Spadaro and Rabl (2008_[11]) assume a simple proportional relationship between mercury emissions and human exposure. There is a major difference across studies in terms of the extent to which emissions are assumed to affect fish caught in different locations. For instance, US EPA (2005_[6]) considered only the consumption of self-caught freshwater fish to be a major source of exposure to methylmercury, whereas Rice and Hammitt (2005_[10]), Trasande et al. (2006_[29]), Shih and Tseng (2015_[27]) and Giang and Selin (2016_[26]) (and Spadaro and Rabl (2008_[11]) implicitly) covered the consumption of all types of fish (freshwater and marine, commercial and non-commercial). According to US EPA (2005_[22]), Rice and Hammitt (2005_[10]) made two strong assumptions: that changes in the concentration of total mercury in the seas are proportional to changes in mercury air deposition, and that this produced proportional changes in methylmercury concentrations in marine fish. It argued that the former assumption does not agree with “some basic physical oceanographic principles”, and that assuming (as US EPA (2005_[6]) does) that marine fish are not significantly affected by domestic emissions is more consistent with the evidence. Although studies like Rice and Hammitt (2005_[10]) do not find a large impact on marine fish mercury concentrations of changes in domestic omissions, these changes affect a large consumer population and hence have significant impacts on economic values. This is illustrated by the difference between the original US EPA (2005_[6]) study and the subsequent upper-bound update (US EPA, 2005_[22]) – the former focussed solely on

recreational freshwater fishing, resulting in a population at risk of around 500 000 people and implied values per tonne of around USD 1 000 per kg, while the latter considered (effectively as no more than a thought experiment) all potential sources of fish, generating a 4 million population at risk and implied values around USD 7 000 per kg.

Note that there is no dispute that United States (“domestic”) emissions of mercury do not affect the methylmercury content of marine fish, or thereby maternal consumption and foetal exposure. All mercury emissions ultimately enter the “global pool” and all methylmercury in fish ultimately comes from that pool. The issue is the extent to which domestic emissions affect marine fish concentrations, and the US EPA (2005_[22]) position has been that that extent is both small and temporally distant. This raises two important questions:

1. To what extent, if at all, should analyses take account of the “external” (international) benefits of emissions reductions?
2. To what extent should the timing of the benefits of emissions reductions be taken into account in socio-economic analyses?

On (1), most studies take a strongly “domestic” line in counting only the “domestic” benefits of emissions reductions, either explicitly (e.g. US EPA (2005_[6]), Rice and Hammitt (2005_[10]) and Gayer and Hahn (2006_[25])) or implicitly (the four regulatory impact assessments, which largely do not recognise or address the transboundary issue). The work by Spadaro and Rabl (2008_[11]) took a global perspective but also estimated their United States value on the basis of the expected domestic and international balance of impacts. Giang and Selin (2016_[26]) estimate that the external benefits to the United States of being part of the Minamata convention outweigh the “internal” (domestic) benefits by between two and three times. But clearly, if domestic policy was based solely on a consideration of domestic benefits, it might not make sense to join a convention like Minamata, and a significant proportion of benefits would be lost.

The question (2) of the timing of impacts has been shown in this review to have an important effect on the values estimated for use in socio-economic analysis. At one extreme, the body of work based on Trasande et al. (2005_[28]) estimates only current costs of exposure and has little to say in practice about the specific benefits of emissions reductions. For unspecified reasons, Rice and Hammitt (2005_[10]) do not account for timings (lags) in their impact-pathway analysis (but do introduce a disease latency adjustment (but not an ecosystem lag) in their subsequent paper (Rice et al. (2010_[24])). ECHA (2011_[2]) model the emissions of mercury from products made using mercury catalysts, taking account of factors such as product service life, but do not account for how long those emissions might take to have health and any other impacts (or of the fact that they expected most releases to be in the “global pollutant”, elemental form of mercury). US EPA (2005_[6]) and (2005_[22]), on the other hand, assumes ecosystem lags of 10-20 years for freshwater ecosystems, and 30 years for marine fish, whilst recognising that marine systems might not even fully adjust for centuries. Giang and Selin (2016_[26]) show that Minamata benefits can range by an order of magnitude (USD 60 – 575 billion) depending on whether a zero lag or 50-year lag is assumed for ocean waters.

It should be noted, however, that the US EPA (2011_[7]) changed its approach, compared with the earlier (2005_[6]) methodology, and assumed no ecosystem lag in its benefits estimates, stating that:

“If a lag in the response of methylmercury levels in fish were assumed, the monetized benefits could be significantly lower, depending on the length of the

lag and the discount rate used [... T]he MMaps approach does not provide any information on the time lag of response.”

The reason for this change in approach is not provided by US EPA (2011_[7]). What does seem true is that ecosystem lags are a feature of the processes by which mercury emissions have health and environmental effects, and assuming that they do not exist will in general overstate the value of the benefits of reductions, possibly significantly (and critically from a policy benefit-cost perspective).

6. Concluding remarks on existing values and future socio-economic analysis

A number of concluding remarks can be made regarding existing studies of the impacts of mercury, the resulting economic value estimates, and their applicability to future socio-economic analysis of mercury policy.

First, the general context of existing studies is undeniably coal-fired electricity generation in the United States. This is the context in which all of the most sophisticated analyses have been undertaken. (In fact, no other OECD member country has undertaken a study even remotely close in analytical rigour or sophistication as those undertaken in the United States.) This has important implications in terms of the parameters and relationships governing speciation, deposition, fish consumption patterns and so on, all of which this review has demonstrated to be relevant to the valuation of mercury emission reductions. Because no similarly sophisticated studies have been conducted in other contexts and countries, it is not clear just how transferable to other contexts the values obtained from these US studies actually is. (Spadaro and Rabl (2008_[11]) examined this issue to an extent but did not publish the full results of their analysis.) Future socio-economic analyses are likely to focus (via Minamata) on non-coal contexts in the developed world and coal (and ASGM) contexts in the developing world. Although mercury is a global pollutant, the value of emission reductions does vary, and possibly significantly, across countries.

Second, there is an important related issue concerning the geographical and temporal coverage of analyses. Depending on speciation (and other factors, such as prevailing winds and the geographical location of sources and population centres), a relatively high proportion of the benefits of mercury emission reductions could be international (“external”). In addition, there is disagreement in the literature about the extent to which marine fish should be a pathway to exposure affected by individual country emissions. Evidence suggests that the time taken for emission reductions to show up in ecosystem and thereafter health impacts could be extended – decades or longer. This implies a need for long time horizons for analysis but also underlines the importance of discounting (although recent US EPA practice (which is important for setting benchmarks on analytical standards) has moved away from acknowledging ecosystem lags but without a clear explanation for the change in position). However, values are being transferred from detailed impact-pathway analyses for use in simpler studies (including governmental regulatory impact assessments) without a clear recognition of these (and other) important factors which affect their transferability. As a result, it is not clear that the transferred

values provide an accurate indicator of the true values of the benefits of emission reductions.

Third, the focus in terms of economic outcomes so far has been on IQ-related earnings loss. Other health and environmental impacts of mercury are possible, but the evidence underpinning them is comparatively weak (or worse) and insufficient to support quantitative analysis. Some studies have included cardiovascular effects which dominate values when they are included, but their existence is very uncertain. Estimating willingness-to-pay for general mercury impacts would be one way of addressing this scientific uncertainty but would itself be subject to significant uncertainty, which would affect the reliability of resulting values. The conclusion is that current state of scientific knowledge about mercury's effects means there are not significant practical gaps in the valuation of endpoints. Gaps in the coverage of endpoints are due to gaps in the underlying science, not to gaps in the economic evidence.

Fourth, some significant variation in values has been observed as a result of studies using different dose-response assumptions (particular the slope and functional form, and whether a threshold is employed). Different assumptions can cause values to increase by an order of magnitude (although variation within impact-pathway studies specifically is smaller). There is a more general issue about the strength of the evidence covering the mercury-IQ-earnings relationship, in particular in terms of establishing the continuation of observed childhood impacts into adulthood, which is currently based largely on assumption. It has been suggested that new evidence on the mercury-IQ (and general cognitive development) connection is likely to be forthcoming over the next few years.

Finally, it is not considered currently possible to make generalisations about the "best values" to be used in future socio-economic analyses (although it is apparent that some existing transfers between studies probably have not generated accurate estimates of value in the contexts transferred to).²² Useful future analysis would undertake a systematic, quantitative assessment of how the various value-relevant parameters affect transferability, and indicate what adjustments might be appropriate to make transfers more accurate.

²² For instance, ECHA (2011_[2]) and (2011_[3]) makes reference to values from Rice and Hammitt (2005_[10]) which arguably were generated using a dose-response slope value which was too high, assuming no ecosystem lag, and a speciation which would result in a higher rate of local deposition than would be the case for the relevant contexts in the EU. This would indicate the Rice and Hammitt (2005_[10]) values exaggerate the true value of the benefits in the ECHA (2011_[2]) and (2011_[3]) studies.

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