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A Note on Trasande et al., "Public Health and Economic Consequences of Methylmercury Toxicity to the Developing Brain" [1]

In 2005, EPA promulgated the Clean Air Mercury Rule (CAMR) to permanently cap and reduce mercury emissions from coal-fired power plants. During the final stages of promulgating this rule, an article was published by Trasande et al. that raised some issues regarding how to measure benefits from reducing mercury. Using one of the models presented by Trasande, we introduce the assumptions that the EPA used in its CAMR analysis and discuss the implication of introducing these assumptions. The impact of introducing all of the EPA assumptions except for those related to discounting would decrease the estimated monetized impact of anthropogenic emissions in the Trasande model by 81% and would decrease the estimated impact of U.S. sources (including power plants) by almost 97%. Including discounting decreases Trasande's estimate of global impacts by 88%, and decreases the impact of American and U.S. power plant impacts by 98%.

Subject Area Classification: Air Pollution; Economic Damages/Benefits; Health

Keywords: Mercury, Methylmercury, Dose-Response, IQ, Fish Consumption, CAMR, Benefits

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On March 15, 2005, EPA promulgated the Clean Air Mercury Rule (CAMR), which is the first Federal rule, and the first one in the world, to permanently cap and reduce mercury emissions from coal-fired power plants. During the final stages of promulgating this rule, an article entitled "Public Health and Economic Consequences of Methylmercury Toxicity to the Developing Brain" (Trasande, Landrigan, and Schecter 2005) was published and raised some issues regarding how to measure certain benefits that society will receive from reducing mercury. Trasande et al. (hereafter, Trasande) analyzed the economic costs of methyl mercury toxicity from anthropogenic mercury emissions, measured as a decrease in IQ. In an upcoming letter (Trasande et al. 2006), they revise their initial estimates and report that the monetized cost of global anthropogenic mercury emissions amount to $7.0 billion (range $0.5 - $13.5 billion, 2000 dollars) per year, of which, they claim, $1.0 billion (range $37 million - $2.0 billion) can be attributed to American power plants. In contrast, the EPA has reported that the upper bound of benefits from reduced IQ decrements, from removing mercury emissions from U.S. power plants after implementing its Clean Air Interstate Rule (CAIR), was $210 million per year. (EPA 2005b, EPA 2006)

While the EPA value does fall in Trasande's range, the difference between Trasande's primary estimate the EPA's estimate is striking. This raises the question if the two values can be compared. Stated briefly, it is impossible to directly compare these two estimates because both the approach and what was being measured are fundamentally different. However, because the economic endpoint analyzed is the same, a comparison of the assumptions used and how they affect the results can be done. Using the Trasande approach, it is possible to illustrate why the assumptions used by the EPA produce a lower estimate and, in our opinion, why the values reported by Trasande are overstated.

For their analysis, Trasande focused on decrements in intelligence quotient (IQ) associated with prenatal mercury exposure using an environmentally attributable fraction model to estimate the damages done by exposure to anthropogenic source of mercury. This model is specified as:

$$\text{Costs} = (\text{Disease rate}) \times (\text{Exposed Population}) \times (\text{Cost per case}) \times (\text{EAF}).$$

The disease rate was derived using one of two dose-response estimates (a logarithmic and a linear estimate) from an epidemiological study of prenatal mercury exposure in the Faroe Islands (Grandjean et al. 1997; Budtz-Jorgensen 2004a). The exposed population is based on data from the National Health and Nutrition Examination Survey (NHANES) (Mahaffey, Clickner, and Bodurow 2004), and the cost per case comes from a reduction in lifetime earnings (based on Max et al. 2002) from reduced IQ (Salkever 1995). The additional EAF is the "environmentally attributable fraction" (Smith, Corvalin, and Kjellstrom 1999), which is a factor to proportionally allocate costs to a particular environmental cause. Since Trasande was concerned with estimating the costs associated with anthropogenic source of mercury, the EAF is simply the portion of mercury which can be attributed to anthropogenic sources, which was set at 70% of total global mercury (EPA 1997b; Mason and Sheu 2002).

Using their chosen parameters, the authors initially reported that the cost to the U.S. of global anthropogenic mercury emissions ranges from \$2.2 to \$43.8 billion in (2000 dollars), with their preferred estimate being \$8.7 billion, based on the logarithmic model (Trasande et al. 2005). Their revised estimates of the costs of global anthropogenic emissions are approximately \$7.0 billion, with a range from \$500 million to \$13.5 billion (Trasande et al. 2006). They further report an estimate of the cost of American anthropogenic emissions by multiplying their global value times a weighted average of U.S. mercury content in all fish. This weighted average was derived by estimating the contribution of U.S. emissions to both domestically caught and imported fish. The authors report that the cost of U.S. anthropogenic emissions range from \$100 million to \$4.8 billion. Finally, the authors report an estimate of the cost of U.S. power plant emissions range from $37 million to $1 billion by multiplying their U.S. cost figures times the percent of American emissions attributable to American power plants.

For CAMR promulgated in March 2005, the EPA used a spatially explicit model of air quality to model the location of mercury deposition from U.S. power plants. Based on models of fishing behavior, the EPA evaluated the benefits from what it considered to be the most important environmental pathway for mercury exposure: prenatal exposure from the consumption of recreationally caught freshwater fish. The EPA reported monetized benefits, measured as reduced decreases in IQ, of implementing CAMR as $0.8 - $3.0 million per year (EPA 1995a). For a number of reasons, including the fact that the EPA estimated the impact from a single pathway for methyl mercury toxicity, the EPA was petitioned to reconsider CAMR. In the technical support document for this reconsideration, the EPA estimated an upper bound on the potential benefits, considering all exposure pathways, which could possibly be obtained from CAMR. This was done by estimating the benefits from removing all remaining mercury emissions from all U.S. power plants after the implementation of CAIR. The EPA's upper bound estimate was $210 million per year (EPA 2005b, EPA 2006).[2]

Because Trasande evaluated the economic costs of IQ decrements due to mercury exposure and the EPA estimated the benefits of reducing IQ decrements due to CAMR, it is tempting to compare these two results. The reasoning is that if Trasande is correctly reporting the cost that U.S. power plants place on society, then this is an estimate of the benefits of CAMR. In fact, it was the very large values initially reported by Trasande and the small values reported by the EPA that made some question the underlying assumptions. A direct comparison between these analyses, however, is not appropriate for a number of reasons.

[2] In the technical support document for the CAMR reconsideration (EPA 2005b), EPA estimated the upper bound benefits to be $168 million per year. This was based on a dose-response estimate that was subsequently revised upward after pubic comment. The revised estimate of $210 million was reported in the response to comments (EPA 2006).

First, Trasande is evaluating the benefits of eliminating *all* anthropogenic mercury, and then parses out how much is attributable to U.S. power plants, whereas CAMR reduces 70% of emissions from U.S. power plants. Under CAMR, coal-fired power plants will be required to reduce emissions from their current level of 48 tons per year to a maximum of 15 tons of mercury per year beginning in 2018, but mercury emissions are not totally eliminated. Furthermore, on March 10, 2005, five days before promulgating CAMR, EPA issued the Clean Air Interstate Rule (CAIR), which was designed to permanently cap emissions of sulfur dioxide (SO_2) and nitrogen oxides (NO_x) from American power plants. One of the additional benefits from CAIR, its so-called "co-benefits", is that the technology used to reduce SO_2 and NO_x will also reduce mercury emissions. Therefore, a portion of the benefits from mercury emissions over the next couple of years will be due to the implementation of CAIR. The correct measure of benefits from CAMR reflects the difference between the state of affairs after the implementation of CAIR and the state of affairs with 15 tons of mercury emissions. This is quite different from the elimination of all mercury emissions.

Second, even if we were to take the proportion of mercury reduced under CAMR as a proportion of the total reported by Trasande, the environmentally attributable fraction model is a relatively simple approach compared to the EPA's economic analysis for CAMR. EPA modeled the location of mercury deposition using a spatially explicit air quality model to assess how contaminated the fish will be and a behavioral model to assess who would be eating them (EPA 2005a). However, this more sophisticated approach was only applied to consumption of recreationally caught freshwater fish. One advantage of the fractional approach used by Trasande is that it can be applied to all exposure pathways, but it does so by assuming that fish contamination levels and consumption patterns are uniform across the U.S. In short, it is impossible to directly compare these two analyses as they use two fundamentally different approaches (the EPA uses a deposition model and Trasande uses a fractional model) and they do not estimate impacts for the same populations (with EPA's analysis addressing a subset of the total because it only estimates the mercury reduction attributable to CAMR).

Third, the Trasande approach does not account for the either the response time in implementing mercury reductions or the response time of the environment to these reductions. As mentioned above, because of prior mercury reductions from CAIR, benefits from CAMR do not begin until the implementation of the 15 ton cap in 2018. Additionally, the ecosystem takes time before reductions in the air deposition of mercury reductions are translated into changes in fish tissue concentration. This environmental response time by itself has been estimated to be on the order of decades before the benefits of mercury reductions are fully realized. In short, the Trasande estimates cannot be construed as a measure of the benefits from regulatory actions to reduce mercury. At best, they would be an estimate of the impact of the *instantaneous* elimination of *all* anthropogenic mercury from the environment.

While the two analyses cannot be directly compared, there may be some utility from understanding the assumptions used by both Trasande and the EPA. A careful, well-reasoned assessment of the current costs imposed by all anthropogenic mercury exposure in the U.S. could serve as a possible starting point for a discussion of the benefits of reducing mercury. In what follows, we discuss the difference in assumptions used by Trasande and the EPA. We then use one of the models presented by Trasande and introduce the assumptions that the EPA used in its CAMR analysis and discuss the implication of introducing these assumptions.

Assumptions

Model choice

The model we use to compare assumptions is Trasande's linear model with a cord/maternal blood ratio of 1.7 and calculated health effects to children whose mother had a blood mercury level of 4.84 μg/L or more. Note that this model is presented in the article as a sensitivity analysis, rather than the primary analysis. Their primary analysis uses a logarithmic model of the dose-response relationship.

We chose this model for the comparisons for two reasons. First, a logarithmic model assumes that there is a supralinear relationship between mercury exposure and IQ decrements. However, in Toxicological Effects of Methylmercury (NRC 2000), the National Research Council recommended use of a linear dose-response relationship and cautioned against using a log-transformed (i.e. supralinear) dose-response function. Based on this recommendation, the EPA analysis used a linear dose-response function in its analysis. Therefore, Trasande's linear model is the only model that is appropriate for a direct comparison. Second, the assumptions of cord/maternal blood ratio of 1.7 and effects to children whose mother had a blood mercury level of 4.84 μg/L or more produce the highest values of all of the linear models. Therefore, our discussion can revolve around the upper bound of the costs of anthropogenic mercury exposure, assuming a linear dose-response relationship.

Dose-response slope for cord blood measurement

In the published version of the paper, Trasande's linear model used an dose-response relationship of 0.59-1.24 IQ point decrements for every 1 μg/L increase in cord blood mercury concentration. In their recent letter (Trasande et al. 2006), the authors revise this value downward by a factor of ten to correct for error in the conversion of the relationship between cord blood and neurodevelopmental effects, as reported in Budtz-Jorgensen (2004a). In the initial version, Trasande began with the assumption that IQ is normally distributed with a standard deviation of 15 points. They then stated

> "The Faroes researchers found that, for those children whose mothers had hair mercury concentrations < 10 μg/g, a 1-μg/L increase of cord blood mercury concentration was associated with adverse impacts on neurodevelopmental tests

ranging from 3.95 to 8.33% of a standard deviation, or 0.59–1.24 IQ points (average = 0.93 IQ points) (Jorgensen [sic.] et al. 2004)."

The 3.95 to 8.33% of a standard deviation range comes from those neurological tests reported Table 2 in Budtz-Jorgensen (2004a) that had statistically significant p-value of 5% or less. However, the Budtz-Jorgensen results were based on a 10 μg/L increase in the cord blood concentration, therefore the values initially reported for the linear model were overstated by an order of magnitude. In their letter with the corrected values, the author's have also revised the mean estimate of the linear dose-response slope to -0.085 IQ points for each 1 ppb of mercury in cord blood.

For its analysis, EPA used a statistical analysis to integrate data from the three major studies investigating the potential neurotoxicity of low-level, chronic mercury exposure: the New Zealand study (Kjellstrom et al. 1989, Crump et al. 1998), the Seychelles Child Development Study (Davidson et al. 1998, Myers et al. 2003), and the Faroe Isands study (Grandjean et al. 1997, Budtz-Jorgensen et al. 2004a). The integrated statistical analysis produced a dose-response relationship with a central estimate of -0.16 IQ points per ppm of mercury in hair (Ryan 2005; EPA 2006)[3]. This implies a relationship of -0.032 IQ points for each 1 ppb in cord blood,[4] substantially lower than the value used by Trasande.

We note that the Trasande value of -0.085 IQ points for each 1 ppb of mercury in cord blood implies a 0.465 IQ decrement for each ppm of mercury in hair. While this value is in the range of what has been found in some studies, it is on the high end. By comparison, Ryan's

[3] The original integrated analysis (Ryan 2005) reported a dose-response relationship with a central estimate of -0.13 IQ points per ppm of mercury in hair. The revised estimate of -0.16 IQ points per ppm of mercury in hair was reported in the response to comments after the public comment for the Reconsideration of CAMR (EPA 2006).

[4] A relationship of -0.16 IQ points for each 1 ppm of mercury in hair means a -1.6e-4 IQ points for each 1ppb of mercury in hair. The median ratio of mercury in hair to mercury in cord blood in the Faroe Islands study is approximately 200 (Budtz-Jorgensen et al. 2004b). Thus, a relationship of -1.6e-4 IQ points for

(2005) evaluation of the Faroe Islands data, the same dataset used by Trasande, indicates a linear dose response relationship of -0.12 IQ points per ppm in maternal hair.

Lifetime Earnings

For both Trasande and the EPA, a decrement in IQ was translated into a decrease in lifetime earnings[5]. Trasande used a value for lifetime earnings (in 2000 dollars) of $1,032,002 for males and $763,468 for females based on Max et al. (2004). These values were derived by starting with the mean annual earnings for full-time, year-round workers (Arias 2002) in five year intervals, adjusted upward by 1.6 for wage supplements (e.g., fringe benefits and employer contribution to insurance benefits), and supplemented with a small additional amount for the imputed value of household production. This earnings figure was then multiplied times a labor force participation rate, based on the percent of the population whose major activity in the past week was working at a job or business, as reported in the 2000 National Health Interview Survey. The earnings were then summed across age intervals, assuming a 3% discount rate and a 1% annual gain in productivity.

The EPA estimated the average present value of future earnings using the total average annual earnings for the population, also in five-year intervals, and broken out by gender, education level as reported in the 1992 Current Population Survey (DOC 1993). The earnings were also summed across age intervals, assuming a 3% discount rate and a 1% annual gain in productivity. EPA reports total lifetime earnings for both sexes combined of $366,021 in 1992 dollars (EPA 2000). Using a GDP deflator, this would imply a value of $472,465 in 2000 dollars.

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each 1ppb of mercury in hair implies -1.6e-4 IQ points for each (1/200) ppm in blood, or -0.032 IQ point for each 1 ppm in blood.

[5] It should be noted that lost earnings from IQ loss is not the conceptually correct metric for valuing benefits of reduced mercury exposure. Ideally, we should use a measure of willingness-to-pay (WTP) to avoid neurobehavioral damage caused by mercury exposure. However, there is currently no acceptable estimate of WTP in the economics literature.

There appear to be two major differences between the EPA's value and the Trasande value. First, the EPA value does not appear to include wage supplements and household production values. Second, the EPA used total average earnings for the population rather than multiplying a participation rate times the earnings for full time workers to arrive at average earnings

Both EPA and Trasande use Salkever (1995) to estimate the effect of a 1 IQ point decrement in earnings. Since the Trasande earnings are gender specific, they use Salkever's gender specific results. For each 1 IQ point decrement, males experience a 1.93% decrease in lifetime earnings and females experience a 3.23% decrease. EPA used a participation-weighted average of 2.379% for the combined lifetime earnings figure. One other important distinction between the two analyses is that, following Salkever, EPA adjusted their dollar value per IQ point for the cost associated with reduced years in schooling, whereas Trasande appear not to have made this adjustment. While this adjustment may be correct, it is difficult to implement in replicating the Trasande et al analysis, so it is not included in future discussions.

Percent of Fish Consumption affected by U.S. Sources

In determining the amount of Hg in fish attributable to U.S. sources, Trasande note that 42 percent of the supply of edible fish in the U.S. is imported (NMFS 2003) and they estimate that 2 percent of the Hg content of imported fish is due to American anthropogenic sources. The remaining 58 percent of the U.S. fish supply is not imported. Using 1995 emissions estimates from the EPA's Mercury Study Report to Congress (1997b), they estimate that 87 tons of Mercury was deposited on U.S. soil in 1995, 60% of which came from U.S. anthropogenic sources. They then attribute this 60% contribution from U.S. sources to the 58% of the domestically caught fish supply. Collecting the assumption for both domestic and imported sources, this implies that approximately 36% of mercury exposure from fish consumption is due to U.S. anthropogenic sources. Using data from the 1999 National Emissions Inventories for

Hazardous Air Pollutants (EPA 2003), they estimate that 41% of the U.S. anthropogenic sources come from U.S. electric power plants. This implies that 15% of all mercury exposure from fish consumption is due to U.S. electric utilities.

There are a number of reasons to question some of these values. The estimate that 58 percent of the U.S. edible fish supply is domestically caught is based on landings. However, marine species comprise approximately 96 percent of the market share of seafood, which includes freshwater and marine fin and shell fish (Carrington, Montwill, and Bolger 2004). Many of these marine species spend at least part of their life cycle in the open ocean, so their mercury content is likely influenced more by the global Mercury pool than by domestic deposition. Another problem can be seen by looking at the location of where the fish are caught. For example, it is highly unlikely that 60% percent of Mercury content in pollock, which has an 11 percent share of the seafood market, is due to U.S. anthropogenic sources. Well over 95 percent of the pollock supply in the U.S. is Alaskan Pollock from the Pacific Ocean. U.S. sources are located east of the Pacific and the prevailing winds in the U.S. are easterly. Similarly, over 90 percent of the cod supplied in the U.S. is Pacific cod.

As mentioned above, in evaluating the impact of mercury emissions, EPA used a spatially explicit air quality model to simulate the location of mercury deposition. This makes a comparison between the Trasande estimate of the amount of fish consumption affected by U.S. sources and EPA's estimate very difficult. At best, the Trasande model can be thought of as a special case of the spatially explicit EPA model, where many or all of the spatially differentiated variables are assumed equal. However, to evaluate the impact of different assumptions, we can use the EPA's estimate of the total mercury deposition in the U.S., as estimated for the CAMR reconsideration. EPA estimated that 144 tons of mercury was deposited in the continental U.S. in 2001, and that 121 (or 84%) came from sources outside of the U.S. and Canada. This means that, on average, 16% of the total mercury deposition in the U.S. comes from American and Canadian sources. Again, this value is an average value and does not reflect the percentage content of mercury in American freshwater fish that can be attributed to American sources, but it does provide an EPA assumption equivalent to that used by Trasande.

To estimate the portion of consumption affected by the domestic deposition of mercury, we use EPA's upper bound estimate in the Technical Support Document for the CAMR reconsideration (2005b). This analysis used the consumption rates from the U.S. EPA's Exposure Factors Handbook (US EPA 1997a), which recommends using a mean consumption rate for the general population of 20.1 grams of fish per day. Of this 20.1 grams of fish per day, 70% (14.1grams) is associated with the consumption of marine fish and 30% (6 grams) is associated with freshwater, estuarine, or aquaculture fish consumption. Following Trasande, we can assume that the 70% is affected by the global pool and 30% is affected by U.S. anthropogenic emissions.

Ecosystem Adjustment

The last large difference in assumptions between the EPA and Trasande is in accounting for the ecosystem response time. As mentioned above, the Trasande analysis could be characterized as an estimate of the economic costs associated with IQ decrements due to anthropogenic mercury exposure, whereas the EPA's analysis is estimating the benefits from reductions in mercury emissions. Also mentioned above, while it is tempting to interpret the Trasande results as estimates of benefits, this would be wrong. A proper economic benefits analysis must account for the timing of the impacts, which in this case involves the response time the ecosystem needs to manifest mercury reductions in fish tissue.

Estimating this response time is difficult because different ecosystems exhibit dramatically different responses to changes in mercury loading depending on their chemical and physical attributes. Among the five freshwater ecosystems investigated by EPA for CAMR, the time required for mercury to reach equilibrium (measured as reaching 90% of its steady state level) after a decrease in mercury loading ranged from less than 5 years to 30 years or more (EPA 2005a). The time required for ocean environments to reach steady state can range from approximately 30 years for the Atlantic Ocean to as much as 200 years for the Pacific Ocean (EPA 2005b). Naturally, benefits will build over time during the transition path from the current conditions to the new equilibrium, but they are not immediate. This transition path can be represented by choosing an average period over which to discount the benefits. For the Technical Support Document for the CAMR reconsideration (2005b), EPA used an average 15 year response lag with a 3% discount rate.

Model Comparison

Table 1 lists the results of Trasande base case linear model, using the corrected dose-response slope, with a cord/maternal blood ratio of 1.7. Each segment of the child-bearing population is assigned a mercury concentration with effects occurring to children whose mother have a blood mercury level of 4.84 μg/L or more (affecting approximately 8% of the population). The change in concentration from a total elimination of mercury exposure is then calculated, assuming a no effect concentration of 3.41 μg/L.

Multiplying the change in concentration times the dose-response slope gives the estimated IQ points lost due to the mercury exposure. Multiplying the estimated lifetime earnings times the percentage change in earnings gives the monetized cost of a 1 IQ point loss for both boys and girls. Multiplying the number of IQ points lost, times the cost of a 1 IQ point lost, times the number of children of each sex, gives the monetized cost of mercury exposure. Multiplying times the EAF, gives the impact of global anthropogenic mercury exposure on U.S. children. Summing across the four segments of the population analyzed produces a base case estimate of approximately $3 billion from the linear model.

Table 1: Trasande's base case linear model of global anthropogenic mercury emissions, using the corrected dose-response slope, with a cord/maternal blood ratio of 1.7

Segment of the Population	92 - 92.1%	92.2 - 94.9%	95 - 99.3%	99.4% and above
Hg concentration range	4.84 - 5.8 ug/L	5.8 - 7.13 ug/L	7.13 - 15.0 ug/L	> 15.0 ug/L
Maternal Hg concentration	4.84	5.8	7.13	15
No Effect concentration	3.41	3.41	3.41	3.41
Change in Concentration	2.431	4.063	6.324	19.703
Dose-Response Slope	0.085	0.085	0.085	0.085
IQ points lost	0.21	0.35	0.54	1.67
Lifetime earnings				
Boys	$1,032,002	$1,032,002	$1,032,002	$1,032,002
Girls	$763,468	$763,468	$763,468	$763,468
Decrease in Lifetime earning for loss of 1 IQ point				
Boys	1.93%	1.93%	1.93%	1.93%
Girls	3.23%	3.23%	3.23%	3.23%
Births				
Boys	45,693	58,155	91,387	12,462
Girls	43,601	55,492	87,201	11,891
EAF	70.00%	70.00%	70.00%	70.00%
Economic Impact				
Boys	$130 Million	$280 Million	$690 Million	$290 Million
Girls	$160 Million	$330 Million	$810 Million	$340 Million
Total	$290 Million	$610 Million	$1.5 Billion	$630 Million

Table 2 provides a comparison of the monetized impact of IQ decrements from anthropogenic mercury emissions under assumptions by the assumptions used by Trasande and by the EPA. The first column of numbers lists the values originally reported by Trasande (2005). The second column of numbers lists the Trasande values with the corrected dose-response coefficient (Trasande et al. 2006). In the second column, the undiscounted monetized impact of anthropogenic emissions is reported to be approximately $3 billion.

Table 2: Comparison of the monetized impact of IQ decrements from anthropogenic mercury emissions under assumptions by Trasande et al. and EPA

	Trasande (Original)	**Trasande (Corrected)**	**EPA**
Monetized Impacts			
Undiscounted Effects			
Monetized Impact of Anthropogenic Emissions	$33 Billion	$3 Billion	$580 Million
Monetized Impact of U.S. Anthropogenic Emissions	$12 Billion	$1 Billion	$35 Million
Monetized Impact of U.S. Power Plant Emissions	$5 Billion	$440 Million	$15 Million
Discounted Effects			
Monetized Impact of Anthropogenic Emissions	$33 Billion	$3 Billion	$370 Million
Monetized Impact of U.S. Anthropogenic Emissions	$12 Billion	$1 Billion	$25 Million
Monetized Impact of U.S. Power Plant Emissions	$5 Billion	$440 Million	$10 Million
Assumptions			
Linear Dose-Response Slope	0.93	0.085	0.032
Male Lifetime Earnings	$1,032,002	$1,032,002	$472,465
Female Lifetime Earnings	$763,468	$763,468	$472,465
Male Earning Loss for 1 IQ Point Decrement	1.93%	1.93%	2.38%
Female Earning Loss for 1 IQ Point Decrement	3.23%	3.23%	2.38%
% of Fish Consumption Affected by Domestic Deposition	58%	58%	30%
% of Fish Consumption Affected by Global Sources	42%	42%	70%
% of Domestic Deposition Attributable to U.S. Sources	60%	60%	16%
% of Global Deposition Attributable to U.S. Sources	2%	2%	2%
% of U.S. Emissions Attributable to U.S. Power Plants	41%	41%	41%
Discount Rate	0%	0%	3%
Average Number of Years for Ecosystem Adjustment	0	0	15

Following the logic in the paper, the impact of U.S. anthropogenic emissions is found by multiplying this $3 billion times the weighted sum of fish consumption affected by U.S. sources. This weighted sum is the percent of fish consumption affected by domestic deposition times the percent of domestic deposition attributable to U.S. sources plus the percent of fish consumption affected by global sources times the percent of global deposition attributable to U.S. sources. By this calculation, U.S. sources have a monetized impact of approximately $1 billion. The impact of U.S. power plants approximately $440 million, found by multiplying the monetized impact of U.S. sources times the percent of U.S. emissions due to U.S. power plants[6]. The discounted effects section of this column simply repeats the undiscounted values for later comparison with

[6] The numbers have been rounded to avoid false precision.

the EPA assumptions. The assumptions section at the bottom lists the assumptions that were introduced to derive these values. The only change between the first and second column of numbers is the linear dose-response slope, which reduces the original values by over an order of magnitude.

The final column of numbers shows what this model would produce if the EPA assumptions were introduced. The new assumptions are listed in the lower section. Using the all of the EPA assumptions except for the discount rate and ecosystem response time in this model gives the undiscounted effects. Using the EPA assumptions, the undiscounted monetized impact of all global anthropogenic emissions would on the order of $580 million, or about 20% of that reported by Trasande. The undiscounted monetized impact of U.S. anthropogenic emissions is on the order of $35 million, and the impact of U.S. power plants is approximately $15 million.

The undiscounted impacts under the EPA column reflect the monetized impact of IQ decrements from anthropogenic mercury emissions. As stated above, they could only be translated into benefits values if the discount rate and time for ecosystem adjustment were included. Introducing the EPA's assumptions of a 3% discount rate and an average 15 year adjustment period produces the discounted effects in the last column. The discounted impact of all global anthropogenic emissions is estimated to be approximately $370 million. Of this, the discounted impact of U.S. anthropogenic emissions is approximately $25 million, with the impact of U.S. power plants estimated to be around $10 million.

As the results in Table 2 show, the impact of introducing all of the EPA assumptions except for those related to discounting would decrease the estimated monetized impact of anthropogenic emissions in the corrected Trasande model by 81% and would decrease the estimated impact of U.S. sources (including power plants) by almost 97%. Including discounting makes the difference even starker, with the EPA assumptions decreasing Trasande's estimate of global impacts by 88%, and decreasing the impact of American and U.S. power plant impacts by 98%. This, however, is taking all the assumptions as a whole. Table 3 illustrates the impact of introducing the EPA assumptions individually. The global estimate column lists the percentage decrease experienced in Trasande's estimated impact of global anthropogenic emissions from introducing the EPA assumptions. The U.S. estimate column lists the percentage decrease in Trasande's monetized impact of U.S. anthropogenic emissions.

One of the larger impacts comes from revising the dose-response curve from the corrected Trasande estimate of -0.085 to the implicit slope of -0.032 from Ryan (2005). As described above, Ryan used a statistical analysis to integrate data from the three major studies whereas Trasande relied on their interpretation of the Faroe Island study. This change alone reduced the estimate of the undiscounted monetized impact of all global anthropogenic emissions by 62%. This change does not affect the estimate of the impact from U.S. anthropogenic emissions in any way other that through its reduction in global impacts.

Table 3: Sensitivity analysis of the impact of EPA assumptions on the Trasande (corrected) results

Assumptions	Impact of the EPA Assumption	
	Global Estimate	U.S. Estimate
Linear Dose-Response Slope	-62%	-62%
Male Lifetime Earnings Female Lifetime Earnings	-46%	-46%
Male Earning Loss for 1 IQ Point Decrement Female Earning Loss for 1 IQ Point Decrement	-4%	-4%
% of Fish Consumption Affected by Domestic Deposition % of Fish Consumption Affected by Global Sources	0%	-46%
% of Domestic Deposition Attributable to U.S. Sources % of Global Deposition Attributable to U.S. Sources	0%	-72%
% of U.S. Emissions Attributable to U.S. Power Plants	Unchanged	
Discount Rate Average number of Years for Ecosystem Adjustment	-36%	-36%
All Assumptions		
Undiscounted Effects	-81%	-97%
Discounted Effects	-88%	-98%

The next model component evaluated is the choice of a lifetime earnings value. The lower value used by EPA reduces the global estimate by 46% and, as with the dose-response curve, does not have any additional effect on the estimate of the impact of U.S. sources. This impact almost exclusively comes from a change in the lifetime earnings value. This earnings value is a base to which the loss associated with a one IQ decrement is multiplied. While Transande uses a gender specific factor and the EPA uses a participation-weighted factor for the whole population, the impact of not using a gender-specific change on both the global and U.S. sources is small, decreasing Trasande's results by approximately 4%.

Changing the percent of fish consumption directly affected by domestic deposition and the percent of fish consumption affected by the global sources does not affect the global estimate (that is, the monetized impact of anthropogenic emissions), but it does affect the U.S. estimate (that is, the monetized impact of U.S. anthropogenic emissions as well as those from U.S. power plants). Estimating the percent of fish consumption affected by domestic deposition based on consumption patterns rather than landings data reduces the estimate of the impact of U.S. sources on U.S. fish consumption by 46%.

The EPA assumption that has the largest impact on the Trasande values is the percent of domestic deposition attributable to U.S. sources. Using its air quality model, EPA estimated that U.S. sources are responsible for 16% of the mercury deposition in the continental U.S., rather than the 60% assumed by Trasande. This change alone reduced Trasande's estimate of the impact of American sources by 72%. As with the percent of fish consumption directly affected by U.S. sources, this change does not affect the global estimate. The percent of emissions attributable to power plants was kept the same for this exercise, so it does not affect the results.

The final change is the discount rate and the average ecosystem response time. Introducing the two EPA assumptions alone decreases the corrected Trasande results by 36%. As previously mentioned, introducing all of the assumptions decreases the undiscounted global impacts by 81% and the U.S. impacts by 97%, and the discounted results by 88% and the U.S. impacts by 98%.

Discussion

This analysis shows that the impact of introducing the EPA assumptions into the Trasande model produces dramatic changes in the monetized impact. In our view, the EPA assumptions are more appropriate than those of Trasande.

The first important decision is model choice. The base case model presented by Trasande is one which assumes a logarithmic dose-response relationship between IQ decrements and mercury exposure. While Budtz-Jorgensen et al. (2004a) did present both a logarithmic model and a linear model for the Faroe Islands results, the National Academy of Sciences' National Research Council explicitly argued against using a supralinear (e.g., logarithmic) model for mercury exposure (NRC 2000). As such, the linear model seems to be the most appropriate model for this analysis.

As can be been from Table 3, the choice of the slope of the dose-response curve is extremely important to the overall results. We believe that a statistical analysis incorporating the data from the three major studies investigating the potential neurotoxicity of low-level, chronic mercury exposure (New Zealand, the Seychelles, and the Faroe Island) is the correct method. Ryan (2005) conducts this integrated analysis and finds a dose-response slope much lower than that of Trasande. It should also be noted that to conduct this integrated analysis, Ryan did reanalyze the Faroe Island data, which is what Trasande relies on, and found a dose-response slope much lower than that reported by Trasande, further supporting our position that their dose-response curve is very high.

Another important difference in assumptions between the EPA and Trasande involves the calculation of lifetime earnings. The EPA used an approach similar to one use that it has used for other rules. This approach uses lifetime earnings for the population as a whole. Trasande relied on Max et al. (2004). While Max et al. did attempt to produce a population level average by multiplying the mean annual earnings for full-time, year-round workers by the percent of the population whose major activity in the past week was working at a job or business, but it is unclear why this approach would be superior to obtaining the population level average as reported in the Current Population Survey. One area where EPA might consider improving its estimate is in the inclusion of wage supplements and non-market household production.

Both analyses include similar approaches to assess the impact on lifetime earnings of a decrement in IQ. While Trasande's include a gender-specific approach that more closely follows Salkever (1995), the EPA's participation-weighted approach produces nearly the same result. (On the other hand, in the economic analysis for CAMR, the EPA includes the impact of that IQ decrements have on the years of schooling which was not included here. While the impact of including this factor is probably very small, it is technically appropriate.)

The last two sets of assumptions addressed here are the percentage of fish consumption affected by domestic and global deposition and the percent of global and domestic deposition affected by U.S. sources. A spatially-explicit model of air quality and deposition is clearly preferable, but this type of modeling is both difficult and expensive. As such, broad assumptions, such as those used here are sometimes necessary. That said, Trasande's particular assumption that 60% of U.S. deposition is due U.S. sources seems implausibly high in light of EPA's air dispersion modeling results, which suggest a figure of approximately 16%. We also believe that the use of consumption data percent of fish consumption affected by global and domestic sources is more accurately estimated using consumption data as opposed to landings data, which ignores some very important location issues.

Finally, we end with three important caveats. First, this analysis evaluates decrements in intelligence quotient (IQ) associated with prenatal mercury exposure and monetizes these results by evaluating changes in lifetime earnings. In this case, IQ is being used as a surrogate for other subtle neurobehavioral endpoints. It does not address any other possible health outcome from mercury exposure (e.g., cardiovascular effects) nor does it address other possible issues associated with IQ decrements such as increased cases of mental retardation. Second, Trasande's analysis includes a threshold for mercury impacts. Prenatal exposure to mercury from mothers who have a blood mercury level less than 4.84 μg/L is estimated to have no impact. The EPA's upper bound estimate of $210 million per year (EPA 2005b) assumed no threshold. All prenatal exposure was assumed to have an impact. This is one of the reasons why introducing the EPA assumptions produced a monetized impact of U.S. power plant emissions of $10 million per year, rather than one closer to the $210 million per year. Finally, while it has been stated a number of times that the results of Trasande cannot be considered an estimate of the benefits of mercury reduction, this is often done. If one were to do so, they must include a measure of the ecosystem response time and a discount rate.

References

Arias, Elizabeth Arias. 2002. "United States Life Tables, 2000". *National Vital Statistics Reports*. 51(3):1-42.

Budtz-Jorgensen E, Debes F, Weihe P, Grandjean P. 2004a. "Adverse Mercury Effects in 7 Year-Old Children as Expressed as Loss in 'IQ'". Odense:University of Southern Denmark. Unpublished manuscript. Available: http://www.chef-project.dk/PDF/iq04louise5.pdf . [accessed April 17, 2006].

Budtz-Jorgensen E, Grandjean P, Jorgensen PJ, Weihe P, Keiding N. 2004b. "Association between mercury concentrations in blood and hair in methylmercury-exposed subjects at different ages". *Environmental Research*. 95:385-393.

Carrington, C.D., Montwill, B., and Bolger, P.M. 2004. "An intervention analysis for the reduction of exposure to methylmercury from the consumption of seafood by women of child-bearing age." *Regulatory Toxicology and Pharmacology*. 40: 272–280.

Crump KS, Kjellstrom T, Shipp AM, Silvers A, Stewart A. 1998. "Influence of prenatal mercury exposure upon scholastic and psychological test performance: benchmark analysis of a New Zealand cohort". *Risk Analysis*. 18:701-713.

Davidson PW, Myers GJ, Cox C, Axtell C, Shamlaye C, Sloane-Reeves J, et al. 1998. "Effects of prenatal and postnatal methylmercury exposure from fish consumption on neurodevelopment: outcomes at 66 months of age in the Seychelles Child Development Study". *JAMA: Journal of the American Medical Association*. 280:701-707.

DOC (U.S. Department of Commerce). 1993. "Money Income of Households, Families, and Persons in the United States: 1992." *Current Population Reports, Consumer Income*. Series P60-184.

EPA (U.S. Environmental Protection Agency). 1997a. *Exposure Factors Handbook: Volume II – Food Ingestion Factors*. EPA/600/P-95/002Fa. Washington, D.C. OAR-2002-0056-5955.

EPA (U.S. Environmental Protection Agency). 1997b. *Mercury Study Report to Congress*. Office of Air Quality Planning & Standards and Office of Research and Development: EPA-452/R-97-003. Available: http://www.epa.gov/mercury/report.htm. [accessed April 17, 2006].

EPA (U.S. Environmental Protection Agency). 2000. *Economic Analysis of Toxic Substance Control Act Section 403: Lead-Based Paint Hazard Standards*. Economic and Policy Analysis Branch; Economics, Exposure and Technology Division; Office of Pollution Prevention and Toxic: Available: http://www.epa.gov/lead/pubs/403_ea_d21.pdf. [accessed April 17, 2006].

EPA (U.S. Environmental Protection Agency). 2003. *1999 Emissions Inventory for Hazardous Air Pollutants: Version 3.* Available: http://www.epa.gov/ttn/chief/net/1999inventory.html. [accessed April 17, 2006].

EPA (U.S. Environmental Protection Agency). 2005a. *Regulatory Impact Analysis of the Clean Air Mercury Rule*. Office of Air Quality Planning and Standards Environmental Air

Quality Strategies and Standards Division: EPA-452/R-05-003. Available: http://www.epa.gov/ttn/atw/utility/ria_final.pdf. [accessed April 17, 2006].

EPA (U.S. Environmental Protection Agency). 2005b. *Technical Support Document: Revision of December 2000 Regulatory Finding on the Emissions of Hazardous Air Pollutants From Electric Utility Steam Generating Units and the Removal of Coal and Oil-Fired Electric Utility Steam Generating Units from the Section 112(c) List: Reconsideration* October 21. Available: http://www.epa.gov/ttn/atw/utility/TSD-112-final.pdf [accessed April 17, 2006].

EPA (U.S. Environmental Protection Agency). 2006. "Response to Significant Public Comments Received in Response to: Revision of December 2000 Regulatory Finding on the Emissions of Hazardous Air Pollutants From Electric Utility Steam Generating Units and the Removal of Coal- and Oil-Fired Electric Utility Steam Generating Units from the Section 112(c) List: Reconsideration (70 FR 62200; October 28, 2005) and Standards of Performance for New and Existing Stationary Sources: Electric Utility Steam Generating Units: Reconsideration (70 FR 62213; October 28, 2005)" Forthcoming.

Grandjean P, Weihe P, White RF, Debes F, Araki S, Yokoyama K, et al. 1997. "Cognitive Deficit in 7-Year-Old Children with Prenatal Exposure to Methylmercury." *Neurotoxicology and Teratology*. 19(6):417-428.

Kjellstrom T, Kennedy P, Wallis P, Mantell C. 1989. Physical and mental development of children with prenatal exposure to mercury from fish. Stage 2: Interviews and psychological tests at age 6, Report 3642. Solna, Sweden: National Swedish Environmental Protection Board.

Mahaffey, Kathryn., Robert P. Clickner, and Catherine C. Bodurow. 2004. "Blood Organic Mercury and Dietary Mercury Intake: National Health and Nutrition Examination Survey, 1999 and 2000" *Environmental Health Perspectives*. 112(5): 562-70.

Mason, R. P. and G.-R. Sheu. 2002. "Role of the ocean in the global mercury cycle". *Global Biogeochemical Cycles*. 16(4), 1093.

Max, Wendy, Dorothy P. Rice, Hai-Yen Sung, and Martha Michel. 2004. "Valuing Human Life: Estimating the Present Value of Lifetime Earnings, 2000" Center for Tobacco Control Research and Education. Economic Studies and Related Methods. Paper PVLE2000. Unpublished manuscript. Available: http://repositories.cdlib.org/ctcre/esarm/PVLE2000. [accessed April 17, 2006].

Myers GJ, Davidson PW, Cox C, Shamlaye CF, Palumbo D, Cernichiari E, et al. 2003. Prenatal methylmercury exposure from ocean fish consumption in the Seychelles child development study. *The Lancet* 361:1686-1692.

NMFS (National Marine Fisheries Service). 2003. *Fisheries of the United States: 2002*. Silver Spring, MD: National Marine Fisheries Service, Office of Science and Technology, Fisheries Statistics and Economics Division.

NRC (National Research Council). 2000. *Toxicological Effects of Methylmercury*. Washington, D.C.: Committee on the Toxicological Effects of Methylmercury, Board on Environmental Studies and Toxicology, Commission on Life Sciences, National Research Council. National Academy Press.

Ryan LM. 2005. Effects of Prenatal Methylmercury on Childhood IQ: A Synthesis of Three Studies. Report to the U.S. Environmental Protection Agency. Available: http://www.regulations.gov [Document ID EPA-HQ-OAR-2002-0056-6048 and EPA-HQ-OAR-2002-0056-6049.]. [accessed January 20, 2006].

Salkever, David S. 1995. "Updated Estimates of Earnings Benefits from Reduced Exposure of Children to Environmental Lead" *Environmental Research*. 70:1-6.

Smith, Kirk R., Carlos F. Corvalán, and Tord Kjellström. 1999. "How Much Global Ill Health Is Attributable to Environmental Factors?" Epidemiology. 10(5): 573-84.

Trasande, Leonardo; Philip J. Landrigan; and Clyde Schechter. 2005. "Public Health and Economic Consequences of Methyl Mercury Toxicity to the Developing Brain" *Environmental Health Perspectives*. 11(5): 590-6.

Trasande, Leonardo; Philip J. Landrigan; and Clyde Schechter. 2006. Letter to the Editor of *Environmental Health Perspectives*. Forthcoming.