Chapter VII: Benefit-Cost Analysis

While relative cost-effectiveness is the principal economic policy criterion established for potential Tier 2 standards in the Clean Air Act, further insight regarding the merits of the proposed standards can be provided by benefit-cost analysis (BCA). In its traditional application, BCA estimates the economic "efficiency" of proposed standards by defining and quantifying the various expected consequences and representing those consequences in terms of dollars. Expressing the effects of the potential standards in dollar terms provides a means for comparing the expected benefits of our proposed standards to the expected costs.

The basic question we sought to answer in the BCA was: "What are the net yearly economic benefits to society of the reduction in mobile source emissions likely to be achieved by today's proposed standards?" In designing an analysis to answer this question, we adopted an analytical structure and sequence similar to that used in the so-called "Section 812 studies"[a] to estimate the total benefits and costs of the entire Clean Air Act. Moreover, we used many of the same data sets, models, and assumptions actually used in the Section 812 studies and/or the recent Regulatory Impact Analyses (RIAs) for the PM and Ozone NAAQS, and the NO_x SIP Call.[b] By adopting the major design elements, data sets, models, and assumptions developed for recent RIAs, we have largely relied on methods which have already received review by other Federal Agencies, and the public. Furthermore, the data sets adopted from the Section 812 studies have received extensive review by the independent Science Advisory Board and the public.

The BCA that we performed for our proposed standards can be thought of as having four parts, each of which will be discussed separately in the Sections that follow. These four steps are:

1. Calculation of the impact that our proposed standards will have on the nationwide inventories for NOx, NMHC, SO_2, and PM.

2. Air quality modeling to determine the changes in ambient concentrations of various pollutants that will result from our proposed standards.

3. A benefits analysis to determine the changes in human health and welfare, both in terms of number of incidences and monetary value, that result from the changes in

[a] The "Section 812 studies" refers to (1) USEPA, Report to Congress: The Benefits and Costs of the Clean Air Act, 1970 to 1990, October 1997 (also known as the "Section 812 Retrospective); and (2) the first in the ongoing series of prospective studies estimating the total costs and benefits of the Clean Air Act, expected to be published later in 1999.

[b] "Regulatory Impact Analysis for the NO_x SIP Call, FIP, and Section 126 Petitions" September 1998, EPA-452/R-98-003

ambient concentrations of various pollutants.

4. Calculation of the costs of our proposed standards for purposes of comparison to the monetized benefits.

Our BCA does contain a number of limitations common to all BCAs. Critical limitations on the availability, validity, or reliability of data; limitations in the scope and capabilities of environmental and economic effect models; and controversies and uncertainties surrounding key underlying scientific and economic literature all contribute to an inability to estimate the economic effects of environmental policy changes in exact and unambiguous terms. Under these circumstances, we consider it most appropriate to view BCA as a tool to inform, but not dictate, regulatory decisions such as the ones reflected in today's proposal. The limitations of the assessment of benefits will be discussed in each of the following Sections as appropriate.

Despite these important uncertainties, we believe the preliminary BCA is indicative of the range of benefits and costs associated with the standards proposed today. This is because the analysis focuses on estimating the economic effects of the changes in air quality conditions expected to result from today's proposed rules, rather than focusing on developing a precise prediction of the absolute levels of air quality likely to prevail at some particular time in the future. An analysis focusing on the changes in air quality can give useful insights into the likely economic effects of emissions reductions of the magnitude expected to result from today's proposed rule.

A. Emissions

In order to determine the air quality impact of our proposed standards, we first calculated the reductions in vehicle emissions that are expected to occur as a result of those standards, and then determined the impact of those emission reductions on the nationwide[c] inventories for NOx, NMHC, SO2, and PM. This Section describes how these inventory impacts were determined.

Our analysis used the Section 812 post-CAAA scenario for 2010 as the baseline emission estimates. This baseline inventory was also used to produce the control inventory through the application of the estimated changes in emissions associated with our proposed Tier 2 rule.[d] We also updated the fugitive dust PM_{10} and $PM_{2.5}$ emissions for the Section 812 inventory using the National Pollutant Inventory in order to reflect significant changes to the base year methodologies for fugitive dust categories. These changes reduced the estimates of primary PM

[c] For the purposes of air quality modeling, 'nationwide' is taken to mean the contiguous 48-states. Also, the proposed Tier 2/gasoline sulfur standards are assumed to have no effect on vehicle emissions in California, though air quality in California may be affected through meteorological boundary conditions.

[d] Preparation of the baseline inventory is described in some detail by Woolfolk et al. (1998). E.H. Pechan (1999) provide emissions data reflecting the incorporation of the Tier 2 rule.

emissions. Fugitive dust PM_{10} and $PM_{2.5}$ emitters whose 1990 emissions estimates were revised include agricultural tilling, paved and unpaved roads, prescribed burning, construction activity, and wind erosion.

The Tier 2/gasoline sulfur program we are proposing has various emission-related components which begin at various times and in some cases phase in over time. This means that during the early years of the program there will not be a consistent match between costs and benefits. This is due to the fact that the full vehicle cost is incurred at the time of vehicle purchase, while the fuel cost along with the emission reductions and benefits occur throughout the lifetime of the vehicle. In order to more appropriately match the costs and emission reductions of our proposed program, therefore, our BCA assumes some future year when the fleet is fully turned over. For today's proposal this stability does not occur until well into the future. However, for the purpose of the benefit calculations, we have no available baseline data set beyond the year 2010, since the Section 812 inventory was developed only for this year. We have therefore made adjustments to allow the use of 2010 as a surrogate for a future year in which the fleet consists entirely of Tier 2 vehicles.

For emissions, we calculated reductions by treating 2010 as if the fleet had already turned over. We did this by applying the control case emission reductions from a fully turned over fleet (for the year 2040) to the fleet mileages for this year. Clearly, this approach does not, nor is it intended to, predict actual expected emission reductions for 2010. This is not its purpose. It is intended to portray the characteristics of the vehicle fleet after it is fully turned over, within the constraint that 2010 was the latest year for which we could perform the analysis.

The resulting analysis represents a snapshot of benefits and costs in a future year in which the light-duty fleet consists entirely of Tier 2 vehicles. As such, it depicts the maximum emission reductions (and resultant benefits) and among the lowest costs that would be achieved in any one year by the program on a "per mile" basis. (Note, however, that net benefits would continue to grow over time beyond those resulting from this analysis, but only because of growth in vehicle miles traveled.) Thus, based on the long-term costs for a fully turned over fleet, the resulting benefit-cost ratio will be close to its maximum point (for those benefits which we have been able to value).

At the time that we undertook the development of the benefit estimates for this rule, we did not have quantitative estimates of the VOC emission reductions that would result from the evaporative emission standards in the proposal. Therefore, the benefit estimates do not include the value of the evaporative emission standard. Consistent with this, the program cost estimates also exclude the evaporative emission control cost. Since the evaporative emission reductions and costs are both relatively small compared to the rest of the program, they are not expected to significantly affect the overall cost-benefit ratio.

For the purposes of assessing benefits, we estimated that the proposed Tier 2/gasoline sulfur standards would reduce NMHC emissions by 214,443 tons and NO_x emissions 1,789,318 tons for a hypothetical fully turned-over fleet of light-duty gasoline vehicles and trucks (Korotney, 1998). These reductions would occur in all States except California, which already meets this standard. Measured from the Section 812 2010 post-CAA scenario emission estimates for highway vehicles, these reductions translate into a 6.1 percent and 48.9 percent annual reduction in VOC and NO_x emissions, respectively, for these States and vehicle categories. These percent reductions were used to estimate the 47-State VOC and NO_x emission reductions from light-duty vehicles and trucks in every county. Reductions (based on percent VOC) were also used to estimate the soluble organic aerosol (SOA) emissions.

A reduction to the SO_2 inventory was also made to account for expected gasoline sulfur reduction. SO_2 reductions are based on reducing the gasoline sulfur content from 330 parts per million (ppm) to 30 ppm. There are some uncertainties introduced by the SO_2 emission estimation methods. For one, the baseline emission estimates do not account for the lower sulfur levels in Federal or California reformulated gasoline. Thus, the baseline emission estimates likely overestimate gasoline vehicle emitted SO_2 nationwide by about 10 percent (in comparison to the combined conventional + reformulated gasoline baseline sulfur of 305 ppm, as described in Section VI.A.2), and in California by a factor of 10 (in comparison to their average sulfur limit of 30 ppm). These differences are expected to have only a modest impact on SO_2 benefits attributed to the Tier 2 rule, however, because no motor vehicle SO_2 benefit was estimated for California, and 47 State benefits are only slightly overstated in Federal reformulated gasoline areas.

Table VII-1 summarizes the emissions inventories in the 47 contiguous states for both the baseline and control scenarios.

Table VII-1. Emission Estimates by Vehicle Type and Reductions Associated with Adoption of the Tier 2 Rule

Continental U.S. minus California -- Section 812 2010 CAA Highway Vehicle Emissions								
Vehicle type	*VOC*	*NO_x*	*CO*	*SO_2*	*PM_{10}*	*$PM_{2.5}$*	*SOA*	*NH_3*
Light duty gas vehicle	2,197,781	2,296,033	22,746,343	153,912	65,117	37,491	13,406	271,483
Light duty gas truck 1	743,149	750,514	7,681,457	55,797	20,062	12,010	4,533	70,314
Light duty gas truck 2	574,236	609,133	5,947,424	28,430	10,072	6,095	3,503	32,084
Heavy duty gas vehicle	136,919	272,760	1,526,289	12,416	5,840	3,837	1,000	2,564
Motorcycle	40,697	14,467	221,551	396	453	227	248	45
Light duty diesel vehicle[a]	7	24	22	0	0	0	0	0
Light duty diesel trucks	365	870	819	58	65	50	9	1
Heavy duty diesel vehicle	139,013	1,297,002	2,123,937	107,054	78,764	65,856	3,295	439
2010 baseline emissions	3,832,166	5,240,802	40,247,842	358,062	180,372	125,566	25,994	376,930
Reductions due to Tier 2 rule	214,443	1,789,318	0	228,137	0	0	1,308	0
47-state emission estimates under Tier 2 rule	3,617,723	3,451,484	40,247,842	129,925	180,372	125,566	24,686	376,930
California emissions								
Light duty gas vehicle	65,841	106,110	965,593	24,105	10,198	5,872	402	42,528
Light duty gas truck 1	17,450	33,335	304,932	8,177	2,936	1,752	106	10,302
Light duty gas truck 2	8,756	22,425	154,846	4,167	1,474	892	53	4,701
Heavy duty gas vehicle	5,250	23,561	112,979	1,635	769	507	38	338
Motorcycle	3,647	2,030	24,311	60	68	37	22	7
Light duty diesel vehicle	2	7	11	0	0	0	0	0
Light duty diesel trucks	39	146	198	13	14	11	1	0
Heavy duty diesel vehicle	12,740	145,980	84,364	13,013	9,599	8,030	302	53
California emissions for baseline and under Tier 2	113,725	333,595	1,647,234	51,170	25,059	17,101	925	57,930
48-state emission estimates for control scenario	3,731,448	3,785,079	41,895,076	181,095	205,431	142,667	25,611	434,860

[a]Future year emissions of SO_2, PM_{10}, $PM_{2.5}$, SOA, and NH_3 from light-duty diesel vehicles are projected to be zero due to low projected vehicle mile traveled (VMT) levels.

B. Air Quality Impacts

EPA has used a regional-scale version of the Urban Airshed Model (UAM-V) to estimate ozone air quality. Our analysis uses a Source-Receptor Matrix (S-R Matrix) based on the Climatological Regional Dispersion Model (CRDM) is used to estimate nitrogen deposition, PM air quality, and visibility degradation.

Section VII.B.1 covers the estimation of ozone air quality using UAM-V. Section VII.B.2 covers the estimation of particulate matter air quality, and Section VII.B.3 discusses the estimation of nitrogen deposition. Finally, Section VII.B.4 covers the estimation of visibility degradation.

1. Ozone Air Quality Estimates

EPA used the emissions inputs with a regional-scale version of UAM-V to estimate ozone air quality. Because it accounts for spatial and temporal variations as well as differences in the reactivity of emissions, the UAM-V is useful for evaluating the air-quality effects of the Tier 2 rule.[e]

Our analysis applies the modeling system for a base-year of 1990 and for two future-year scenarios: a 2010 baseline and a control scenario. As discussed later, we used the two separate years because ambient air quality observations from 1990 are used to calibrate the model. The UAM-V modeling system requires a variety of input files that contain information pertaining to the modeling domain and simulation period. These include gridded, day-specific emissions estimates and meteorological fields, initial and boundary conditions, and land-use information.

The model divides the U.S. into two regions: East and West. The model then segments the area in each region into grids, each of which has several layers of air conditions that are considered in the analysis. Using this data, the UAM-V model generates predictions of hourly ozone concentrations for every grid. We then used the results of this process to develop 2010 ozone profiles at monitor sites by applying derived adjustment factors to the actual 1990 ozone data at each monitor site. For areas (grids) without ozone monitoring data, we interpolated ozone values using data from monitors surrounding the area. After completing this process, we calculated daily and seasonal ozone metrics as inputs for the health and agriculture benefits analysis. The Sections below provide a more detailed discussion of each of the steps in this evaluation and a summary of the results.

[e] Douglas and Iwamiya (1999) provide further information on the UAM-V modeling used in this analysis.

a. Modeling Domain

For the eastern U.S., the domain is the same as the eastern U.S. domain used in EPA's (1998b) recent analysis, "Regulatory Impact Analysis for the NO_x SIP Call, FIP, and Section 126 Petitions." The domain encompasses most of the eastern U.S. and consists of two grids, as illustrated in Figure VII-1. The shaded area of Figure VII-1 uses a relatively fine grid of 12 km, which consists of seven vertical layers. The unshaded area of Figure VII-1 has less resolution, as it uses a 36 km grid, which consists of five vertical layers. The top of the modeling domain is 4000 meters above ground level, for both the shaded and unshaded regions.

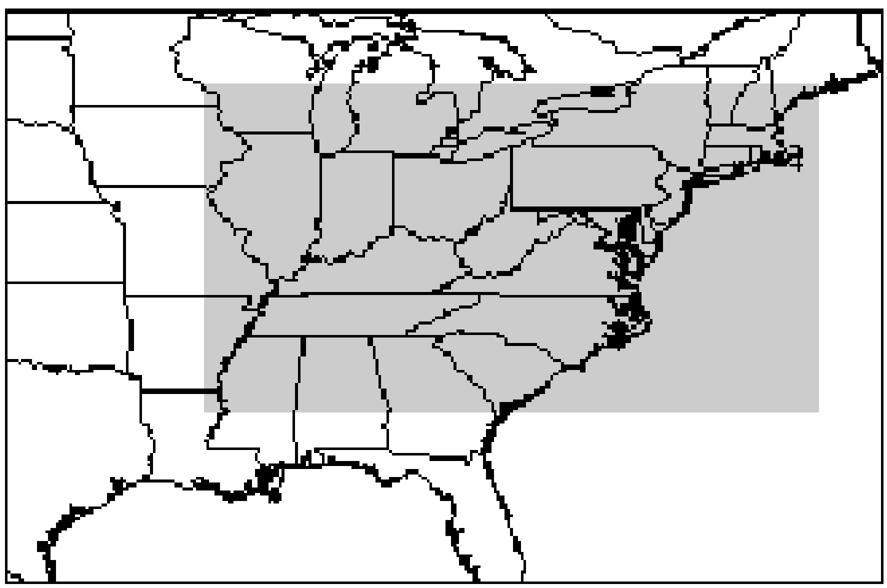

Figure VII-1. UAM-V Modeling Domain for Eastern U.S.

The modeling domain used to obtain results for the western U.S. comprises the contiguous 48 states. Note that although the domain includes the entire contiguous 48 states, results using this domain configuration were only used to estimate the effects of the Tier 2 rule in the West (defined as the region not shown in Figure VII-1). The domain extends from 126 degrees west longitude to 66 degrees west longitude, and from 24 degrees north latitude to 52 degrees north latitude. The analysis used a grid cell size of 2/3 longitude by ½ latitude (approximately 56 by 56 km) resulting in a 90 by 56 grid (5,040 cells) for each vertical layer, with eight vertical layers in all.

b. Simulation Periods

A simulation period is generally characterized by high ozone concentrations in one or more portions of the U.S.; exceedances of the 1-hour National Ambient Air Quality Standard for ozone were recorded at monitors during these periods. This study used three multi-day simulation periods to prepare the future-year ozone profiles. For the eastern U.S. ozone analysis, we modeled two simulation periods: 20-30 July 1993 and 7-18 July 1995. For the western U.S. the simulation period was 1-10 July 1990.

c. UAM-V Model Output

Standard output from the UAM-V modeling system includes: (1) hourly, surface-layer ozone concentrations (provided as hourly averages) for each grid cell; and (2) instantaneous ozone values for all grid cells and layers for each hour of the simulation. This study extracted hourly, surface-layer ozone concentrations for each grid-cell from the file containing hourly average ozone values. We then used this information to calculate a set of adjustment factors for forecasting 2010 ozone concentrations, as described in the following Section.

d. Converting Episode Estimates to Full-Season Profiles

The UAM-V runs generate surface layer hourly average ozone concentration estimates for the limited modeled episodes which are used in conjunction with actual 1990 concentrations to generate ozone concentrations for the entire ozone season.[f] We mapped individual monitors onto the gridded UAM-V output, and used the modeled concentrations of the corresponding grid cells to calculate an adjustment factor.

We multiplied hourly ozone concentrations for 1990 by the adjustment factors to estimate 2010 ozone concentrations. Using the calculated adjustment factors and the observed monitor concentrations, we created a data set containing modified observed hourly ozone concentrations for each of the two scenarios. The Technical Support Document for this analysis details the steps involved.

[f] The five-month ozone season for this analysis is defined as May to September for health benefits. For agricultural benefits for some crops, the relevant growing season extends into April and into October and November. In this analysis, no changes in ozone concentrations are assumed to occur outside the five-month ozone season. However, the ozone metric used to estimate certain crop yield benefits requires that the baseline level of ozone concentrations be estimated for months outside the five-month ozone season.

e. Extrapolating from Monitored to Unmonitored Locations

To model whole U.S., we needed ozone data for every location. Since actual ozone data is only available from limited monitor sites, we needed a method to extrapolate to unmonitored locations, in order to estimate the effects of several ozone-related health and welfare effects.[g] Given available ozone monitoring data, we obtained ozone measures (e.g., daily average) for each location in the contiguous 48 states in two steps: (1) we converted hourly data to an ozone measure of interest, such as the daily average, and (2) we used monitor-specific ozone measures to extrapolate ozone measures to a grid of eight km by eight km population grid-cells. The conversion from hourly data to ozone measures of interest is straightforward. The estimation of ozone measures at each grid-cell uses a Voronoi Neighbor Averaging (VNA) spatial interpolation procedure.[h]

The VNA procedure interpolates air quality estimates from the monitors to the center of each population grid-cell. The VNA procedure is a generalization of planar interpolation. Rather than limit the selection of monitors to, say, three, VNA identifies the set of monitors that best "surrounds" the center of each grid-cell. The result of VNA is illustrated in Figure VII-2. VNA determines the set of monitors that best surround the grid-cell by identifying which monitor is closest (considering both angular direction and horizontal distance) in each direction from the grid-cell center. Each selected monitor will likely be the closest monitor for multiple directions. The set of monitors found using this approach forms a polygon around the grid-cell center.

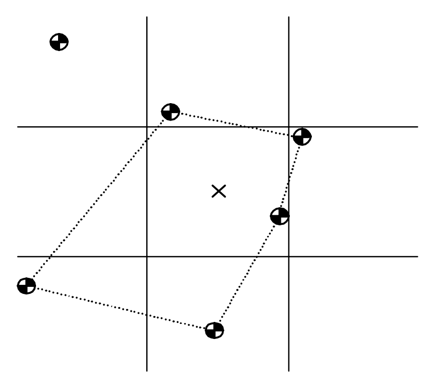

Figure VII-2. VNA Spatial Interpolation

The analysis of ozone impacts on agriculture adjusts the VNA approach slightly. Because calculating the benefits for this welfare category is best accomplished by using air quality data at

[g]The Technical Support Document (Abt Associates, 1999) has a map of the location of ozone monitors in the U.S. The map shows that some areas of the country do not have many ozone monitors in close proximity to each other.

[h]Interpolation between monitors is conducted using the same method as used by Abt Associates (1998) for the NO_x SIP call analysis; previously termed the "convex polygon" method, it is more accurately described as Voronoi Neighbor Averaging (VNA) spatial interpolation, which will be used throughout this document.

the county level, we used the VNA approach to estimate ozone measures for the center of each county, rather than the eight km by eight km population grid-cell level. To provide estimates for all counties, the analysis includes monitors that are up to 400 km from a county centroid. (Using a shorter distance would result in some county centroids not receiving an estimate.)

f. Ozone Air Quality Results

A summary of the ozone air quality profiles used to assess the benefits of the proposed standards is presented in Table VII-2. The change in seasonal ozone values across the U.S. ranges from an increase of 0.0016 ppm to -0.0028 ppm from the base case to the control, with a spatial average of -0.0008 ppm. The population-weighted average change is somewhat lower, -0.0004 ppm, which reflects that urban regions have smaller reductions in ozone than less populated rural regions. The air quality technical support document for this Regulatory Impact Assessment (RIA) (Abt Associates, 1999) contains maps showing the base case ozone concentrations and ozone concentration changes for the control scenario. These maps only convey information about the five-month ozone season used for the health benefits analysis. The change in the ozone index used in the agriculture analysis (termed “SUM06" and defined in Table VII-2) ranges from -0.0132 to 0.0087 ppm, with a spatial average of -0.0025 ppm and a population weighted average of -0.0026 ppm.

Table VII-2. Summary of UAM-V Derived Hourly Ozone Air Quality

Statistic	*2010 Base Case*[a]	*Change*[a]	*Percent Change*
Seasonal Average			
Minimum (ppm) [b]	0.0168	- 0.0028	-16.7%
Maximum (ppm) [b]	0.0611	0.0016	2.6%
Spatial Average (ppm)	0.0305	- 0.0008	-2.6%
Population-Weighted Average (person-ppm) [c]	0.0302	- 0.0004	-1.3%
Seasonal SUM06[d]			
Minimum (ppm) [b]	0.0000	- 0.0132	0.0%
Maximum (ppm) [b]	0.1052	0.0087	8.3%
Spatial Average (ppm)	0.0122	- 0.0025	-20.5%
Population-Weighted Average (person-ppm) [c]	0.0193	- 0.0026	-13.5%

[a] All values are calculated at the county centroid, using VNA spatial interpolation and allowing all monitors with a maximum distance of 400 km. The change is defined as the control case value minus the base case value.

[b] The base case minimum (maximum) is the value for the county with the lowest (highest) seasonal average, where the season is defined as May through September and all hours are included in the calculation. The change relative to the base case picks the minimum (maximum) from the set of changes in all counties.

[c] Calculated by summing the product of the projected 2010 county population and the estimated 2010 county centroid seasonal (or SUM06) ozone concentration, and then dividing by the total population.

[d] SUM06 is defined as the cumulative sum of hourly ozone concentrations over 0.06 ppm that occur from 8am to 8pm in the months of May through September.

2. PM Air Quality Estimates

Changes in concentrations of PM have an important effect on people's health and welfare. Our analysis uses the S-R Matrix model to evaluate the air quality effects of the Tier 2 rule. The S-R Matrix reflects the relationship between annual average PM concentration values at a single receptor in each county (a hypothetical monitor sited at the county population centroid) and the contribution by PM species to this concentration from each emission source (E.H. Pechan, 1996). The modeled receptors include all U.S. county centroids plus receptors in ten Canadian provinces and 29 Mexican cities/states. The methodology used in this RIA for estimating PM air quality concentrations using the S-R Matrix is similar to the method used in the July 1997 PM and Ozone NAAQS RIA (U.S. EPA, 1997e). Below is a detailed discussion of the steps taken to run the S-R Matrix and to derive the resulting changes in PM air quality.

a. Climatological Regional Dispersion Model

The CRDM uses assumptions similar to the Industrial Source Complex Short Term model (ISCST3), an EPA-recommended short range Gaussian dispersion model. CRDM incorporates terms for wet and dry deposition and chemical conversion of SO_2 and NO_x to PM, and uses climatological summaries (annual average mixing heights and joint frequency distributions of wind speed and direction) from 100 upper air meteorological sites throughout North America. The analysis used meteorological data for 1990 coupled with emissions data from version 2.0 of the 1990 National Particulate Inventory to develop the S-R Matrix.

b. Development of the S-R Matrix

To develop the S-R Matrix, we modeled a nationwide total of 5,944 sources (i.e., industrial point, utility, area, nonroad, and motor vehicle) of primary and precursor emissions with CRDM. In addition, we modeled secondary organic aerosols formed from anthropogenic and biogenic VOC emissions, as well as natural sources of PM_{10} and $PM_{2.5}$ (i.e., wind erosion and wild fires). We modeled emissions of SO_2, NO_x, and ammonia in order to calculate ammonium sulfate and ammonium nitrate concentrations, the primary particulate forms of sulfate and nitrate. The CRDM produced a matrix of transfer coefficients for each of these primary and precursor emissions. These coefficients can be applied to the emissions of any unit (area source or individual point source) to calculate a particular source's contribution to a county receptor's total annual average PM_{10} or $PM_{2.5}$ concentration. Each individual unit in the inventory is associated with one of the modeled source types (i.e., area, point sources with effective stack height of 0 to 250 m, 250 m to 500 m, and individual point sources with effective stack height above 500 m) for each county.

The relative concentrations in the atmosphere of ammonium sulfate and ammonium nitrate depend on complex chemical reactions. In the presence of sulfate and nitric acid (the gas phase oxidation product of NO_x), ammonia reacts preferentially with sulfate to form particulate ammonium sulfate rather than react with nitric acid to form particulate ammonium nitrate. We adjusted the S-R Matrix transfer coefficients to reflect concentrations of secondarily-formed particulates (Latimer, 1996). First, we multiplied the transfer coefficients for SO_2, NO_x, and ammonia by the ratios of the molecular weights of sulfate/SO_2, nitrate/nitrogen dioxide and ammonium/ammonia to obtain concentrations of sulfate, nitrate and ammonium.[i] Ammonium nitrate forms under conditions of excess ammonium and low temperatures. For each county receptor, the sulfate-nitrate-ammonium equilibrium is estimated based on the following simplifying assumptions:

[i] Ratio of molecular weights: Sulfate/SO_2= 1.50; nitrate/nitrogen dioxide = 1.35; ammonium/ammonia = 1.06.

1. All sulfate is neutralized by ammonium;
2. Ammonium nitrate forms only when there is excess ammonium;
3. Because ammonium nitrate forms only under relatively low temperatures, annual average particle nitrate concentrations are divided by four assuming that sufficiently low temperatures are present only one-quarter of the year.

Finally, we calculated the total particle mass of ammonium sulfate and ammonium nitrate.[j]

c. Fugitive Dust Adjustment Factor

The 1990 CRDM predictions for fugitive dust are not consistent with measured ambient data. The CRDM-predicted average fugitive dust contribution to total $PM_{2.5}$ mass is 31 percent in the East and 32 percent in the West (as cited in: U.S. EPA, 1998b, p. 3-15). Monitoring data from the IMPROVE network show that minerals (i.e., crustal material) comprise approximately five percent of $PM_{2.5}$ mass in the East and approximately 15 percent of $PM_{2.5}$ mass in the West (U.S. EPA, 1996b). These disparate results suggest a systematic overestimate in the fugitive dust contribution to total PM. This overestimate is further complicated by the recognition that the 1990 National Particulate Inventory (NPI) significantly overestimates fugitive dust emissions. The most recent National Emissions Trends inventory indicates that the NPI overestimates fugitive dust PM_{10} and $PM_{2.5}$ emissions by 40 percent and 73 percent respectively[k] (U.S. EPA, 1997d).

To address this bias, we applied a multiplicative factor of 0.25 nationally to fugitive dust emissions as a reasonable first-order attempt to reconcile differences between modeled predictions of $PM_{2.5}$ and actual ambient data. This is the same adjustment that was used in the NO_x SIP call analysis (U.S. EPA, 1998b). This adjustment results in a fugitive dust contribution to modeled ambient $PM_{2.5}$ concentrations of 10 percent to 17 percent.[l] Even after this adjustment the fugitive dust fraction of total eastern $PM_{2.5}$ mass is 10.4 percent, which is still greater than the five percent indicated by IMPROVE monitors. However, given that the adjustment factor appears to bring the modeled fugitive dust contribution to $PM_{2.5}$ mass more within the range of

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[j] To calculate total particle mass of ammonium sulfate and ammonium nitrate, the anion concentrations of sulfate and nitrate are multiplied by 1.375 and 1.290 respectively.

[k] Natural and man-made fugitive dust emissions account for 86 percent of PM_{10} emissions and 59 percent of $PM_{2.5}$ emissions in the most recent 1990 estimates in the National Emission Trends Inventory.

[l] See U.S. EPA (1997b, page 6-5) for a map delineating modeling regions. Using 0.25 multiplicative factor, fugitive dust as percentage of $PM_{2.5}$ mass for: Central U.S. = 17.2 percent; Eastern U.S.= 10.4 percent; Western U.S.= 10.6 percent. By comparison, without using a multiplicative factor, fugitive dust as a percentage of $PM_{2.5}$ mass for: Central U.S. = 44.6 percent; Eastern U.S. = 30.9 percent; Western U.S. = 31.5 percent.

values reported from monitoring data, we adjusted the fugitive dust contribution to total PM that is estimated by the S-R Matrix by this factor. This factor still may result in an overprediction of the fugitive dust contribution.

d. Normalizing S-R Matrix Results to Measured Data

In an attempt to further ensure comparability between S-R Matrix results and measured annual average PM values, the analysis calibrated the S-R results using factors developed for the PM and Ozone NAAQS RIA (U.S. EPA, 1997e). For the NAAQS RIA, a "calibration factor" was developed for each monitored county.[m] This analysis calibrated all S-R Matrix predictions regardless of overprediction or underprediction relative to monitored values. We applied this factor equally across all particle species contributing to the annual average PM value at a county-level receptor.

The calibration procedure employed 1993 - 1995 PM_{10} ambient monitoring data from the AIRS database following the assumptions of data completeness discussed above. The PM_{10} data represent the annual average of design value monitors averaged over three years (U.S. EPA, 1997f). We eliminated the standardization for temperature and pressure from this concentration data based upon proposed revisions to the reference method for PM_{10}.[n]

Because there is little $PM_{2.5}$ monitoring data available, we developed a general linear model to predict $PM_{2.5}$ concentrations directly from the 1993 - 1995 PM_{10} values (U.S. EPA, 1996a). The analysis used a SAS™ general linear model (i.e., GLM) procedure to predict $PM_{2.5}$ values as a function of season, region, and measured PM_{10} value. We then used these derived $PM_{2.5}$ data to calibrate model predictions of annual average $PM_{2.5}$.

[m] The normalization procedure was conducted for county-level modeled PM_{10} and $PM_{2.5}$ estimates falling into one of four air quality data tiers. The tiering scheme reflects increasing relaxation of data completeness criteria and therefore increasing uncertainty for the annual design value (U.S. EPA, 1997c). Nationwide, Tier 1 monitored counties cover the 504 counties with at least 50 percent data completeness and therefore have the highest level of certainty associated with the annual design value. Tier 2 monitored counties cover 100 additional counties with at least one data point (i.e., one 24-hour value) for each of the three years during the period 1993 -1995. Tier 3 monitored counties cover 107 additional counties with missing monitoring data for one or two of the three years 1993 - 1995. In total, Tiers 1, 2 and 3 cover 711 counties currently monitored for PM_{10} in the 48 contiguous states. In 1997 the PM_{10} monitoring network consisted of approximately 1600 individual monitors with a coverage of approximately 711 counties in the 48 contiguous states. Tier 4 covers the remaining 2369 non-monitored counties.

[n] See Appendix J - Reference Method for PM_{10} , Final Rule for National Ambient Air Quality Standards for Particulate Matter (Federal Register, Vol. 62, No. 138, p. 41, July 18, 1997).

e. Development of Annual Median $PM_{2.5}$ Concentrations

The CRDM procedure does not directly produce estimates of daily 24-hour average PM concentrations or annual median PM concentrations. Some health benefits have concentration-response (C-R) functions that rely on estimates of either the daily 24-hour average or annual median concentrations. Using historical data, EPA therefore developed 24-hour average estimates corresponding to the 99th percentile value for PM_{10} and the 98th percentile value for $PM_{2.5}$ reflecting forms of PM_{10} and $PM_{2.5}$ daily standards.

Peak-to-mean ratios (i.e., ratio of the 24-hour average value to annual average value) are established from actual PM_{10} monitor data for 1993 to 1995. For PM_{10}, the peak value is defined exactly the way it is for the new PM_{10} NAAQS, i.e., the value corresponding to the 99th percentile value of the distribution of actual daily 24-hour average PM_{10} values. For $PM_{2.5}$, the peak value is also defined exactly the way it is for the new $PM_{2.5}$ NAAQS, i.e., the value corresponding to the 98th percentile value of the distribution of estimated daily 24-hour average $PM_{2.5}$ values. In this analysis, we assumed that these historical peak-to-mean ratios hold for the 2010 model year, and applied them to the annual average PM estimates generated by the S-R Matrix.

Starting with the annual mean and peak values developed from the S-R Matrix, we used maximum likelihood to estimate the parameters of a distribution that are most consistent with the S-R Matrix results. Using the parameters of the distribution, we then estimated the annual median concentration and other representative concentrations in the distribution (e.g., 5^{th} percentile).

f. PM Air Quality Results

Table VII-3 provides a summary of the predicted ambient PM_{10} and $PM_{2.5}$ concentrations used in this study. The concentration changes are generally very small. The technical support document for this RIA (Abt Associates, 1999) contains maps showing the base case PM concentrations and PM concentration changes generated.

Table VII-3. Summary of S-R Matrix Derived PM Air Quality

Statistic	*2010 Base Case*	*Change*[a]	*Percent Change*
PM_{10}			
Minimum Annual Mean PM_{10} ($\mu g/m^3$) [b]	5.96	-0.64	-10.7%
Maximum Annual Mean PM_{10} ($\mu g/m^3$) [b]	63.18	0.00	0.0%
Average Annual Mean PM_{10} ($\mu g/m^3$)	22.46	-0.14	-0.6%
Population-Weighted Average Annual Mean PM_{10} (person-$\mu g/m^3$) [c]	28.31	-0.20	-0.7%
$PM_{2.5}$			
Minimum Annual Mean $PM_{2.5}$ ($\mu g/m^3$) [b]	0.86	-0.64	-74.4%
Maximum Annual Mean $PM_{2.5}$ ($\mu g/m^3$) [b]	28.02	0.00	0.0%
Average Annual Mean $PM_{2.5}$ ($\mu g/m^3$)	10.75	-0.14	-1.3%
Population-Weighted Average Annual Mean $PM_{2.5}$ (person-$\mu g/m^3$) [c]	13.00	-0.20	-1.5%

[a] The change is defined as the control case value minus the base case value.

[b] The base case minimum (maximum) is the value for the county with the lowest (highest) annual average. The change relative to the base case picks the minimum (maximum) from the set of changes in all counties.

[c] Calculated by summing the product of the projected 2010 county population and the estimated 2010 county PM concentration, and then dividing by the total population in the 48 contiguous states.

3. Nitrogen Deposition Estimates

The analysis used RADM to generate nitrogen deposition estimates. The RADM was developed over a ten year period, 1984 - 1993, under the auspices of the National Acid Precipitation Assessment Program (NAPAP), to address policy and technical issues associated with acidic deposition. The model provides a scientific basis for predicting changes in deposition and air quality resulting from changes in precursor emissions and to predict the levels of acidic deposition in certain sensitive receptor regions. To do so requires that RADM be a multipollutant model that predicts the oxidizing capacity of the atmosphere, including the prediction of ozone, and chemical transformations involving oxides of sulfur and nitrogen.

NAPAP has extensively documented the development, application, and evaluation of the RADM (Chang et al., 1987; Chang et al., 1990; Dennis et al., 1990). Several recent studies of acidic deposition have used RADM, including EPA's 1995 *Acid Deposition Standard Feasibility Study Report to Congress* (U.S. EPA, 1995), EPA's 1997 *Deposition of Air Pollutants to the Great Waters* Report to Congress (U.S. EPA, 1997a), work estimating the nitrogen deposition airshed of the Chesapeake Bay watershed (Dennis, 1997), and in the NO_x SIP call (U.S. EPA, 1998a)

RADM estimates deposition in units of kilograms per hectare (kg/ha). The model estimates wet deposition in the form of SO_4^{2-}, NO_3^-, NH_3, H^+. It estimates dry deposition in the form of SO_2, SO_4 as aerosol, O_3, HNO_3, NO_2, H_2O_2. The model then maps the deposition estimates to specific East Coast and Gulf Coast estuaries and their watersheds, which are subject to eutrophication problems. Land-deposited nitrogen in each watershed is multiplied by a factor of 10 percent to obtain the nitrogen load delivered via export (pass-through) to the corresponding estuary.

Table VII-4 provides a summary of the change in nitrogen deposition estimates for selected estuaries as a result of the Tier 2 rule[1]. The results represent a 10.8 percent reduction in the average annual deposition across these estuaries.

Table VII-4. Summary of 2010 Nitrogen Deposition in Selected Estuaries (million kg/year)

Estuary	*2010 Base Case*	*Change[a]*	*Percent Change*
Albemarle/Pamlico Sound	11.87	-1.27	-10.7%
Cape Cod Bay	3.96	-0.42	-10.6%
Chesapeake Bay	18.05	-1.91	-10.6%
Delaware Bay	3.37	-0.34	-10.1%
Delaware Inland Bays	0.44	-0.04	-9.1%
Gardiners Bay	1.24	-0.13	-10.5%
Hudson River/Raritan Bay	3.95	-0.45	-11.4%
Long Island Sound	5.78	-0.66	-11.4%
Massachusetts Bay	1.33	-0.14	-10.5%
Narragansett Bay	1.17	-0.12	-10.3%
Sarasota Bay	0.37	-0.04	-10.8%
Tampa Bay	2.27	-0.28	-12.3%
All Selected Estuaries	53.8	-5.8	-10.8%

[a] Change is defined here as the emissions level after implementing the Tier 2 rule minus the base case emissions.

4. Visibility Degradation Estimates Using the S-R Matrix

Visibility degradation is often directly proportional to decreases in light transmittance in the atmosphere. Scattering and absorption by both gases and particles decrease light transmittance. To quantify changes in visibility, our analysis used a light-extinction coefficient, based on the work of Sisler (1996), which shows the total fraction of light that is decreased per unit distance.

The light extinction coefficient accounts for the scattering and absorption of light by both particles and gases, and a number of factors are included in its estimation. Because fine particles are much more efficient at light scattering than coarse particles, the analysis specifies several fine particle species, whereas coarse particles are kept as one category. Fine particles with significant light-extinction efficiencies include sulfates, nitrates, organic carbon, elemental carbon (soot), and soil (Sisler, 1996).

Once we determined the light-extinction coefficient, we calculated a unitless visibility index, called a "deciview," which we used in the valuation of visibility. The deciview metric provides a linear scale for perceived visual changes over the entire range of conditions, from clear to hazy. Under many scenic conditions, the average person can generally perceive a change of one deciview.

The analysis generated visibility degradation estimates in "recreational" (e.g., federally designated Class I areas such as national parks and recreation areas) and "residential" (non-Class I areas) areas at the county level using the results of the S-R Matrix. The visibility benefits analysis (see Section VII.C) distinguishes between general regional visibility degradation and visibility degradation in certain Federally-designated Class I areas (i.e., national parks, forests, recreation areas, wilderness areas, etc.). Therefore we separated visibility degradation estimates into "residential" and "recreational" categories depending upon the geographic area covered by the estimate, and summed from the county-level to one of six regions (defined in part by the underlying study) and the nation.

Table VII-5 provides a summary of the visibility degradation estimates in terms of deciviews. The valuation methodology for recreational visibility requires separate treatment of visibility changes in the different regions in the U.S. Table VII-5 provides residential and recreational visibility degradation estimates for each region. All predicted visibility changes are small (less than one deciview), with the largest changes occurring in the Southeast and Northeast. The air quality technical support document for this RIA (Abt Associates, 1999) contains maps showing the base case visibility degradation and visibility degradation changes.

Table VII-5. Summary of 2010 Visibility Degradation Estimates (deciviews)

Visibility Degradation	*2010 Base Case*	*Change*[a]	*Percent Change*
Southeast			
Annual Average--Residential	23.44	-0.19	-0.8%
Annual Average--Recreational[b]	21.65	-0.23	-1.1%
Southwest			
Annual Average--Residential	17.89	-0.08	-0.4%
Annual Average--Recreational[b]	18.69	-0.08	-0.4%
California & Nevada			
Annual Average--Residential	19.29	-0.04	-0.2%
Annual Average--Recreational[b]	19.93	-0.06	-0.3%
Northeast			
Annual Average--Residential	21.80	-0.17	-0.8%
Annual Average--Recreational[b]	17.66	-0.06	-0.3%
North Central			
Annual Average--Residential	18.55	-0.11	-0.6%
Annual Average--Recreational[b]	19.13	-0.08	-0.4%
Northwest			
Annual Average--Residential	20.70	-0.21	-1.0%
Annual Average--Recreational[b]	21.65	-0.15	-0.7%
National			
Annual Average--Residential	21.77	-0.16	-0.7%
Annual Average--Recreational[b]	19.51	-0.09	-0.5%

[a] The change is defined as the control case deciview level minus the base case deciview level.
[b] Recreational visibility averages are from the 41 Class I areas used in the benefits analysis. See Table VII-14 for list of Class I areas.

C. Benefits Assessment

The changes in ozone, PM, nitrogen oxides, and visibility levels described in Section VII.B will result in changes in the health and welfare impacts associated with elevated ambient concentrations of these pollutants. This Section describes the methods for estimating the physical magnitude and monetary value of these impacts.

Section VII.C.1 provides an overview of the benefits methodology. Section VII.C.2 discusses issues in estimating health effects. Section VII.C.3 discusses methods and provides estimated values for avoided incidences and monetary benefits for ozone- and PM-related health effects. Section VII.C.4 discusses methods and provides estimated values for air pollution-related welfare effects. Section VII.C.5 discusses the aggregation of health and welfare benefits, and presents an estimate of total benefits. Section VII.C.6 presents sensitivity analyses, and Section VII.C.7 discusses potential benefit categories that are not quantified due to data and/or methodological limitations, and provides a list of analytical uncertainties, limitations, and biases.

1. Overview of Benefits Estimation

Most of the specific methods and information used in this benefit analysis are similar to those used in the §812 Retrospective of the Benefits and Costs of the Clean Air Act and forthcoming §812 Prospective EPA Reports to Congress, which were reviewed by EPA's Science Advisory Board (U.S. EPA, 1997g), as well as the approach used by EPA in support of revising the ozone and PM NAAQS (U.S. EPA, 1997e; U.S. EPA, 1997h) and the Regional NOx SIP call (U.S. EPA, 1998b). Prior to describing the details of the approach for the benefits analysis, it is useful to provide an overview of the approach. The overview is intended to help the reader better identify the role of each issue described later in this Section.

The general term "benefits" refers to any and all outcomes of the regulation that contribute to an enhanced level of social welfare. The value of "benefits" refers to the dollar value associated with all the expected positive impacts of the regulation; that is, all regulatory outcomes that lead to higher social welfare. If the benefits are associated with market goods and services, the monetary value of the benefits is approximated by the sum of the predicted changes in "consumer (and producer) surplus." If the benefits are non-market benefits (such as the risk reductions associated with environmental quality improvements), however, other methods of measuring benefits must be used as discussed in the text. The total value of such a good is the sum of the dollar amounts that all those who benefit are willing to pay.

In addition to benefits, regulatory actions may also lead to unintended nonmarket costs, that some might term "disbenefits." An example of a disbenefit of reduced ozone concentrations is that there will be less protection from UV radiation. In order to quantify the impact of a regulatory action, both the benefits and disbenefits should be included. However, like many benefits, disbenefits are difficult to quantify. EPA's approach is to present as complete a set of quantified estimates of benefits and disbenefits as possible, given the state of science at the time of the analysis.

This conceptual economic foundation raises several relevant issues and potential limitations for the benefits analysis of the regulation. First, the standard economic approach to estimating environmental benefits is anthropocentric– all benefits values arise from how environmental changes are perceived and valued by people. Thus, all near-term as well as temporally distant future physical outcomes associated with reduced pollutant loadings need to be predicted and then translated into the framework of present-day human activities and concerns. Second, as noted below, it is not possible to quantify or to value all of the benefits resulting from environmental quality improvements.

Conducting a benefits analysis for anticipated changes in air emissions is a challenging exercise, as it requires a series to steps to be specified and understood. Figure VII-3 illustrates these steps, which include: (1) institutional relationships and policy-making; (2) the technical feasibility of pollution abatement; (3) the physical-chemical properties of air pollutants and their consequent linkages to biological or ecological responses in the environment, and (4) human responses and values associated with these changes.

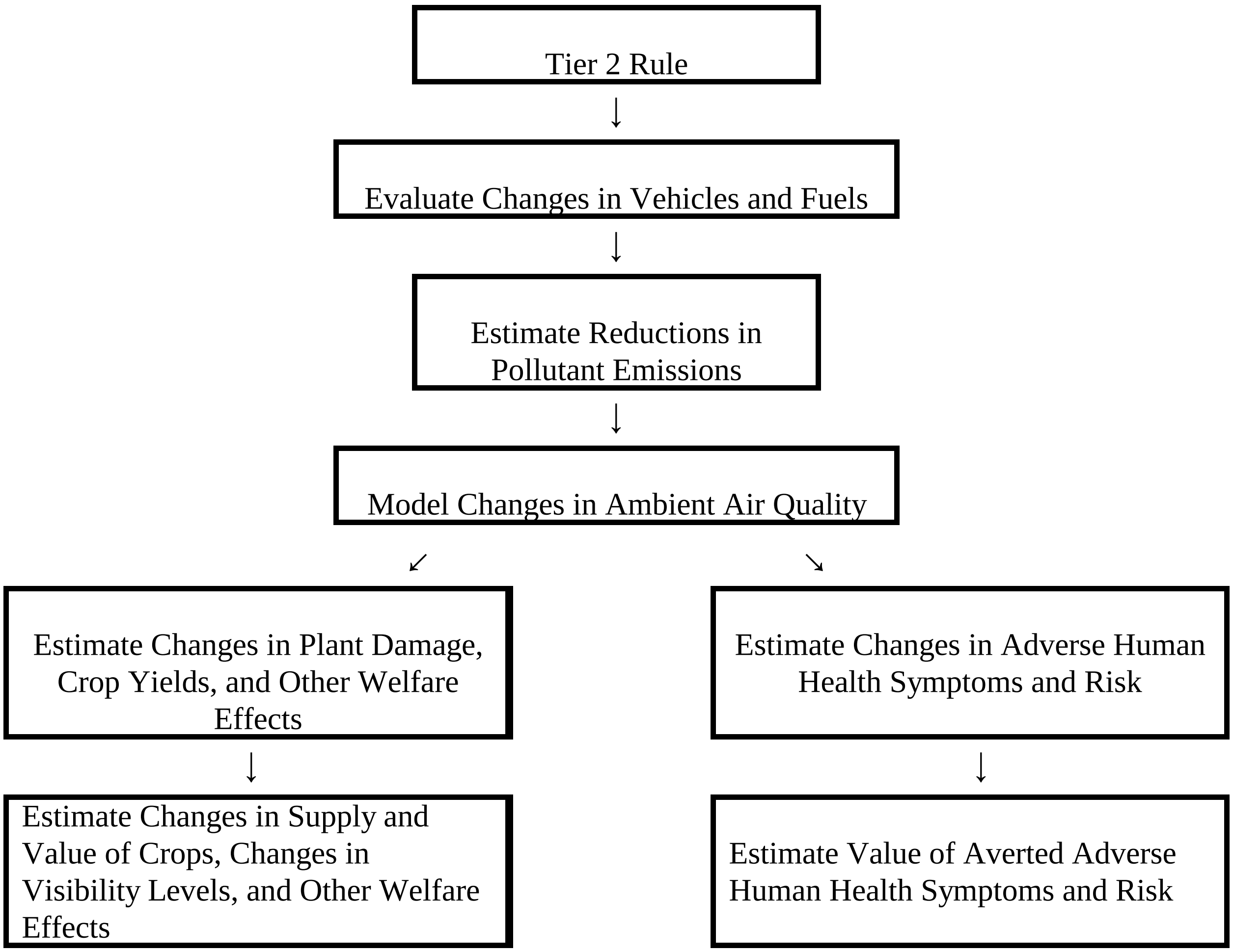

Figure VII-3. Example Benefits Analysis Method

Our analysis mainly uses a "damage function" approach to estimate the adverse physical effects from air pollution that will be avoided in the United States due to implementation of the emission reductions required by the Tier 2 rule.[o] This approach examines individual physical effects, such as, say, hospital admissions, that may be affected by reductions in specific pollutants. The total value for a given physical effect is simply the product of the number of incidences avoided and the value per incidence avoided. The damage function approach assumes that the benefits from individual effects are additive and independent, i.e., benefits for one effect do not depend on benefits for a separate effect. Alternative approaches include market-based measures include: hedonic prices, which measure the total value of a reduction in air pollution

[o]The exception to this is the estimation of nitrogen deposition benefits, which uses an avoided cost approach.

using a single metric, such as the price of a house, or contingent valuation, which asks individuals for their total willingness to pay (WTP) for a reduction in air pollution. If the single metric approach successfully captures the full WTP for a reduction in air pollution, then the damage function approach should yield an estimate that is less than or equal to the estimate from the single metric approach. All monetized estimates of benefits presented are in 1997 dollars.[p]

Some of the estimates of the economic value of avoided health and welfare effects are derived from contingent valuation (CV) studies. Concerns about the reliability of value estimates that come from CV studies have dominated debates about the methodology, since research has shown that bias can be introduced easily into these studies, especially if they are not carefully done. Accurately measuring willingness to pay for avoided health and welfare losses depends on the reliability and validity of the data collected. There are several issues to consider when evaluating study quality, including but not limited to 1) whether the sample estimates of WTP are representative of the population WTP, 2) whether the good to be valued is comprehended and accepted by the respondent, 3) whether the WTP elicitation format is designed to minimize strategic responses, 4) whether WTP is sensitive to respondent familiarity with the good, to the size of the change in the good, and to income, 5) whether the estimates of WTP are broadly consistent with other estimates of WTP for similar goods, and (6) the extent to which responses are consistent with established economic principles. This benefits analysis does not attempt to list the individual strengths and weaknesses of each CV study used. However, in some instances, such as for valuation of chronic bronchitis and residential visibility, when the CV study reliability is questionable, we adopt alternative estimates as conservative measures of benefits, which are presented in the low-end estimate of the range of monetized benefits. In other instances, for example the study used to value changes in visibility at Class I areas, we recognize potential weaknesses, but do not alter the estimates presented in the study.

In this analysis, the valuation of avoided incidences of health effects and avoided degradation of welfare effects relies on benefits transfer. The benefits transfer approach takes values or value functions generated by previous research and transfers them from the study to the policy of interest. For example, we obtained the value of reduced mortality from a distribution of values of statistical life based on 26 wage-risk and contingent valuation studies. None of the values for the health and welfare categories valued in this benefit analysis were generated specifically in the context of the Tier 2 rule. The validity of this approach relies on the correlation between attributes of the policy and the studies from which the values were obtained. Where possible, we selected studies that valued effects matching those in the policy analysis. When studies were not available that exactly matched the studied effect and the policy effect, we

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[p]Recent analyses have been presented in 1990 dollars, such as the NO_x SIP call (U.S. EPA, 1998b) and the §812 Retrospective of the Benefits and Costs of the Clean Air Act (U.S. EPA, 1997g). The method of adjusting from 1990 dollars to 1997 dollars depends on the basis of the benefits estimates. Benefits estimates based on cost-of-illness are adjusted by using the consumer price indexes (CPI-Us) for medical care, while benefits estimates based directly on estimates of WTP have been adjusted using the CPI-U for "all items."

selected studies that matched as closely as possible, and note the differences (and where known, potential drawbacks to their application) in the text.

The first step in a benefits analysis using this approach is the identification of the types or categories of benefits associated with the anticipated changes in ambient air quality conditions. The second step is the identification of relevant studies examining the relationships between air quality and these benefit categories and studies estimating the value of avoiding damages. The most prominent avoided damages are those related to human health risk reductions, effects on crops and plant life, visibility, and materials damage.

It is difficult to identify all the types of benefits that might result from environmental regulation and to value those benefits that are identified, due to the non-market nature of many benefits categories. Since many pollution effects (e.g., adverse health or ecological effects) traditionally have not been traded as market commodities, economists and analysts cannot look to changes in market prices and quantities to estimate the value of these effects. This lack of observable markets may lead to the omission of significant benefits categories from an environmental benefits analysis. It is not possible to quantify the magnitude of this underestimation. The more important of these omitted effect categories are shown in Table VII-6.

Table VII-6. Unquantified Benefit Categories*

	Unquantified Benefit Categories Associated with Ozone and Nitrogen Oxides	*Unquantified Benefit Categories Associated with PM*
Health Categories	Airway responsiveness. Pulmonary inflammation. Increased susceptibility to respiratory infection. Acute inflammation and respiratory cell damage. Chronic respiratory damage/premature aging of lungs. Ultraviolet-B radiation (cost).	Changes in pulmonary function. Morphological changes. Altered host defense mechanisms. Cancer. Other chronic respiratory disease.
Welfare Categories	Ecosystem and vegetation effects in Class I areas (e.g., national parks). Damage to urban ornamentals (e.g.,grass, flowers, shrubs, and trees in urban areas). Fruit and vegetable crops. Reduced yields of tree seedlings, commercial and non-commercial forests. Damage to ecosystems. Materials damage (other than consumer cleaning cost savings). Nitrates in drinking water. Brown clouds.	Materials damage (other than consumer cleaning cost savings). Damage to ecosystems (e.g., acid sulfate deposition). Nitrates in drinking water. Brown clouds.

* Note that there are other pollutants that are reduced in conjunction with the Tier 2 rule that are not considered in this analysis, such as carbon (a pollutant associated with global climate change).

Within each effect category, there may be several possible estimates of health and welfare effects. Each of these possibilities represents a health or welfare “endpoint.” The basic structure of the analysis is to create a set of benefit estimates reflecting key assumptions concerning environmental conditions and the responsiveness of human health and the environment to changes in air quality. Total benefits are presented as the sum of non-overlapping endpoints, to avoid double-counting benefits.

We made subjective judgements in our analysis because of a lack of information. To reflect the range of uncertainty regarding key assumptions– such as the appropriate PM threshold– this analysis uses two suites of assumptions. This RIA has adopted the approach of presenting a range of monetized benefits that reflects these uncertainties by selecting alternative values for each of several key assumptions. Taken together, these alternative sets of assumptions define a “high end” and a “low end” estimate for the benefits that have been monetized in this analysis.

Table VII-7 lists the specific health and welfare effects that are included in at least one of the assumptions sets, indicating the specific effect categories that are included in the plausible range of benefits. This table also includes the estimates of mean WTP, or “unit values” used to monetize the benefits for each effect. Table VII-8 highlights the key differences between the assumption sets.

Table VII-7. Quantified and Monetized Primary Health and Welfare Effects

Effect	*Pollutant*	*Value per incident ($1997)* *LOW*	*HIGH*
Health Effects in the Benefits Analysis			
Mortality, long-term exposure - over age 30	$PM_{2.5}$	$2,730,000	$5,894,400
Mortality, short-term exposure	Ozone	$0	$5,894,400
Chronic bronchitis - all ages	PM_{10}	$74,500	$319,280
Hospital admissions - all respiratory, all ages	Ozone & $PM_{2.5}$	$9,672 (Ozone) $9,142 (PM)	$9,672 (Ozone) $9,142 (PM)
Hospital admissions - congestive heart failure	PM_{10}	$11,931	$11,931
Hospital admissions - ischemic heart disease	PM_{10}	$14,854	$14,854
Any of 19 acute respiratory symptoms -adult	Ozone	$22	$22
Acute bronchitis - children	$PM_{2.5}$	$55	$55
Lower respiratory symptoms (LRS) - children	PM_{10}	$15	$15
Upper respiratory symptoms (URS) - children	PM_{10}	$23	$23
Work loss days (WLD) - adult	$PM_{2.5}$	$102	$102
Minor restricted activity days (MRAD) - adult	$PM_{2.5}$	$47	$47
Welfare Effects in the Benefits Analysis			
Agriculture - select commodity crops	Ozone	n/a	n/a
Household soiling (annual value)	PM_{10}	$3.09/household/ $\mu g/m^3$ change in PM_{10}	$3.09/household/$\mu g/m^3$ change in PM_{10}
Nitrogen deposition: (annual value)	NO_x		
Albemarle-Pamilico Sound		$90/kg of nitrogen	$90/kg of nitrogen
Chesapeake Bay		$59/kg of nitrogen	$59/kg of nitrogen
Tampa Bay		$238/kg of nitrogen	$238/kg of nitrogen
Average nine estuaries		$129/kg of nitrogen	$129/kg of nitrogen
Decreased worker productivity	Ozone	$1/worker/10% change in ozone	$1/worker/10% change in ozone
Visibility - residential	PM and gases	not valued	$17/household per deciview
In-region recreational visibility: (annual value)	PM and gases		
California		$6.43/household /deciview	$12.89/household /deciview
Southwest		$8.41/household	$16.82/household

Effect	*Pollutant*	*Value per incident ($1997)* *LOW*	*HIGH*
Southeast		$3.99/household /deciview	$7.98/household /deciview
Out-of-region recreational visibility: California		$4.48/household /deciview	$8.96/household /deciview
Southwest		$6.76/household /deciview	$13.51/household /deciview
Southeast		$2.46/household /deciview	$4.91/household /deciview

Table VII-8. Key Differences Between Low and High Assumption Sets

Assumption	*Low*	*High*
Threshold for PM effect	15 $\mu g/m^3$	background
PM Mortality	value of statistical life year lost	value of statistical life
Ozone-related short-term exposure mortality	excluded	included
Agriculture	low crop sensitivity to ozone	high crop sensitivity to ozone
Visibility	no residential visibility valuation	recreational and residential visibility valued
Infant mortality	excluded	included

### 2.	Issues in Estimating Changes in Health Effects

This benefits analysis relies on concentration-response (C-R) functions estimated in published epidemiological studies relating adverse health to ambient air quality. The specific C-R functions used are included in Table VII-9. While a broad range of adverse health effects have been associated with exposure to elevated ozone and PM levels (as noted for example in Table VII-6), in this quantified benefit analysis only a subset of health effects are included. Health effects are excluded from the current analysis for three reasons: (1) the possibility of double counting (such as hospital admissions for specific respiratory diseases); (2) uncertainties in applying effect relationships based on clinical studies (where human subjects are exposed to various levels of air pollution in a carefully controlled and monitored laboratory situation) to the affected population; or (3) a lack of an established C-R relationship.

When a single published study is selected as the basis of the C-R relationship between a pollutant and a given health effect, or "endpoint," applying the C-R function is straightforward. This is the case for most of the endpoints selected for inclusion in the benefits analysis. A single C-R function may be chosen over other potential functions because the underlying epidemiological study used superior methods, data or techniques, or because the C-R function is more generalized and comprehensive. For example, the study that estimated the effects of PM on hospital admissions for all ages and all respiratory diseases is selected over studies limited to the over age 65 population or specific categories of respiratory diseases.

An exception to the "single study" selection in the benefits analysis is mortality associated with exposure to ozone. Estimates of premature mortality associated with short-term exposure to $PM_{2.5}$ and PM_{10}, are also based on multiple estimates of the relationship between PM and mortality, but are presented as a sensitivity analysis. When several estimated C-R relationships between a pollutant and a given health endpoint have been selected, they are combined or pooled to derive a single estimate of the relationship. A separate technical support document provides details of the procedures used to combine multiple C-R functions (Abt Associates, 1999).

Whether the C-R relationship between a pollutant and a given health endpoint is estimated by a single function from a single study or by a pooled function of C-R functions from several studies, we apply that same C-R relationship everywhere in the benefits analysis. Although the C-R relationship may in fact vary somewhat from one location to another (for example, due to differences in population susceptibilities or differences in the composition of PM), location-specific C-R functions are generally not available. While a single function applied everywhere may result in overestimates of incidence changes in some locations and underestimates of incidence changes in other locations, these location-specific biases will to some extent cancel each other out when the total incidence change is calculated. It is not possible to know the extent or direction of the bias in the total incidence change based on application of a single C-R function everywhere.

The remainder of this Section discusses two key issues involving the use of C-R functions to estimate the benefits of the Tier 2 rule: baseline incidences and health effect thresholds, i.e. levels of pollution below which changes in air quality have no impacts on health.

Table VII-9. PM and Ozone Health Concentration-Response Function Summary Data

Endpoint	*Pollutant*	*Concentration-Response Function*		*Averaging Time*		*Population*[b]	*Pollutant Coefficient*[c]
		Source	*Functional Form*[a]	*Studied*	*Applied*		
Mortality							
Mortality (long-term exposure) - $PM_{2.5}$	$PM_{2.5}$	Pope et al. (1995)	log-linear	annual median	annual median[c]	ages 30+	0.006408
Mortality (short-term exposure)	Ozone	Kinney et al., (1995)	log-linear	daily 1-hour max	daily 1-hour max	all	0.000000
	Ozone	Ito and Thurston (1996)	log-linear	1-day average	1-day average	all	0.000677
	Ozone	Moolgavkar et al. (1995)	log-linear	1-day average	1-day average	all	0.000611
	Ozone	Samet et al. (1997)	log-linear	1-day average	1-day average	all	0.000936
Hospital Admissions							
All respiratory illnesses	$PM_{2.5}$/ PM_{10}	Thurston et al. (1994)	linear	1-day average	1-day average	all	3.45×10^{-8}
Congestive heart failure	PM_{10}	Schwartz & Morris (1995)	log-linear	2-day average	1-day average	age 65+	0.00098
Ischemic heart disease	PM_{10}	Schwartz & Morris (1995)	log-linear	1-day average	1-day average	age 65+	0.00056
All respiratory illnesses	Ozone	Thurston et al. (1992)	linear	daily 1-hour max	daily 1-hour max	all	0.00137

Endpoint	*Pollutant*	*Concentration-Response Function*		*Averaging Time*		*Population*[b]	*Pollutant Coefficient*[c]
		Source	*Functional Form*[a]	*Studied*	*Applied*		
Respiratory Symptoms/Illnesses not requiring hospitalization							
Development of chronic bronchitis	PM_{10}	Schwartz (1993)	logistic	annual mean	annual mean	all	0.012
Acute bronchitis	$PM_{2.5}$	Dockery et al.(1989)	logistic	annual mean	annual mean[d]	ages 10-12	0.0298
Upper respiratory symptoms (URS)	PM_{10}	Pope et al. (1991)	log-linear	1-day average	1-day average	asthmatics, ages 9-11	0.0036
Lower respiratory symptoms (LRS)	PM_{10}	Schwartz et al. (1994)	logistic	1-day average	1-day average	ages 8-12	0.01823
Any of 19 acute respiratory symptoms	Ozone	Krupnick et al. (1990)	logistic	daily 1-hour max	daily 1-hour max	ages 18-65	0.00014
Minor restricted activity days (MRAD)	$PM_{2.5}$	Ostro and Rothschild (1989)	log-linear	2-week average	1-day average	ages 18-65	0.00741
Work loss days (WLD)	$PM_{2.5}$	Ostro (1987)	log-linear	2-week average	1-day average	ages 18-65	0.0046
Decreased worker productivity	Ozone	Crocker & Horst (1981) and EPA (1994)	percent change	1-day average	1-day average	laborers	n/a

[a] The log-linear is the most common concentration-response relationship; in this case, the relationship between a change in pollutant level, ΔPM, and the change in incidence of the health effect, Δy, is: Δy = population* incidence rate*[exp(B*ΔPM)-1].

[b] The population examined in the study and to which this analysis applies the reported concentration-response (C-R) relationship. In general, epidemiological studies analyzed the C-R relationship for a specific age group (e.g., ages 65+) in a specific geographical area. This analysis applies the reported pollutant coefficient to all individuals in the age group nationwide.

[c] A single pollutant coefficient reported for several studies indicates a pooled analysis; see text for discussion of pooling C-R relationships across studies.

a. Baseline Incidences

The epidemiological studies of the association between pollution levels and adverse health effects generally provide a direct estimate of the relationship of air quality changes to the relative risk of a health effect, rather than an estimate of the absolute number of avoided cases. For example, a typical result might be that a ten $\mu g/m^3$ decrease in daily $PM_{2.5}$ levels might decrease hospital admissions by three percent. The baseline incidence of the health effect is necessary to convert this relative change into a number of cases.

Because most PM and ozone studies that estimate C-R functions for mortality considered only non-accidental mortality, we adjusted county-specific baseline mortality rates used in the estimation of PM- and ozone-related mortality to provide a better estimate of county-specific non-accidental mortality. We multiplied each county-specific mortality rate by the ratio of national non-accidental mortality to national total mortality (0.93). We estimated county-specific baseline mortality incidences among individuals aged 30 and over– necessary for $PM_{2.5}$-related long-term exposure mortality, estimated by Pope et al. (1995)– by applying national age-specific death rates to county-specific age distributions, and adjusting the resulting estimated age-specific incidences so that the estimated total incidences (including all ages) equals the actual county-specific total incidences.

County-level incidence rates are not available for other endpoints. The analysis used national incidence whenever possible, because these data are most applicable to a national assessment of benefits. However, for some studies, the only available incidence information come from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence at the national level.

b. Thresholds

A very important issue in applied modeling of changes in PM is whether to apply the C-R functions to all predicted changes in ambient concentrations, even small changes occurring at levels approaching "anthropogenic background". Different assumptions about whether to model thresholds, and if so, at what level, can have a major effect on the resulting benefits estimates. We use two thresholds– a different threshold for the low, primary, and high sets of assumptions– which are set respectively at: 1) 15 $\mu g/m^3$ for all effects except those that have a lowest observed level higher than 15 $\mu g/m^3$; and 2) the background level of the pollutant (i.e., the pollutant level that would occur after removing all anthropogenic emissions).

3. PM- and Ozone-related Health Effects

This Section discusses the methods used to estimate the change in the incidence of PM- and ozone-related health effects due to the Tier 2 rule and the methods used to value this change.

a. Premature Mortality

Both ozone and particulate matter have been associated with increased risk of premature mortality, which is a very important health endpoint in this economic analysis due to the high monetary value associated with risks to life. There are two types of exposure to elevated levels of air pollution that may result in premature mortality. Acute (short-term) exposure (e.g., exposure on a given day) to peak pollutant concentrations may result in excess mortality on the same day or within a few days of the elevated exposure. Chronic (long-term) exposure (e.g., exposure over a period of a year or more) to levels of pollution that are generally higher may result in mortality in excess of what it would be if pollution levels were generally lower. The excess mortality that occurs will not necessarily be associated with any particular episode of elevated air pollution levels. Both types of effects are biologically plausible, and there is an increasing body of consistent corroborating evidence from animal toxicity studies indicating that both types of effects exist.

There are, similarly, two basic types of epidemiological studies of the relationship between mortality and exposure to pollutants. Long-term studies (e.g., Pope et al., 1995) estimate the association between long-term (chronic) exposure to air pollution and the survival of members of a large study population over an extended period of time. Such studies examine the health endpoint of concern in relation to the general long-term level of the pollutant of concern–for example, relating annual mortality to some measure of annual pollutant level. Daily peak concentrations would impact the results only insofar as they affect the measure of long-term (e.g., annual) pollutant concentration. In contrast, short-term studies relate daily levels of the pollutant to daily mortality. By their basic design, daily studies can detect acute effects but cannot detect the effects of long-term exposures. A chronic exposure study design (a prospective cohort study, such as the Pope study) is best able to identify the long-term exposure effects, and may detect some of the short-term exposure effects as well. Because a long-term exposure study may detect some of the same short-term exposure effects detected by short-term studies, including both types of study in a benefit analysis would likely result in some degree of double counting of benefits.

Another major advantage of the long-term study design concerns the issue of the degree of prematurity of mortality associated with air pollution. It is possible that the short-term studies are detecting an association between air pollution and mortality that is primarily occurring among terminally ill people. Critics of the use of short-term studies for policy analysis purposes correctly point out that an added risk factor that results in a terminally ill person dying a few days or weeks earlier than they otherwise would have (known as “short-term harvesting”) is potentially included in the measured air pollutant mortality “signal” detected in such a study. As the short-term study design does not examine individual people (it examines daily mortality rates in large populations, typically a large city population), it is impossible to know anything about the overall health status of the specific population that is detected as dying early. While some of the detected excess deaths may have resulted in a substantial loss of life (measuring loss of life in terms of lost years of remaining life), others may have lost a relatively short amount of lifespan.

While the long-term study design is preferred, these types of studies are expensive to conduct and consequently there are relatively few well designed long-term studies. For PM, there has only been one high quality study accepted by the Science Advisory Board, and for ozone, no acceptable long-term studies have been published. For this reason, our analysis used short-term ozone mortality studies as the basis for determining ozone-related mortality benefits. The next two Sections provide details on the measurement of changes in incidences of premature mortality associated with changes in PM and ozone arising from implementation of the Tier 2 rule.

Estimating PM-related Premature Mortality

The benefits analysis estimated PM-related mortality using the $PM_{2.5}$ relationship from Pope et al. (1995). This decision reflects the Science Advisory Board's explicit recommendation for modeling the mortality effects of PM in both the completed §812 Retrospective Report to Congress and the ongoing §812 Prospective Study. The Pope et al. study estimates the association between long-term (chronic) exposure to $PM_{2.5}$ and the survival of members of a large study population. This relationship is selected for use in the benefits analysis instead of short-term (daily pollution) studies for a number of reasons.

We selected the Pope et al. (1995) long-term study as providing the best available estimate of the relationship between PM and mortality. It is used alone– rather than considering the total effect to be the sum of estimated short-term and long-term effects– because summing creates the possibility of double-counting a portion of PM-related mortality. We selected the Pope et al. study in preference to other available long-term studies because it uses better statistical methods, has a much larger sample size, the longest exposure interval, and more locations (51 cities) in the United States, than other studies. It is unlikely that the Pope et al. study contains any significant amount of short-term harvesting. First, the health status of each individual tracked in the study is known at the beginning of the study period. Persons with known pre-existing serious illnesses were excluded from the study population. Second, the statistical model used in the Pope study examines the question of survivability throughout the study period (ten years). Deaths that are premature by only a few days or weeks within the ten-year study period (for example, the deaths of terminally ill patients, triggered by a short duration PM episode) are likely to have little impact on the calculation of the average probability of surviving the entire ten year interval. In relation to the "Six-cities" study by Dockery et al. (1993), the Pope et al. study found a smaller increase in excess mortality for a given PM air quality change.

Estimating Ozone-related Premature Mortality

The literature on the possible relationship between exposure to ambient ozone and premature mortality has been evolving rapidly. Of the 28 time-series epidemiology studies identified in the literature that report results on a possible association between daily ozone concentrations and daily mortality (see (see: U.S. EPA, 1997e, Appendix J), 21 were published or presented since 1995. In particular, a series of studies published in 1995 through 1997 (after closure on the ozone Criteria Document) from multiple cities in western Europe has significantly increased the body of studies finding a positive association. Fifteen of the 28 studies report a statistically significant relationship between ozone and mortality, with the more recent studies tending to find statistical significance more often than the earlier studies. The ozone-mortality datasets have also tended to become larger in more recent studies as longer series of air quality monitoring data have become available over time. This suggests that it may take many years of data before the ozone effect can be separated from the daily weather and seasonal patterns with which it tends to be correlated.

In 1997, as a part of the ozone NAAQS promulgation RIA, EPA staff reviewed this recent literature. They identified nine studies that met a defined set of selection criteria, and conducted a meta-analysis of the results of the nine studies (U.S. EPA, 1997e). Our analysis implements the same basic approach to quantifying ozone mortality as the NAAQS, with the exception that a subset of four of the nine studies is used, representing only U.S. based analyses.[q] In a post-NAAQS RIA review of the methodology for assessing ozone mortality effects, it was determined that the relationships between ambient ozone and mortality in the non-U.S. study locations included in the original NAAQS-related analysis may not be representative of the range of ozone-mortality C-R relationships in the United States. To reduce the potential for applying inappropriate C-R functions of the ozone mortality benefits from the Tier 2 rule, the analysis only included U.S. studies, based on the assumption that demographic and environmental conditions on average would be more similar between the study and policy sites. However, the full body of peer-reviewed ozone mortality studies should be considered when evaluating the weight of evidence regarding the presence of an association between ambient ozone concentrations and premature mortality.

Because of differences in the averaging times used in the underlying studies (some use daily average ozone levels, while others use 1-hour daily maximum values), it is not possible to conduct a meaningful analysis directly on the coefficients of the C-R functions. Instead, the analysis translated each C-R function into a set of predicted mortality incidence changes that would be estimated by that C-R function, given the set of air quality changes. We then combined these studies to estimate the impact of ozone on mortality incidence. The technical support document for this analysis provides additional details of this approach (Abt Associates, 1999)

Infant Mortality

Woodruff et al. (1997) found a significant association between annual PM_{10} levels and post-neonatal mortality (deaths of infants aged 28 - 51 weeks). This estimate should not overlap with the Pope et al. (1995) estimate because the Pope et al. function is based on a population over

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[q]The U.S. study-only approach has been implemented previously in the NO_x SIP call RIA (U.S. EPA, 1998b).

the age of 30. The SAB recently advised the §812 Prospective project, however, to not include this in the §812 primary analysis at this time, primarily because the study is of a new endpoint and the results have not been replicated in other studies in the U.S. Consequently, our analysis includes infant mortality in the high set of assumptions.

Valuing Premature Mortality

To value the benefit of reducing premature mortality, we employ two approaches to the calculated change in incidence. One approach, the "value of statistical lives lost" (VSL) approach, uses information from several value-of-life studies to determine a reasonable benefit of preventing mortality. The mean value of avoiding one statistical death is estimated to be $5.9 million in 1997 dollars (or $4.8 million in 1990 dollars as has been used in previous EPA analyses). This represents an intermediate value from a variety of estimates that appear in the economics literature, and is a value that EPA has frequently used in RIAs for other rules. This estimate is the mean of a distribution fitted to the estimates from 26 value-of-life studies identified in the §812 study as "applicable to policy analysis." The approach and set of selected studies mirrors that of Viscusi (1992) (with the addition of two studies), and uses the same criteria used by Viscusi in his review of value-of-life studies. The $5.9 million estimate is consistent with Viscusi's conclusion (updated to 1997$) that "most of the reasonable estimates of the value of life are clustered in the $3.7 to $8.6 million range." Five of the 26 studies are contingent valuation (CV) studies, which directly solicit WTP information from subjects; the rest are wage-risk studies, which base WTP estimates on estimates of the additional compensation demanded in the labor market for riskier jobs. The 26 studies used to form the distribution of the value of a statistical life are listed in Table VII-10.

The second approach for valuing premature mortality is the value of statistical life-years lost" (VSLY) approach, which incorporates assumptions to account for the age-distribution of the affected population. Moore and Viscusi (1998) suggest one approach for determining the value of a statistical life-year lost. They assume that the willingness to pay to save a statistical life is the value of a single year of life times the expected number of years of life remaining for an individual. They suggest that a typical respondent in a mortal risk study may have a life expectancy of an additional 35 years. Using a mean estimate of $4.8 million (1990 dollars), their approach would yield an estimate of $137,000 per life-year lost or saved. If an individual discounts future additional years using a standard discounting procedure. Using a 35 year life expectancy, a $4.8 million value of a statistical life, and a 5 percent discount rate, the implied value of each life-year lost is $293,000. A higher discount rate would produce a greater value per life-year, and a lower discount rate would produce a lower value per life-year. The Moore and Viscusi procedure is identical to this approach, but uses a zero discount rate. In addition to the VSLY, the expected number of life-years saved is necessary to determine the appropriate value for an avoided incidence of premature mortality. Based on adjustments to reflect age-specific relative premature mortality is determined to be 9.8 years. Using 9.8 years, the value of an avoided incidence of PM-related premature mortality is then $2.2 million (1990$). Thus, for the low-end estimate of premature mortality in this analysis we apply the value of $2.7 million in

1997 dollars per life-year saved, and the high-end estimate applies $5.9 million per life to the full estimate of incidence.

Table VII-10. Summary of Mortality Valuation Estimates[a]

Study	*Type of Estimate*	*Valuation per Statistical Life (millions of 1990 $)*
Kneisner and Leeth (1991) (US)	Labor Market	0.7
Smith and Gilbert (1984)	Labor Market	0.9
Dillingham (1985)	Labor Market	1.1
Butler (1983)	Labor Market	1.4
Miller and Guria (1991)	Contingent Valuation	1.5
Moore and Viscusi (1988)	Labor Market	3.1
Viscusi et al. (1991)	Contingent Valuation	3.3
Gegax et al. (1985)	Contingent Valuation	4.1
Marin and Psacharopoulos (1982)	Labor Market	3.4
Kneisner and Leeth (1991) (Australia)	Labor Market	4.1
Gerking et al. (1988)	Contingent Valuation	4.2
Cousineau et al. (1988)	Labor Market	4.4
Jones-Lee (1989)	Contingent Valuation	4.7
Dillingham (1985)	Labor Market	4.8
Viscusi (1978; 1979)	Labor Market	5.0
R.S. Smith (1976)	Labor Market	5.6
V.K. Smith (1983)	Labor Market	5.8
Olson (1981)	Labor Market	6.4
Viscusi (1981)	Labor Market	8.0
R.S. Smith (1974)	Labor Market	8.8
Moore and Viscusi (1988)	Labor Market	9.0
Kneisner and Leeth (1991) (Japan)	Labor Market	9.3
Herzog and Schlottman (1987)	Labor Market	11.2
Leigh and Folson (1984)	Labor Market	11.9
Leigh (1987)	Labor Market	12.8
Garen (1988)	Labor Market	16.6

[a] Based on Viscusi (1992). The values in Viscusi have been updated to 1997 $, as detailed in (Abt Associates, 1999).

b. Chronic Bronchitis

There are a limited number of studies that have estimated the impact of air pollution on chronic bronchitis. An important hindrance is the lack of long-term health data and the associated air pollution levels. Schwartz (1993) and Abbey et al.(1993; 1995) provide the evidence that long-term PM exposure gives rise to the development of chronic bronchitis in the U.S. Following the NO_x SIP call analysis (U.S. EPA, 1998b), our analysis uses the Schwartz study to develop a C-R function linking PM to chronic bronchitis.

It should be noted that Schwartz used data on the *prevalence* of chronic bronchitis, not its *incidence*. To use Schwartz's study and still estimate the change in incidence, there are at least two possible approaches. The first is to simply assume that it is appropriate to use the baseline *incidence* of chronic bronchitis in a C-R function with the estimated coefficient from Schwartz's study, to directly estimate the change in incidence. The second is to estimate the percentage change in the prevalence rate for chronic bronchitis using the estimated coefficient from Schwartz's study in a C-R function, and then to assume that this percentage change applies to a baseline incidence rate obtained from another source. (That is, if the prevalence declines by 25 percent with a drop in PM, then baseline incidence drops by 25 percent with the same drop in PM.) Following work in the retrospective analysis of the Clean Air Act (U.S. EPA 1997a, pg. D-24), our analysis uses the former approach, and estimates the change in incidence using an annual incidence rate of 0.6 percent.

Valuing Chronic Bronchitis

PM-related chronic bronchitis is the only measured morbidity endpoint that may be expected to last from the initial onset of the illness throughout the rest of the individual's life. WTP to avoid chronic bronchitis would therefore be expected to incorporate the present discounted value of a potentially long stream of costs (e.g., medical expenditures and lost earnings) and pain and suffering associated with the illness. Two studies, Viscusi et al. (1991) and Krupnick and Cropper (1992), provide estimates of WTP to avoid a case of chronic bronchitis.

The Viscusi et al. and the Krupnick and Cropper studies were experimental studies intended to examine new methodologies for eliciting values for morbidity endpoints. Although these studies were not specifically designed for policy analysis, we believe the studies provide reasonable estimates of the WTP for chronic bronchitis. As with other contingent valuation studies, the reliability of the WTP estimates depends on the methods used to obtain the WTP values. Some specific attributes of the studies may raise some questions regarding their reliability. An alternative approach that can be use is the cost of illness (COI) approach, which considers only the expenditures on the illness as a valuation method. This approach, however, underestimates the true value of a change in incidence because it does not consider other components of the valuation such as the amount an individual would be willing to pay to avoid the illness even if they did not have medical expenses to consider. As such, it can serve as a lower bound of the value for chronic bronchitis. Therefore, this analysis values chronic bronchitis by using the COI approach in the low-end estimate and the WTP approach for the high-end estimate.

The COI approach for valuing chronic bronchitis uses average annual lost earnings and average annual medical expenditures reported in Krupnick and Cropper (1990). Using a 5 percent discount rate and assuming that (1) lost earnings continue to age 65, (2) medical expenditures are incurred until death, and (3) life expectancy is unchanged by chronic bronchitis, the present discounted value of the stream of medical expenditures and lost earnings associated with an average case of chronic bronchitis is estimated to be about $94,500 for a 30 year old, $about $71,200 for a 40 year old, about $73,000 for a 50 year old, and about $50,300 for a 60 year old. The midpoint of the COI estimates across the range of ages is $72,400 per case, which is used to value the low-end estimate of benefits for reduce incidence of chronic bronchitis.

For the WTP approach, we use two studies. The study by Viscusi et al. uses a sample that is larger and more representative of the general population than the study by Krupnick and Cropper (which selects people who have a relative with the disease). Thus, the valuation for the high-end estimate is based on the distribution of WTP responses from Viscusi et al. (1991). The WTP to avoid a case of pollution-related chronic bronchitis is derived by starting with the WTP to avoid a severe case of chronic bronchitis, as described by Viscusi et al. (1991)[r], and adjusting it downward to reflect (1) the decrease in severity of a case of pollution-related CB relative to the severe case described in the Viscusi et al. study, and (2) the elasticity of WTP with respect to severity reported in the Krupnick and Cropper (Krupnick et al., 1992) study. The technical support document describes the adjustment procedure in more detail (Abt Associates, 1999). The mean value of the adjusted distribution is $319,280. This is the WTP for chronic bronchitis we used in our benefits analysis.

As expected, the WTP estimate is greater than the full COI estimate in part because it reflects the willingness to pay to avoid the pain and suffering associated with the illness. Thus, the COI approach has a known downward bias because it does not include a measure of an individual's willingness to pay some amount to avoid the illness even if no medical expenses and no loss of earnings occurred. The WTP estimate of $319,280 is from 3.4 times the COI estimate for 30 year olds to 6.3 times the estimate for 60 year olds.

[r]As previously mentioned, the Schwartz (1993) study defines a case of chronic bronchitis for the purpose of estimating a concentration-response function. This function only examines the relationship of chronic bronchitis and PM without differentiating between severity of cases. Therefore, an adjustment for severity is necessary to value the benefits of reduced incidences.

c. Hospital Admissions

Both ozone and particulate matter have been associated with increased risk of premature mortality. Each is discussed below.

Estimating Ozone-related Hospital Admissions

Our analysis estimates ozone-related hospital admissions for "all respiratory diseases," using a C-R function based on the work of Thurston et al. (1992). Thurston et al. examined hospital admissions for all ages in the population. Because of the comprehensiveness of the Thurston et al. study, it is selected over other available studies that are restricted to limited age ranges (e.g., the population aged 65 years and older), and/or specific diagnoses (e.g., hospital admissions for pneumonia). The age- and disease-specific effect categories are subsets of the all-age, all-respiratory disease hospital admission category. Therefore, the benefits of avoided hospital admissions for respiratory illnesses for all ages should be larger than the benefits for more restricted categories. However, that is not true for the estimated benefits, based on the available studies. The estimated relationship produces fewer benefits than either of the two available alternatives: all respiratory disease admissions for the population over 65; or the sum of pneumonia and chronic obstructive pulmonary disease (COPD) admissions for the population over 65. Clearly adding the results for these study types would involve a serious amount of double counting. Therefore, selecting the Thurston et al. study may underestimate the total benefits of hospital admissions.

Estimating PM-related Hospital Admissions

The benefits analysis includes three PM-related hospital admissions, due to all respiratory illnesses (Thurston et al., 1994), congestive heart failure (Schwartz and Morris, 1995), and ischemic heart disease (Schwartz and Morris, 1995). As with ozone-induced hospital admissions, the benefits analysis relies on a study of all respiratory hospital admissions for all age groups, rather than studies examining the population over 65.

Valuing Hospital Admissions

An individual's WTP to avoid a hospital admission will include, at a minimum, the amount of money they pay for medical expenses (i.e., what they pay towards the hospital charge and the associated physician charge) and the loss in earnings. In addition, however, an individual is likely to be willing to pay some amount to avoid the pain and suffering associated with the illness itself. That is, even if they incurred no medical expenses and no loss in earnings, most individuals would still be willing to pay something to avoid the illness.

Because medical expenditures are to a significant extent shared by society, via medical insurance, Medicare, etc., the medical expenditures actually incurred by the individual are likely to be less than the total medical cost to society. The total value to society of an individual's avoidance of hospital admission, then, might be thought of as having two components: (1) the cost of illness (COI) to society, including the total medical costs plus the value of the lost productivity, as well as (2) the individual's WTP to avoid the illness itself.

In the absence of estimates of social WTP to avoid hospital admissions for specific illnesses (components 1 plus 2 above), estimates of total COI (component 1) are typically used as conservative (lower bound) estimates. Because these estimates do not include the value of avoiding the illness itself (component 2), they are biased downward. Some analyses adjust COI estimates upward by multiplying by an estimate of the ratio of WTP to COI, to better approximate total WTP. Other analyses have avoided making this adjustment because of the possibility of over adjusting -- that is, possibly replacing a known downward bias with an upward bias. The previous RIAs for PM and ozone, as well as the revised RIA for ozone and PM NAAQS, did adjust the COI estimate upward. The COI values used in this benefits analysis will not be adjusted to better reflect the total WTP. This is consistent with the guidance offered by the §812 Science Advisory Board (SAB) committee.

The COI estimates used in our analysis consist of three components: estimated physician charges (based on the average length of a hospital stay for the illness), the estimated opportunity cost of time spent in the hospital, and estimated hospital charges.

Our analysis assumes that physician charges associated with hospital care for asthma and chronic obstructive pulmonary disease (COPD) (two endpoints not estimated for this analysis) provide reasonably good estimates of average physician charges associated with hospital stays for the illness categories considered here. Abt Associates (1992) estimated that physician charges for the first day of hospital care for asthma (in 1988) or COPD (in 1989) averaged $135 (in 1997 $); physician charges for subsequent days of hospital care averaged $50. Estimated physician charges for a hospital stay of n days for any of the illness categories discussed below, then, would be $135 + $50(n-1).

The opportunity cost of a day spent in the hospital is estimated, for people in the workforce, as the value of the lost daily wage. This is estimated at $102. The study on PM and work loss days from which this value is derived (Ostro, 1987), however, considers only individuals 18 to 65 years old, while two of the hospital admission studies used in this analysis ("all respiratory, all ages", Thurston et al., 1994; and Thurston et al., 1992), considers all ages for both ozone and PM. It should be noted that, because the value of a PM-related work loss day (WLD) is elsewhere added into the total benefits analysis as a separate health endpoint, including it as a component of the WTP to avoid a PM-related hospital admission associated would be double counting. Additionally, because there is a not a separate work loss function for ozone, the lost productivity is included in the cost of an ozone hospital admission, but not for PM.

To derive estimates of the opportunity cost of a day spent in the hospital for respiratory illness based on Thurston et al. (1994) or Thurston et al. (1992), which considered individuals of all ages, we assumed that half of the PM- or ozone-related hospital admissions are among individuals who are not employed, including the young and the elderly.[5] We therefore estimated the expected opportunity cost of a day spent in the hospital for an individual randomly selected from among those admitted to the hospital for PM- or ozone-related respiratory illnesses to be (0.5)($102) +(0.5)($51) = $76.50. However, because the value of work loss days for those in the labor force is a separate component of the total benefit for PM, only the second component of opportunity cost enters the PM-related "all respiratory" hospital admissions benefit, which is, then, (0.5)($51) = $25.50.

To estimate the opportunity cost of a day spent in the hospital for an individual aged 65 or older (necessary for the ischemic heart disease and congestive heart failure hospital admission functions for individuals 65 years and over), we assumed that such an individual is not in the workforce. Although the value of a WLD may be an inappropriate way to estimate the opportunity cost of a day spent in the hospital for someone who is not employed (including the young and the elderly), this opportunity cost is positive and should not be ignored. As a rough approximation, we assumed that, for the young, the elderly, and any other unemployed individuals, the opportunity cost of a day spent in the hospital is one-half what it is for individuals in the workforce, or $51.

Finally, for all hospital admissions included in this analysis, we based estimates of hospital charges on discharge statistics provided by Elixhauser et al. (1993). The resulting Cost of Illness values for hospital admissions are shown in Table VII-11.

[5]This is approximately the same as the ratio of employed to total population in the United States. In 1994, for example, this ratio was (123 million)/(260 million), or 47 percent.

Table VII-11. Derivation of Cost of Illness (COI) and Total WTP Estimates for Hospital Admissions Endpoints (1997$[a])

Hospital Admissions For:	*Hospital Charge (1)*	*Physician Charge (2)*	*Opportunity Cost*			*Total Cost of Illness (COI) (1) + (2) + (3) (Standard Deviation)*
			Opportunity Cost per day	*Avg Length of Stay (days)*	*Total Opportunity Cost (3)*	
Ischemic Heart Disease, age ≥ 65 (ICD codes 410-414)	$13,996	$438	$50.96	7	$357	$14,791 ($126)
Congestive Heart Failure, age ≥ 65 (ICD code 428)	$10,854	$539	$50.96	9	$459	$11,852 ($166)
PM-Related "all respiratory illnesses," all ages (ICD codes 466, 480-482, 485, 490-493)	$8,414	$488	$25.48	8	$204	$9,106 ($115)
Ozone-Related "all respiratory illnesses," all ages (ICD codes 466, 480-486, 490-493)	$8,607	$438	$76.44	7	$535	$9,580 ($93)

[a] Note: Two different escalation factors were used in the adjustment to 1997$. Hospital and physician charges both used escalation factors based upon the CPI-U for medical care. The opportunity cost adjustment used an escalation factor base upon the CPI-U for "all items." The standard deviation in the Total Cost of Illness column is based upon a weighted average of each of the three COI components.

d. Acute Bronchitis

Dockery et al. (1989) examined the relationship between PM and other pollutants on the reported rates of chronic cough, bronchitis and chest illness, in a study of 5,422 children aged ten to twelve. Bronchitis and chronic cough were both found to be significantly related to PM concentrations.

Estimating WTP to avoid a case of acute bronchitis is difficult for several reasons. First, WTP to avoid acute bronchitis itself has not been estimated. Estimation of WTP to avoid this health endpoint therefore must be based on estimates of WTP to avoid symptoms that occur with this illness. Second, a case of acute bronchitis may last more than one day, whereas it is a day of avoided symptoms that is typically valued. Finally, the C-R function used in the benefit analysis for acute bronchitis was estimated for children, whereas WTP estimates for those symptoms associated with acute bronchitis were obtained from adults.

With these caveats in mind, we estimate WTP to avoid a case of acute bronchitis as the midpoint between a low estimate and a high estimate. The low estimate ($16.32) is the sum of the midrange values recommended by IEc (1994) for two symptoms believed to be associated with acute bronchitis: coughing ($7.72) and chest tightness ($8.60). The high estimate was taken to be twice the value of a minor respiratory restricted activity day ($47.12), or $94.24. The midpoint between the low and high estimates is $55.26.

e. PM-related Upper Respiratory Symptoms

The benefits analysis used the C-R function for PM-related Upper Respiratory Symptoms (URS) from Pope et al. (1991). Pope et al. describe URS as consisting of one or more of the following symptoms: runny or stuffy nose; wet cough; and burning, aching, or red eyes. The children in the Pope et al. study were asked to record respiratory symptoms in a daily diary, and the daily occurrences of URS and LRS, as defined above, were related to daily PM_{10} concentrations. Estimates of WTP to avoid a day of symptoms are therefore appropriate measures of benefit.

Willingness to pay to avoid a day of URS is based on symptom-specific WTPs to avoid those symptoms identified by Pope et al. as part of the URS complex of symptoms. Three contingent valuation (CV) studies have estimated WTP to avoid various morbidity symptoms that are either within the URS symptom complex defined by Pope et al. (1991) or are similar to those symptoms identified by Pope et al. In each CV study, participants were asked their WTP to avoid a day of each of several symptoms. The three individual symptoms that were identified as most closely matching those listed by Pope et al. for URS are cough, head/sinus congestion, and eye irritation. A day of URS could consist of any one of seven possible “symptom complexes” consisting of at least one of these symptoms. It is assumed that each of the seven types of URS is equally likely. The mean WTP to avoid a day of URS is therefore the average of the mean WTPs to avoid each type of URS, or $22.96. This is the point estimate for the dollar value for PM-related URS used in the benefit analysis. Finally, it is worth emphasizing that what is being valued here is URS *as defined by Pope et al.* While other definitions of URS are certainly possible, we used this definition of URS in the benefits analysis because it is the incidence of this specific definition of URS that has been related to PM exposure by Pope et al.(1991).

f. PM-related Lower Respiratory Symptoms

Schwartz et al. (1994) estimated the relationship between Lower Respiratory Symptoms (LRS) and PM-10 concentrations. The method for deriving a point estimate of mean WTP to avoid a day of LRS is the same as for URS. Schwartz et al. define LRS as at least two of the following symptoms: cough, chest pain, phlegm, and wheeze. The symptoms for which WTP estimates are available that reasonably match those listed by Schwartz et al. for LRS are cough (C), chest tightness (CT), coughing up phlegm (CP), and wheeze (W). A day of LRS, as defined by Schwartz et al., could consist of any one of the 11 combinations of at least two of these four symptoms.

We assumed that each of the eleven types of LRS is equally likely. The mean WTP to avoid a day of LRS as defined by Schwartz et al. (1994) is therefore the average of the mean WTPs to avoid each type of LRS, or $14.51. This is the point estimate used in the benefit analysis for the dollar value for LRS as defined by Schwartz et al. The WTP estimates are based on studies which considered the value of a *day* of avoided symptoms, whereas the Schwartz et al. study used as its measure a *case* of LRS. Because a case of LRS usually lasts at least one day, and often more, WTP to avoid a day of LRS should be a conservative estimate of WTP to avoid a case of LRS.

Finally, as with URS, it is worth emphasizing that what is being valued here is LRS *as defined by Schwartz et al. (1994).* While other definitions of LRS are certainly possible, this definition of LRS is used in this benefit analysis because it is the incidence of this specific definition of LRS that has been related to PM exposure by Schwartz et al.

The point estimates derived for mean WTP to avoid a day of URS and a case of LRS are based on the assumption that WTPs are additive. For example, if WTP to avoid a day of cough is $8.60, and WTP to avoid a day of shortness of breath is $6.14, then WTP to avoid a day of both cough and shortness of breath is $14.74. If there are no synergistic effects among symptoms, then it is likely that the marginal utility of avoiding symptoms decreases with the number of symptoms being avoided. If this is the case, adding WTPs would tend to overestimate WTP for avoidance of multiple symptoms. However, there may be synergistic effects– that is, the discomfort from two or more simultaneous symptoms may exceed the sum of the discomforts associated with each of the individual symptoms. If this is the case, adding WTPs would tend to underestimate WTP for avoidance of multiple symptoms. It is also possible that people may experience additional symptoms for which WTPs are not available, again leading to an underestimate of the correct WTP. However, for small numbers of symptoms, the assumption of additivity of WTPs is unlikely to result in substantive bias.

There are three sources of uncertainty in the valuation of both URS and LRS: (1) an occurrence of URS or of LRS may be comprised of one or more of a variety of symptoms (i.e., URS and LRS are each potentially a “complex of symptoms”), so that what is being valued may vary from one occurrence to another; (2) for a given symptom, there is uncertainty about the mean WTP to avoid the symptom; and (3) the WTP to avoid an occurrence of multiple symptoms may be greater or less than the sum of the WTPs to avoid the individual symptoms.

g. Ozone-related Any of 19 Respiratory Symptoms

The presence of “any of 19 acute respiratory symptoms” is a somewhat subjective health effect used by Krupnick et al. (1990). Moreover, not all 19 symptoms are listed in the Krupnick et al. study. It is therefore not clear exactly what symptoms were included in the study. Even if all 19 symptoms were known, it is unlikely that WTP estimates could be obtained for all of the symptoms. Finally, even if all 19 symptoms were known and WTP estimates could be obtained for all 19 symptoms, the assumption of additivity of WTPs becomes tenuous with such a large number of symptoms. The likelihood that all 19 symptoms would occur simultaneously, moreover, is very small.

Acute respiratory symptoms must be either upper respiratory symptoms or lower respiratory symptoms. In the absence of further knowledge about which of the two types of symptoms is more likely to occur among the “any of 19 acute respiratory symptoms,” we assumed that they occur with equal probability. Because this health endpoint may also consist of combinations of symptoms, it was also assumed that there is some (smaller) probability that upper and lower respiratory symptoms occur together.

To value avoidance of a day of “the presence of any of 19 acute respiratory symptoms” we therefore assumed that this health endpoint consists either of URS, or LRS, or both. We also assumed that it is as likely to be URS as LRS and that it is half as likely to be both together. That is, it was assumed that “the presence of any of 19 acute respiratory symptoms” is a day of URS with 40 percent probability, a day of LRS with 40 percent probability, and a day of both URS and LRS with 20 percent probability. Using the point estimates of WTP to avoid a day of URS and LRS derived above, the point estimate of WTP to avoid a day of “the presence of any of 19 acute respiratory symptoms” is:

$$(0.40)(\$22.96) + (0.40)(\$14.51) + (0.20)(\$22.96 + \$14.51) = \$22.48$$

Because this health endpoint is only vaguely defined, and because of the lack of information on the relative frequencies of the different combinations of acute respiratory symptoms that might qualify as "any of 19 acute respiratory symptoms," the unit dollar value derived for this health endpoint must be considered only a rough approximation.

h. Work Loss Days

Ostro (1987) estimated the impact of PM on the incidence of work-loss days (WLD) in a national sample of the adult working population, ages 18 to 65, living in metropolitan areas. Separate coefficients were developed for each year in the analysis (1976-1981); we then combined these coefficients for use in this analysis.

Willingness to pay to avoid the loss of one day of work was estimated by dividing the median weekly wage for 1990 (U.S. Bureau of the Census, 1992) by five (to get the median daily wage). This values the loss of a day of work at the median wage for the day lost. Valuing the loss of a day's work at the wages lost is consistent with economic theory, which assumes that an individual is paid exactly the value of his labor.

The use of the median rather than the mean, however, requires some comment. If all individuals in society were equally likely to be affected by air pollution to the extent that they lose a day of work because of it, then the appropriate measure of the value of a work loss day would be the mean daily wage. It is highly likely, however, that the loss of work days due to pollution exposure does not occur with equal probability among all individuals, but instead is more likely to occur among lower income individuals than among high income individuals. It is probable, for example, that individuals who are vulnerable enough to the negative effects of air pollution to lose a day of work as a result of exposure tend to be those with generally poorer health care. Individuals with poorer health care have, on average, lower incomes. To estimate the average lost wages of individuals who lose a day of work because of exposure to PM pollution, then, would require a weighted average of all daily wages, with higher weights on the low end of the wage scale and lower weights on the high end of the wage scale. Because the appropriate weights are not known, however, the median wage was used rather than the mean wage. The median is more likely to approximate the correct value than the mean because means are highly susceptible to the influence of large values in the tail of a distribution (in this case, the small percentage of very large incomes in the United States), whereas the median is not susceptible to these large values. The median daily wage in 1990 was $101.92 (adjusted to 1997 $). This is the value that was used to represent work loss days (WLD).

i. Minor Restricted Activity Days

Ostro and Rothschild (1989) estimated the impact of $PM_{2.5}$ on the incidence of minor restricted activity days (MRAD) in a national sample of the adult working population, ages 18 to 65, living in metropolitan areas. We developed separate coefficients for each year in the analysis (1976-1981), which were then combined for use in this analysis.

No studies are reported to have estimated WTP to avoid a minor restricted activity day (MRAD). However, IEc (1993) has derived an estimate of WTP to avoid a minor respiratory restricted activity day (MRRAD), using WTP estimates from Tolley et al. (1986) for avoiding a three-symptom combination of coughing, throat congestion, and sinusitis. This estimate of WTP to avoid a MRRAD, so defined, is $47.12. Although Ostro and Rothschild (1989) estimated the relationship between $PM_{2.5}$ and MRADs, rather than MRRADs (a component of MRADs), it is likely that most of the MRADs associated with exposure to $PM_{2.5}$ are in fact MRRADs. For the purpose of valuing this health endpoint, then, we assumed that MRADs associated with PM exposure may be more specifically defined as MRRADs, and therefore used the estimate of mean WTP to avoid a MRRAD.

Any estimate of mean WTP to avoid a MRRAD (or any other type of restricted activity day other than WLD) will be somewhat arbitrary because the endpoint itself is not precisely defined. Many different combinations of symptoms could presumably result in some minor or less minor restriction in activity. Krupnick and Kopp (1988) argued that mild symptoms will not be sufficient to result in a MRRAD, so that WTP to avoid a MRRAD should exceed WTP to avoid any single mild symptom. A single severe symptom or a combination of symptoms could, however, be sufficient to restrict activity. Therefore WTP to avoid a MRRAD should, these authors argue, not necessarily exceed WTP to avoid a single severe symptom or a combination of symptoms. The “severity” of a symptom, however, is similarly not precisely defined; moreover, one level of severity of a symptom could induce restriction of activity for one individual while not doing so for another. The same is true for any particular combination of symptoms.

Given that there is inherently a substantial degree of arbitrariness in any point estimate of WTP to avoid a MRRAD (or other kinds of restricted activity days), the reasonable bounds on such an estimate must be considered. By definition, a MRRAD does not result in loss of work. WTP to avoid a MRRAD should therefore be less than WTP to avoid a WLD. At the other extreme, WTP to avoid a MRRAD should exceed WTP to avoid a single mild symptom. The highest IEc midrange estimate of WTP to avoid a single symptom is $19.30, for eye irritation. The point estimate of WTP to avoid a WLD in the benefit analysis is $101.92. If all the single symptoms evaluated by the studies are not severe, then the estimate of WTP to avoid a MRRAD should be somewhere between $19.30 and $101.92. Because the IEc estimate of $47.12 falls within this range (and acknowledging the degree of arbitrariness associated with any estimate within this range), we used the IEc estimate as the point estimate of mean WTP to avoid a MRRAD.

j. Worker Productivity

The benefits analysis based the valuation used to monetize benefits associated with increased worker productivity resulting from improved ozone air quality on information reported in Crocker and Horst (1981) and summarized in EPA (1994). Crocker and Horst (1981) examined the impacts of ozone exposure on the productivity of outdoor citrus workers. The study measured productivity impacts as the change in income associated with a change in ozone exposure, given as the elasticity of income with respect to ozone concentration (-0.1427). The reported elasticity translates a ten percent reduction in ozone to a 1.4 percent increase in income. Given the average daily income for outdoor workers engaged in strenuous activity reported by the 1990 U.S. Census, $89.64 per day (adjusted to 1997 $), a ten percent reduction in ozone yields approximately $1 in increased daily wages.

4. Ozone- and PM-Related Welfare Effects

In addition to the effects on human health described above, emission reductions attributed to the Tier 2 rule will also produce welfare (i.e., non-health) benefits. Welfare effects cover a potentially broad range of adverse effects, including adverse impacts on plants, animals, structural materials, visibility, and ecosystem functions. Like health effects, in order to be included in a quantified monetary benefits analysis, all of the analytical links between changes in emissions and the monetary value of the effects must be available. While the required analytical components are available for certain welfare endpoints, our analysis omits many other likely or possible welfare categories. The availability of information on each analytical step limits the total coverage of the welfare effects. All of the welfare benefits that are quantified and included in the benefits analysis were included in the NO_x SIP call. Table VII-12 lists the welfare categories that are included in the benefits analysis; the technical support document for this RIA provides further detail on these endpoints (Abt Associates, 1999). Each of these categories will be discussed separately below.

Table VII-12. Quantified Welfare Effects Included in the Benefits Analysis

Welfare Effect	*Pollutant*	*Study*
Agriculture - commodity crops	Ozone	Taylor (1993)
Nitrogen deposition in estuarine and coastal waters	NO_x	EPA (1998a)
Visibility-recreational	PM and gases	Chestnut et al. (1997)
Visibility-residential	PM and gases	McClelland et al. (1991)
Household soiling	PM	ESEERCO (1994)

a. Commodity Agricultural Crops

The economic value associated with varying levels of yield loss for ozone-sensitive commodity crops is analyzed using the AGSIM© agricultural benefits model (Taylor et al., 1993). AGSIM© is an econometric-simulation model that is based on a large set of statistically estimated demand and supply equations for agricultural commodities produced in the United

States. The model is capable of analyzing the effects of changes in policies (in this case, the implementation of the Tier 2 rule) that affect commodity crop yields or production costs. The technical support document for this RIA provides further details on AGSIM© (Abt Associates, 1999).

The measure of benefits calculated by the model is the net change in consumers' and producers' surplus from baseline ozone concentrations to the ozone concentrations resulting from attainment of particular standards. Using the baseline and post-control equilibria, the model calculates the change in net consumers' and producers' surplus on a crop-by-crop basis[t]. Dollar values are aggregated across crops for each standard. The total dollar value represents a measure of the change in social welfare associated with the Tier 2 rule. Although the model calculates benefits under three alternative welfare measures (perfect competition, price supports, and modified agricultural policy), results presented here are based on the "perfect competition" measure to reflect recent changes in agricultural subsidy programs. Under the recently revised 1996 Farm Bill , most eligible farmers have enrolled in the program to phase out government crop price supports for the AGSIM©-relevant crops: wheat, corn, sorghum, and cotton.

For the purpose of our analysis, the model analyzed the six most economically significant crops: corn, cotton, peanuts, sorghum, soybean, and winter wheat.[u] The model employs biological exposure-response information derived from controlled experiments conducted by the National Crop Loss Assessment Network (NCLAN) (1996).

b. Nitrogen Deposition

Excess nutrient loads, especially that of nitrogen, cause a variety of adverse consequences to the health of estuarine and coastal waters. These effects include toxic and/or noxious algal blooms such as brown and red tides, low (hypoxic) or zero (anoxic) concentrations of dissolved oxygen in bottom waters, the loss of submerged aquatic vegetation due to the light-filtering effect of thick algal mats, and fundamental shifts in phytoplankton community structure. Direct C-R functions relating deposited nitrogen and reductions in estuarine benefits are not available. The preferred willingness-to-pay based measure of benefits depends on the availability of these C-R functions and on estimates of the value of environmental responses. Because neither appropriate C-R functions nor sufficient information to estimate the marginal value of changes in water quality exist at present, this analysis used an avoided cost approach instead of willingness-to-pay to generate estuary-related benefits. The use of the avoided cost approach to establish the value

[t] Agricultural benefits differ from other health and welfare endpoints in the length of the assumed ozone season. For agriculture, the ozone season is assumed to extend from April to September. This assumption is made to ensure proper calculation of the ozone statistic used in the exposure-response functions. The only crop affected by changes in ozone during April is winter wheat.

[u] The total value for these crops in 1997 was $57 billion.

of a reduction in nitrogen deposition is problematic, because there is not a direct link between implementation of the air pollution regulation and the abandonment of a separate costly regulatory program by some other agency, (i.e. a state environmental agency). However, there are currently no readily available alternatives to this approach.[v]

The avoided costs to surrounding communities of reduced nitrogen loadings were calculated for three case study estuaries.[w] These costs are used to estimate the avoided costs for ten East Coast estuaries, and two Gulf Coast case study estuaries for which reduced nitrogen loadings were modeled.[x] The avoided cost estimates for the ten East Coast case study estuaries, which represent approximately half of the estuarine watershed area in square miles along the East Coast, are then used to extrapolate avoided costs to all East Coast estuaries. The three case study estuaries are chosen because they have agreed upon nitrogen reduction goals and the necessary nitrogen control cost data. The remaining estuaries in this analysis are chosen based on their potential representativeness and our ability to estimate the direct and indirect nitrogen load from atmospheric deposition.

Our analysis values atmospheric nitrogen reductions on the basis of avoided costs associated with agreed upon controls of nonpoint water pollution sources. We estimated benefits using a weighted-average, locally-based cost for nitrogen removal from water pollution (U.S. EPA, 1998a). Valuation reflects water pollution control cost avoidance based on the weighted average cost/pound of current non-point source water pollution controls for nitrogen in the three case study estuaries. Taking the weighted cost/pound of these available controls assumes States will combine low cost and high cost controls, which could inflate avoided cost estimates.

Reductions in nitrogen deposition from the Tier 2 rule should impact estuaries all along the eastern seaboard and the Gulf Coast. Nitrogen reduction programs are currently targeting many of the estuaries in these areas due to current impairment of estuarine water quality by excess nutrients. Some of the largest of these estuaries, including the Chesapeake Bay, have established goals for nitrogen reduction and target dates by which these goals should be achieved. Using the best and most easily implemented existing technologies, many of the estuaries will not be able to achieve the stated goals by the target dates. Meeting these additional reductions will

[v] Avoided cost is only a proxy for benefits, and should be viewed as inferior to willingness-to-pay based measures. Current research is underway to develop other approaches for valuing estuarine benefits, including contingent valuation and hedonic property studies. However, this research is still sparse, and does not contain sufficient information on the marginal willingness-to-pay for changes in concentrations of nitrogen (or changes in water quality or water resources as a result of changes in nitrogen concentrations).

[w] The case study estuaries are Albemarle-Pamlico Sounds, Chesapeake Bay, and Tampa Bay.

[x] The ten East Coast estuaries are Albemarle-Pamlico Sounds, Cape Cod Bay, Chesapeake Bay, Delaware Bay, Delaware Inland Bays, Gardiners Bay, Hudson River/Raritan Bay, Long Island Sound, Massachusetts Bays, and Narragansett Bays. The Gulf Coast estuaries are Sarasota Bay and Tampa Bay.

require development of new technologies, implementation of costly existing technologies (such as stormwater controls), or use of technologies with significant implementation difficulties, such as agricultural best management practices (BMPs). Reductions in nitrogen deposition from the atmosphere will directly reduce the need for these additional costly controls. Thus, while the Tier 2 rule does not totally eliminate the need for nutrient management programs already in place, it may substitute for some of the incremental costs and programs (such as an agricultural BMP program) necessary to meet the nutrient reduction goals for each estuary.

The fixed capital costs for non-point controls in the case study estuaries ranged from $0.75 to $55.59 per pound for agricultural and other rural best management practices and from $42.98 to $175.16 per pound for urban nonpoint source controls (stormwater controls, reservoir management, onsite disposal system changes, onsite BMPs).[y] Using these as a base, we calculated the total fixed capital cost per pound (weighted on the basis of fractional relationship of nitrogen load controlled for the estuary goal) for each of the case-study estuaries and applied in the valuation of their avoided nitrogen load controlled. The weighted capital costs per pound for the case-study estuaries are $40.95 for Albemarle-Pamlico Sounds, $26.79 for Chesapeake Bay, and $108.36 for Tampa Bay[z]. For the purposes of our analysis, EPA assumes that estuaries that have not yet established nutrient reduction goals will utilize the same types of nutrient management programs as projected for the case study estuaries. For the other nine estuaries, an average capital cost per pound of nitrogen (from the three case-estuaries) of $58.70/lb is calculated and applied; it is unclear whether this cost understates or overstates the costs associated with reductions in these other estuaries. The other nine estuaries generally represent smaller, more urban estuaries (like Tampa Bay), which typically have fewer technical and financial options available to control nitrogen loadings from nonpoint sources. This may result in higher control costs more similar to the Tampa Bay case. On the other hand, these estuaries may have opportunities to achieve additional point source controls at a lower costs. Also, increased public awareness of nutrification issues and technological innovation may, in the future, result in States finding lower cost solutions to nitrogen removal.

The benefits analysis assumed that the ten included East Coast estuaries are highly or moderately nutrient sensitive, and they represent approximately 45.46 percent of all estuarine watershed area along the East Coast.[aa] Because NOAA data indicate that approximately 92.6 percent of the watershed and surface area of East Coast estuaries are highly or moderately nutrient sensitive, it is reasonable to expect that East Coast estuaries not included in this analysis

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[y]The figures in the original work have been updated to 1997 $ using an all-good CPI index.

[z] The value for Tampa Bay is not a true weighted cost per pound, but a midpoint of a range of $71.89 to $144.47 developed by Apogee Research for the control possibilities (mostly urban BMPs) in the Tampa Bay estuary.

[aa] There are 43 East Coast estuaries of which ten were in the sample, and 31 Gulf of Mexico estuaries of which two are in the sample.

would also benefit from reduced deposition of atmospheric nitrogen. Therefore, we scaled-up total benefits from the ten representative East Coast estuaries to include the remainder of the nutrient sensitive estuaries along the East Coast on the basis of estuary watershed plus water surface area. Since the ten estuaries are assumed to be nutrient sensitive and account for 48 percent of total eastern estuarine area, we scaled-up estimates by multiplying the estimate for the ten East Coast estuaries by 2.037 (equal to 92.6 percent divided by 45.46 percent). We then added this figure to the benefits estimated for the two Gulf Coast estuaries for a total benefits estimate for nitrogen deposition.

We then annualized all capital cost estimates based on a seven percent discount rate and a typical implementation horizon for control strategies. Based on information from the three case study estuaries, this typically ranges from five to ten years. EPA has used the midpoint of 7.5 years for annualization, which yields an annualization factor of 0.1759. Non-capital installation costs and annual operating and maintenance costs are not included in these annual cost estimates. Depending upon the control strategy, these costs can be significant. Reports on the Albemarle-Pamlico Sounds indicate, for instance, that planning costs associated with control measures comprises approximately 15 percent of capital costs. Information received from the Association of National Estuary Programs indicates that operating and maintenance costs are about 30 percent of capital costs, and that permitting, monitoring, and inspections costs are about one to two percent of capital costs. For these reasons, the annual cost estimates may be understated.

c. Household Soiling Damage

Welfare benefits also accrue from avoided air pollution damage, both aesthetic and structural, to architectural materials and to culturally important articles. At this time, data limitations preclude the ability to quantify benefits for all materials whose deterioration may be promoted and accelerated by air pollution exposure. However, our analysis addresses one small effect in this category, the soiling of households by particulate matter.

Assumptions regarding the air quality indicator are necessary to evaluate the C-R function. PM_{10} and $PM_{2.5}$ are both components of TSP. However, it is not clear which components of TSP cause household soiling damage. The Criteria Document cites some evidence that smaller particles may be primarily responsible, in which case these estimates are conservative.

Several studies have provided estimates of the cost to households of PM soiling. The study that is cited by ESEERCO (1994) as one of the most sophisticated and is relied upon by EPA in its 1988 Regulatory Impact Analysis for SO_2 is Manuel et al. (1982). Using a household production function approach and household expenditure data from the 1972-73 Bureau of Labor Statistics Consumer Expenditure Survey for over twenty cities in the United States, Manuel et al. estimate the annual cost of cleaning per $\mu g/m^3$ PM per household as $1.55 ($0.59 per person times 2.63 persons per household). This estimate is low compared with others (e.g., estimates provided by Cummings et al. (1981) and Watson and Jaksch (1982) are about eight times and five times greater, respectively). The ESEERCO report notes, however, that the Manuel et al. estimate is probably downward biased because it does not include the time cost of do-it-yourselfers. Estimating that these costs may comprise at least half the cost of PM-related cleaning costs, they double the Manuel et al. estimate to obtain a point estimate of $3.09 (reported by ESEERCO in 1992 dollars as $2.70).

d. Visibility

Visibility effects reported earlier in this chapter are described in terms of changes in deciview, a unitless measure useful for comparing the effects of air quality on visibility. This measure is used in the WTP function for visibility and is directly related to two other common visibility measures: visual range (measured in km) and light extinction (measured in km^{-1}). Modeled changes in visibility are measured in terms of changes in light extinction, which are then transformed into deciviews. A change of one deciview represents a change of approximately 10 percent in the light extinction budget, "which is a small but perceptible scenic change under many circumstances." (Sisler, 1996) A change of less than 10 percent in the light extinction budget represents a measurable improvement in visibility, but may not be perceptible to the eye in many cases. All of the average regional changes in visibility are substantially less than one deciview (i.e. less than 10 percent of the light extinction budget), and thus less than perceptible. However, this does not mean that these changes are not real or significant. Our assumption is then that individuals can place values on changes in visibility that may not be perceptible. This is quite plausible if individuals are aware that many regulations lead to small improvements in visibility which when considered together amount to perceptible changes in visibility.

The analysis derives the residential visibility valuation estimate from the results of an visibility study (McClelland et al., 1991). We derive a household WTP value by dividing the value reported in McClelland et al. by the corresponding hypothesized change in deciview, yielding an estimate of $17 per unit change in deciview. Due to the somewhat dated methods used in the McClelland study and inconsistencies of the study with current best practices for conducting contingent valuation studies, the reliability of the results of the McClelland is uncertain. EPA recognizes these uncertainties, but believes a non-zero value exists for residential visibility improvements. Without alternative studies to verify the reliability of the WTP estimate from McClelland, the low-end estimate in this analysis does not value residential visibility while the high-end estimate uses the $17 per unit change in deciview obtained from the study. This value is applied to all households – including any households living in or around national parks– in any area estimated to experience a change in visibility.

A separate valuation component is needed for valuing improvements in visibility in national parks and other areas (collectively known as "Class I areas"). Chestnut (1997) developed a method for estimating the value to the U.S. public of visibility improvements in Class I visibility areas. The approach was based on the results of a 1990 Cooperative Agreement project jointly funded by the EPA and the National Park Service, "Preservation Values For Visibility Protection at the National Parks." Based on that contingent valuation study of visibility improvements, Chestnut calculates a household WTP for visibility improvements in Class I-area National Parks, capturing both use and non-use recreational values, and accounts for geographic variations in the willingness to pay. The PM and ozone NAAQS RIA (U.S. EPA, 1997b) analysis used this method. Similar to the McClelland study, the reliability of the results of the Chestnut study are uncertain because of inconsistency with certain elements of best practices for conducting contingent valuation. Contingent valuation is a rapidly developing field and new methodologies for study design are continually evolving. As such, studies developed during the late 1980's and early 1990's may differ in some elements of study design from more recent studies. EPA recognizes that there are some important aspects of the Chestnut study that are still useful for providing valuations associated with recreational visibility improvements. In the author's judgement, the WTP value derived in the Chestnut study "*may be indicative of an accuracy no better than ±50 percent* (Chestnut and Rowe, 1990)." Due to these uncertainties, the low-end estimate presents a conservative estimate of WTP for recreational visibility improvements that reflects the lower-bound of the variation (-50 percent).

More specifically, the Preservation Values study examined the demand for visibility in Class I-area National Parks in three broad regions of the country, California, Southwest, and Southeast. Because the Tier 2 rule has an impact on ambient pollution in all states – even in California, due to drift from neighboring states – all three regions are relevant to the visibility analysis. For a given region, the Preservation Values study asked respondents in Arizona, California, Missouri, New York and Virginia for their willingness to pay to protect visibility at National Parks in that region. Table VII-13 lists the parks included in the study in the study regions, as well as the parks in other regions specifically mentioned in the Preservation Values study. These other parks are used in estimating the visibility benefits in the "transfer regions", as described below.

Table VII-13. Class I Areas Included in Visibility Study By Region

Visibility Region	*National Parks*
California & Nevada	**Yosemite**, Sequoia/Kings Canyon, Redwoods, Pinnacles, Lava Beds, Death Valley, Lassen Volcanic, Joshua Tree, Point Reyes
Southwest	**Grand Canyon**, Mesa Verde, Arches, Bandelier, Capitol Reef, Carlsbad Caverns, Bryce Canyon, Chiricahua, Zion, Saguaro, Canyonlands, Petrified Forest, Rocky Mountain
Southeast	**Shenandoah**, Great Smoky Mountains, Mammoth Cave, Everglades
Transfer Region	*National Parks*
Northwest (transfer from California & Nevada)	Crater Lake, Mount Rainier, North Cascades, Olympic
North Central	**Yellowstone/Grand Tetons**, Badlands, Craters of the Moon, Glacier, Theodore Roosevelt, Wind Cave
Northeast	Acadia, Big Bend, Guadalupe Mountains, Isle Royale, Voyageurs

Note: The “indicator” park (where identified) is shown in bold for each regions. In each case the indicator park is a well-known park in that region. Source: Chestnut (1997).

Photos from each region’s “indicator park” were provided as part of the survey instrument. After a number of preparatory questions, respondents reached the WTP section of the survey. Respondents were first instructed that their answer to the WTP question applied only to the region in their survey, and that they did not have to worry about other regions of the country. After furnishing their WTP, respondents were asked what portion of their stated total value was for visibility at the indicator park alone. To avoid including benefits outside of the region, the reported answers were appropriately adjusted. All of these safeguards make it less likely that there will be overlap between urban (i.e., “residential”) and National Park (i.e., “recreational”) visibility benefits.[bb]

When estimating the benefits attributable to visibility improvement at specific Class I recreational parks, adjustments can be made to account for the location of parks, whether the people valuing the park live “in-region” or “out-of-region,” and whether or not the park is an “indicator park.” These issues are discussed below.

First, because the regional distribution of national parks throughout the U.S. is so varied, the estimated WTP coefficient per change in deciview changes in value depending upon the location of the Class I area. Based on the National Parks Visibility Valuation Study (Chestnut

[bb]There are a number of Class I areas in each region that are *not* National Parks (e.g., Florida’s Okefenokee Wilderness Area), and are thus not included in the estimated value for visibility. We do not attempt to estimate WTP for these other areas, and simply note that they are omitted.

and Rowe, 1990), Chestnut (1997, p. 10) estimated coefficients for the three study visibility regions: California, Southwest, and Southeast. To account for national parks in the rest of the contiguous U.S., however, the same coefficients are transferred to value visibility changes in parks located in adjacent regions, termed here as “transfer regions.” Table VII-14 displays the “in-region” and “out-of-region” coefficients used in each of the different visibility regions.

Table VII-14. Estimated Coefficients Used in the Valuation of WTP for Improved Visibility

*Study Visibility Regions and Transfer Visibility Regions**	*Estimated β for Out-of-Region Households*	*Estimated β for In-Region Households*
California & Northwest	$8.96	$12.89
Southwest & Centralwest	$13.51	$16.82
Southeast & Northeast	$4.91	$7.98

* Transfer regions are groups of states adjacent to the study region from which WTP values are assigned.

The in-region coefficient estimates the WTP of residents within a given visibility region for visibility improvements at all parks located within that same region. The out-of-region coefficient estimates the WTP of residents living outside a given visibility region for visibility improvements at all parks located within that region. The results of the survey suggest that in-region residents are likely to value visibility improvements at their parks more than out-of-region residents. This is consistent with expectations, as in-region households are more likely to visit, know about, and care for these parks.

Because the WTP coefficients are for visibility improvements at more than one park within a given visibility region, the WTP values must be apportioned between parks within a given visibility region. Our analysis assumes that WTP for visibility is related to a park's number of visitors. This is clearly a very crude approximation, since the WTP that we are attempting to estimate includes both use and non-use values, and a visitation rate is a better measure of use value and is not clearly linked to non-use values. On the other hand, short of conducting a survey for individual parks, it is difficult to estimate the relative importance of visibility at each park, and using a visitation rate to weight seems more appropriate than taking a simple average or using some other weighting metric, such as the size of the park.

For each study visibility region, we sum 1997 visitor-days at each Class I park. We then divide this total visitation figure into a WTP coefficient (in- or out-of-region, as appropriate) to create a WTP per visitor-days for the entire study region. Multiplying this new value by each park’s own number of 1997 visitor-days yields an apportioned per-park WTP coefficient for each park present in the study visibility region. Thus, we apply a visibility valuation function from a study region to an extrapolated, transfer region.

For aggregate benefits, the low-end estimate does not value residential visibility and uses the lower-bound estimate for recreational visibility for each region. In the high set of valuation assumptions, total visibility benefits consist of residential visibility benefits, as well as in- and out-of-region recreational visibility benefits (using the WTP estimates reported by Chestnut without adjustments to reflect the upper-bound of variation).

e. Ozone- and PM-related Welfare Effect Benefits Estimation

Table VII-15 presents estimates of the monetary benefits arising from each of the welfare endpoints associated with the air quality changes attributed to the Tier 2 rule.

Table VII-15. Welfare Endpoint Monetary Benefits

Endpoint	*Pollutant*	*Monetary Benefits (millions 1997$)*	
		Low	*High*
Agricultural crop damage	Ozone	-1	301
Nitrogen deposition	NO_x	200	200
Household soiling damage	PM	60.1	60.1
Visibility			
Out-of-region recreational	PM and gases	266.33	266.33
In-region recreational	PM and gases	64.10	64.10
Residential	PM and gases	not valued	371.02

5. Total Aggregated Benefits

In our analysis, we aggregated dollar benefits associated with each of the effects examined, such as hospital admissions, into a total benefits estimate assuming that none of the included health and welfare effects overlap. The point estimate of the total benefits associated with the health and welfare effects in each set (low and high) is just the sum of the separate effects estimates. The estimate of total benefits may be thought of as the end result of a sequential process in which, at each step, the estimate of benefits from an additional source is added. Each time an estimate of dollar benefits from a new source (e.g., a new health effect) is added to the previous estimate of total dollar benefits, the estimated total dollar benefits increases. The uncertainty surrounding the estimate of total dollar benefits, however, also increases.

A significant portion of the uncertainty in the benefit estimate derives from uncertainty about the true value of the coefficient in the C-R functions and the true dollar value of the effects. The analysis relies on estimates of these parameters, but the true values being estimated are unknown. This type of uncertainty can often be probabilistically quantified. For example, the uncertainty about pollutant coefficients is typically quantified by reported standard errors of the estimates of the coefficients in the C-R functions estimated by epidemiological studies. The Technical Support Document for this analysis quantifies the uncertainty associated with each health and welfare endpoint. Another important source of uncertainty derives from the discrete set of assumptions used to select endpoints and concentration-response functions and to determine inputs to the concentration-response functions. This type of uncertainty can be quantified through the use of sensitivity analyses, but is not easily conveyed in probabilistic terms.

6. Sensitivity Analyses

A portion of the uncertainty associated with benefits analysis involves discrete choices between assumptions. We can not easily assign non-arbitrary probabilities to the alternative assumptions, and instead we use a reasonable range of assumptions. Our analysis uses two sets of assumptions that incorporate the following key assumptions:

(1) the choice of the PM threshold (15 $\mu g/m^3$,or background);
(2) the value placed on reduced mortality associated with PM (the value of a statistical life, or the value of statistical life adjusted to reflect age-distributions of the affected population);
(3) the value placed on reduced incidence of chronic bronchitis;
(4) whether PM is associated with infant mortality;
(5) whether ozone is associated with the mortality of someone at any age;
(6) whether plantings of commodity crop cultivars are sensitive or insensitive to ozone; and
(7) the value placed on visibility benefits (both residential and recreational visibility).

Table VII-16 presents the estimates for the impacts and the associated economic value for each set of assumptions. The results shown in the table demonstrate that selected alternative assumptions drastically changes the total benefits that can be assumed for this rule. Actual benefits are likely to be between the Low and High estimates provided.

Table VII-16. Avoided Incidence and Monetized Benefits Associated with the Tier 2 Rule for a Range of Assumption Sets

	Avoided Incidence (cases/year)		*Monetary Benefits (millions 1997$)*	
Endpoint	*Low*[a]	*High*[c]	*Low*	*High*
PM				
Mortality (long-term exp. - ages 30+)	832	2,416	2,275	14,256
Mortality (long-term exp. - infants)	–	10	–	56
Chronic bronchitis	3,885	3,914	281	1,354
Hosp. Admissions - all respiratory (all ages)	504	836	4.6	7.6
Hosp. Admissions - congestive heart failure	127	138	1.5	1.7
Hosp. Admissions - ischemic heart disease	146	159	2.2	2.4
Acute bronchitis	984	4,072	0.1	0.2
Lower respiratory symptoms (LRS)	19,782	37,437	0.3	0.5
Upper respiratory symptoms (URS)	3,093	3,387	0.1	0.1
Work loss days (WLD)	233,000	415,000	23.8	42.3
Minor restricted activity days (MRAD)	1,856,000	3,370,000	87.7	159.3
Household soiling damage	–	–	60.1	60.1
Ozone				
Mortality (short-term; four U.S. studies)	–	388	–	2,312
Hospital admissions - all respiratory (all ages)	549	736	5.3	7.1
Any of 19 acute symptoms	54,101	71,545	1.3	1.7
Decreased worker productivity	–	–	43.0	60.4
Agricultural crop damage	–	–	-1	301
Visibility	–	–	330	701
Nitrogen Deposition	–	–	200	200
Total (PM + ozone + visibility + N deposition)	–	–	3,315	19,525

[a] The low assumption set assumes effects from PM do not occur below concentrations of 15 $\mu g/m^3$, that all mortality and chronic bronchitis effects occur within the same year of the PM reduction (see section 7.a for a discussion of this uncertainty), utilizes the value of statistical life year lost approach, ozone-related mortality and PM-related infant mortality are not included in the benefits estimate, chronic bronchitis valued with the cost of illness approach, plantings of commodity crop cultivars are assumed to be insensitive to ozone, and does not value residential visibility benefits.

[c] The high assumption set assumes a PM threshold of background, utilizes the value of a statistical life approach, both ozone-related mortality and PM-related mortality are included in the estimation of benefits, chronic bronchitis valued with a willingness-to-pay approach, plantings of commodity crop cultivars are assumed to be sensitive to ozone, and full accounting for recreational and residential visibility benefits.

7. Limitations of the Analysis

Given incomplete information, this national benefits analysis yields approximate results because of the uncertainty associated with any estimate. Potentially important sources of uncertainty exist and many of these are summarized in Table VII-17. These uncertainties can cause the total benefits estimate to be understated or overstated. Where possible, we state the direction of the bias presented by the uncertainty. However, in most cases the effect of the uncertainty on total benefits is unknown (i.e., it could increase or decrease benefits depending on specific conditions). The remainder of this Section provides a discussion of four broad areas of uncertainty.

Table VII-17. Sources of Uncertainty in the Benefit Analysis

1. Uncertainties Associated With Concentration-Response Functions
-The value of the ozone- or PM-coefficient in each C-R function. -Application of a single C-R function to pollutant changes and populations in all locations. -Similarity of future year C-R relationships to current C-R relationships. -Correct functional form of each C-R relationship. (e.g., It is uncertain whether there are thresholds and, if so, what they are.) -Extrapolation of C-R relationships beyond the range of ozone or PM concentrations observed in the study.
2. Uncertainties Associated With Ozone and PM Concentrations
-Estimating future-year baseline and hourly ozone and daily PM concentrations. -Estimating the change in ozone and PM resulting from the control policy.
3. Uncertainties Associated with PM Mortality Risk
-No scientific basis supporting a plausible biological mechanism. -Potential causal agents within the complex mixture of PM responsible for the reported adverse health effects have not been identified. -While there were a great number of studies associated with PM_{10}, there were a limited number of studies that directly measured $PM_{2.5}$. -The extent to which adverse health effects are associated with low level exposures that occur many times in the year versus peak exposures. -Estimated health effects levels associated with $PM_{2.5}$ exposure were small. -Possible confounding in the epidemiological studies of $PM_{2.5}$, effects with other factors (e.g., other air pollutants, weather, indoor/outdoor air, etc.). -The extent to which effects reported in the long-term studies are associated with historically higher levels of PM rather than the levels occurring during the period of study. -Reliability of the limited ambient $PM_{2.5}$ monitoring data in reflecting actual $PM_{2.5}$ exposures.
4. Uncertainties Associated With Possible Lagged Effects
-What portion of the PM-related long-term exposure mortality effects associated with changes in annual PM levels would occur in a single year, and what portion might occur in subsequent years. Ignoring lags may lead to an overestimate of benefits.
5. Uncertainties Associated With Baseline Incidence Rates
-Some baseline incidence rates are not location-specific (e.g., those taken from studies) and may therefore not accurately represent the actual location-specific rates. -Current baseline incidence rates may not well approximate what baseline incidence rates will be in the year 2007. -Projected population and demographics -- used to derive incidences – may not well approximate future-year population and demographics.
6. Uncertainties Associated With Economic Valuation
-Unit dollar values associated with health and welfare endpoints are only estimates of mean WTP and therefore have uncertainty surrounding them. -Mean WTP (in constant dollars) for each type of risk reduction may differ from current estimates due to differences in income or other factors.
7. Uncertainties Associated With Aggregation of Monetized Benefits

-Health and welfare benefits estimates are limited to the available C-R functions. Thus, unquantified benefit categories will cause total benefits to be underestimated.

a. PM Mortality Risk and Health Effects

Table VII-20 summarizes a number of the uncertainties associated with estimating mortality risk associated with particulate matter (PM). Most of these uncertainties can serve to increase or decrease the estimated benefits relative to a hypothetical "true" prediction. Some uncertainties may inflate estimates, while others - such as exclusion of effects categories - can result in understatement. The fundamental concentration-response relationships used to estimate benefits are derived from epidemiological studies of community health. Based on these studies and other available information, the EPA Criteria Document concluded that the observed associations between particulate matter and mortality and other serious health effects were "likely causal." The Criteria Document also noted that, as yet, the scientific information did not provide a basis for determining what biological mechanisms might account for such effects. To the extent that some chance remains that no causal mechanisms are found for some PM components or for the PM mix taken as a whole, the benefit estimates derived from the epidemiological studies would be overstated.

Similarly, the evaluation of the epidemiological evidence included an extensive assessment of a number of potential pollutant and weather confounders or effects modifiers. The Criteria Document concluded that these factors could not fully account for the observed PM/effects associations, but it is possible that some portion of the quantitative relationships are affected by the presence of other pollutants. While multiple pollutant effects may be additive, it is also possible that the PM related effects association may be overstated for some studies, which might inflate the benefits estimates derived from such studies.

In addition, following the recommendation of the Advisory Council on Clean Air Compliance analyses (an SAB advisory committee established to review methodology for the 812 study), the PM mortality benefit estimates have been derived from a single study that likely encompasses both short-and long-term mortality effects (Pope et al. 1995). Similarly, the Agency has used a single study (Schwartz 1993) in its estimates of the benefits of reduced cases of chronic bronchitis. The approach used in both cases assumes that the benefits of the PM reductions will occur within a year of the reductions. Because some fraction of the estimated mortality or chronic bronchitis effects may well be associated with multi-year exposures, the benefits of a given reduction in concentrations in one year will not all be realized in that year. To date, however, the available studies have not developed any estimates of the relative proportion of near term as compared to the potential "lagged" consequences of PM reductions (HEES, 1999).

Some analysts believe, however, that this analysis should provide an estimate that reflects the potential effect of considering such lagged effects in presenting the range of estimated benefits. For example, if one were to assume that realization of the full health benefits from reductions in particulate matter resulting from this rule might take up to 5 years, the estimated monetized benefits for reductions in premature mortality and chronic bronchitis would be reduced by $204 million at the low end of the range of total benefits (see Table VII-18 below).

Table VII-18. PM Health Effects and Benefits (No Lag and Lag of up to Five Years)

Health Effects	Benefits (No Lag) (millions 1997$)	Benefits (Lag of up to 5 years)[cc] (millions 1997$)
Chronic Bronchitis	$281	$259
Mortality	$2,278	$2,096
Total	$2,559	$2,355

As discussed above, SAB has concluded that selection of a value for such a lag at this time would be arbitrary and inclusion of pollutant-related time lags in mortality is premature (HEES, 1999). For this reason, we have not incorporated lags into this analysis. The Agency is committed to working with the SAB and others during the development of the final rule to look at how to address this issue in the benefits range for both the Tier 2 final rule and RIA and in future regulatory analyses.

b. Unquantifiable Benefits

In considering the monetized benefits estimates, the reader should be aware that many limitations for conducting these analyses are mentioned throughout this RIA. One significant limitation of both the health and welfare benefits analyses is the inability to quantify many PM and ozone-induced adverse effects listed in Table VII-6. In general, if it were possible to include the unquantified benefits categories in the total monetized benefits, the benefits estimates presented in this RIA would increase. Specific examples of unquantified benefits explored in more detail below include other human health effects, urban ornamental plants, aesthetic injury to forests, nitrogen in drinking water, and brown clouds.

The benefits of reductions in a number of ozone- and PM-induced health effects have not been quantified due to the paucity of C-R and/or economic valuation data. These effects include:

[cc]This approach assumes that 25 percent of the reductions in health effects reduction are realized in year 1, 25 percent in year 2, 16.67 percent in year 3, 16.67 in year 4 and 16.67 in year 5. This is an illustrative example only and does not represent any known lag structure for these health effects.

reduced pulmonary function, morphological changes, altered host defense mechanisms, cancer, other chronic respiratory diseases, infant mortality, airway responsiveness, increased susceptibility to respiratory infection, pulmonary inflammation, acute inflammation and respiratory cell damage, and premature aging of the lungs.

In addition to the above non-monetized health benefits, there are a number of non-monetized welfare benefits including: reduced adverse effects on vegetation, forests, and other natural ecosystems. The CAA and other statutes, through requirements to protect natural and ecological systems, indicate that these are scarce and highly valued resources. Lack of comprehensive information, insufficient valuation tools, and significant uncertainties therefore result in understated welfare benefits estimates in this RIA. However, a number of expert biologists, ecologists, and economists (Costanza et al., 1997) argue that the benefits of protecting natural resources are enormous and increasing as ecosystems become more stressed and scarce in the future. Additionally, agricultural, forest and ecological scientists (Heck and Cowling, 1997) believe that vegetation appears to be more sensitive to ozone than are humans and consequently, that damage is occurring to vegetation and natural resources at concentrations below the ozone NAAQS. Experts also believe that the effect of ozone on plants is both cumulative and long-term. The specific non-monetized benefits from reductions in ambient ozone concentrations would accrue from: decreased foliar injury; averted growth reduction of trees in natural forests; maintained integrity of forest ecosystems (including habitat for native animal species); and the aesthetics and utility of urban ornamentals (e.g., grass, flowers, shrubs and trees). Other welfare categories for which there is incomplete information to estimate the economic value of reduced adverse effects include: materials damage; and reduced sulfate deposition to aquatic and terrestrial ecosystems.

Other Human Health Effects

Human exposure to PM and ozone is known to cause health effects such as: impaired airway responsiveness, increased susceptibility to respiratory infection, acute inflammation and respiratory cell damage, premature aging of the lungs and chronic respiratory damage. An improvement in ambient PM and ozone air quality is expected to reduce the number of incidences within each effect category that the U.S. population would experience. Although these health effects are known to be PM or ozone-induced, C-R data is not available for quantifying the benefits associated with reducing these effects. The inability to quantify these effects leads to an underestimation of the monetized benefits presented in this analysis.

Urban Ornamentals

Urban ornamentals represent an additional vegetation category likely to experience some degree of effects associated with exposure to ambient ozone levels and likely to impact large economic sectors. In the absence of adequate exposure-response functions and economic damage functions for the potential range of effects relevant to these types of vegetation, no direct quantitative economic benefits analysis has been conducted. It is estimated that more than $20 billion (1990 dollars) are spent annually on landscaping using ornamentals (Abt Associates, 1995), both by private property owners/tenants and by governmental units responsible for public areas, making this a potentially important welfare effects category. However, information and valuation methods are not available to allow for plausible estimates of the percentage of these expenditures that may be related to impacts associated with ozone exposure.

Commercial Forests

Any attempt to estimate economic benefits for commercial forests associated with reductions in ozone arising from implementation of the Tier 2 rule is constrained by a lack of exposure-response functions for the commercially important mature trees. Although exposure-response functions have been developed for seedlings for a number of important tree species, these seedling functions cannot be extrapolated to mature trees based on current knowledge. Recognizing this limitation, a study (de Steiger et al., 1990; Pye et al., 1988) involving expert judgment about the effect of ozone levels on percent growth change has been used to develop estimates of ozone-related economic losses for commercial forest products. Our analysis, however, did not quantify benefits from improved production within commercial forests.

Aesthetic Injury to Forests

Ozone is a regionally dispersed air pollutant that has been shown conclusively to cause discernible injury to forest trees (Fox and Mickler, 1996). One of the welfare benefits expected to accrue as a result of reductions in ambient ozone concentrations in the United States is the economic value the public receives from reduced aesthetic injury to forests. There is sufficient scientific information available that ambient ozone levels cause visible injury to foliage and impair the growth of some sensitive plant species (U.S. EPA, 1996c, p. 5-521). However, present analytic tools and resources preclude EPA from quantifying the benefits of improved forest aesthetics.

Nitrates in Drinking Water

Nitrates in drinking water are currently regulated by a maximum contaminant level (MCL) of 10 mg/L on the basis of the risk to infants of methemoglobinemia, a condition which adversely affects the blood’s oxygen carrying capacity. In an analysis of pre-1991 data, Raucher et al.(1993) found that approximately 2 million people were consuming public drinking water supplies which exceed the MCL. Supplementing these findings, the National Research Council concluded that 42 percent of the public drinking water users in the U.S. (approximately 105 million people) are either not exposed to nitrates or are exposed to concentrations below 1.3 mg/L (National Research Council 1995).

In a recent epidemiological study by the National Cancer Institute, a statistically significant relationship between nitrates in drinking water and incidence of non-Hodgkin’s lymphoma were reported (Ward et al., 1996). Though it is generally acknowledged that traditional water pollution sources such as agricultural runoff are mostly responsible for violations of the MCL, other more diffuse sources of nitrate to drinking water supplies, such as that from atmospheric deposition, may also become an important health concern should the cancer link to nitrates be found valid upon further study.

Other Unquantified Benefits Categories

There are other welfare benefits categories for which there is incomplete information to permit a quantitative assessment for this analysis. For some endpoints, gaps exist in the scientific literature or key analytical components and thus do not support an estimation of incidence. In other cases, there is insufficient economic information to allow estimation of the economic value of adverse effects. Potentially significant, but unquantified welfare benefits categories include: existence and user values related to the protection of Class I areas (e.g., Shenendoah National Park), damage to tree seedlings of more than 10 sensitive species (e.g., black cherry, aspen, ponderosa pine), non-commercial forests, ecosystems, materials damage, and reduced sulfate deposition to aquatic and terrestrial ecosystems. Although scientific and economic data are not available to allow quantification of the effect of ozone in these categories, the expectation is that, if quantified, each of these categories would lead to an increase in the monetized benefits presented in this RIA.

c. Potential Disbenefits

In this discussion of unquantified benefits, a discussion of potential disbenefits must also be mentioned. Several of these disbenefit categories are related to nitrogen deposition, while one category is related to the issue of ultraviolet light. Because EPA is not able to quantify these disbenefit categories, total benefits will be overstated.

Passive Fertilization

Several disbenefit categories are related to nitrogen deposition. Nutrients deposited on crops from atmospheric sources are often referred to as passive fertilization. Nitrogen is a fundamental nutrient for primary production in both managed and un-managed ecosystems. Most productive agricultural systems require external sources of nitrogen in order to satisfy nutrient requirements. Nitrogen uptake by crops varies, but typical requirements for wheat and corn are approximately 150 kg/ha/yr and 300 kg/ha/yr, respectively (NAPAP, 1990). These rates compare to estimated rates of passive nitrogen fertilization in the range of 0 to 5.5 kg/ha/yr (NAPAP, 1991). So, for these crops, deposited nitrogen could account for as much as two to four percent of nitrogen needs. Holding all other factors constant, farmers' use of purchased fertilizers or manure may increase as deposited nitrogen is reduced. EPA has not estimated the potential value of this possible increase in the use of purchased fertilizers, but it is likely that the overall value is very small relative to the value of other health and welfare endpoints presented in this analysis. First, reductions in NO_x emissions affect only a fraction of total nitrogen deposition. Approximately 70 to 80 percent of nitrogen deposition is in the form of nitrates (and thus can be traced to NO_x emissions) while most of the remainder is due to ammonia emissions (Dennis, 1997). The annual average change in nitrogen deposition attributable to the Tier 2 rule is about 11 percent of baseline levels, suggesting a relatively small potential change in passive fertilization. Second, some sources of nitrogen, such as animal manure, are available at no cost or at a much lower cost than purchased nitrogen. In addition, in certain areas nitrogen is currently applied at rates which exceed crop uptake rates, usually due to an overabundance of available nutrients from animal waste. Small reductions in passive fertilization in these areas is not likely to have any consequence to fertilizer application. The combination of these factors suggests that the cost associated with compensating for reductions in passive fertilization is relatively minor.

Information on the effects of changes in passive nitrogen deposition on forests and other terrestrial ecosystems is very limited. The multiplicity of factors affecting forests, including other potential stressors such as ozone, and limiting factors such as moisture and other nutrients, confound assessments of marginal changes in any one stressor or nutrient in forest ecosystems. However, reductions in deposition of nitrogen could have negative effects on forest and vegetation growth in ecosystems where nitrogen is a limiting factor (U.S. EPA, 1993).

On the other hand, there is evidence that forest ecosystems in some areas of the United States are nitrogen saturated (U.S. EPA, 1993). Once saturation is reached, adverse effects of additional nitrogen begin to occur such as soil acidification which can lead to leaching of nutrients needed for plant growth and mobilization of harmful elements such as aluminum. Increased soil acidification is also linked to higher amounts of acidic runoff to streams and lakes and leaching of harmful elements into aquatic ecosystems.

Ultraviolet Light

A reduction of tropospheric ozone is likely to increase the penetration of ultraviolet light, specifically UV-b, to ground level. UV-b is an issue of concern because depletion of the stratospheric ozone layer (i.e., ozone in the upper atmosphere) due to chlorofluorocarbons and other ozone-depleting chemicals is associated with increased skin cancer and cataract rates. Currently, EPA is not able to adequately quantify these effects for the purpose of valuing benefits for this policy.

Other EPA programs exist to address the risks posed by changes in UV-b associated with changes in total column ozone. As presented in the Stratospheric Ozone RIA (U.S. EPA, 1992), stratospheric ozone levels are expected to significantly improve over the next century as the major ozone depleting substances are phased out globally. This expected improvement in stratospheric ozone levels is estimated to reduce the number of non-melanoma skin cancers by millions of cases in the U.S. by 2075.

d. Projected Income Growth

Our analysis does not attempt to adjust benefits estimates to reflect expected growth in real income. Economic theory argues, however, that WTP for most goods (such as environmental protection) will increase if real incomes increase. The degree to which WTP may increase for the specific health and welfare benefits provided by the Tier 2 rule cannot be estimated due to insufficient income elasticity information.

D. Cost

Since the benefits assessment has been performed on the basis of a fully turned over fleet of tier 2 vehicles, consistent costs were developed by using the same basis. Costs to be compared to the monetized value of the benefits were therefore developed for a fleet the size of the year 2010 fleet. For this purpose we used the long term cost once the capital costs have been recovered and the manufacturing learning curve reductions have been realized, since this most closely represents the makeup of a fully turned-over fleet.

This analysis also made adjustments in the costs to account for the fact that there is a time difference between when some of the costs are expended and when the benefits are realized. The vehicle costs are expended when the vehicle is sold, while the fuel related costs and the benefits are distributed over the life of the vehicle.

We resolved this difference by using costs distributed over time such that there is a constant cost per ton of emissions reduction and such that the net present value of these distributed costs corresponds to the net present value of the actual costs. A constant ratio of cost to emission reduction over the life of the vehicle would also reflect itself in the ratio of the net present value of the costs and net present value of the emission reductions. This, of course, is how EPA determined the cost effectiveness estimates for the proposed rule. Thus, the simplest way to develop this distributed cost number is simply to multiply the cost effectiveness ratio (dollars per ton) times the emission reduction estimates for the benefits assessment.

The resulting adjusted costs are somewhat greater than the actual annual cost of the program, reflecting the time value adjustment. Thus, both because of the assumption of a fully turned over fleet and because of the time value adjustment, the costs presented in this section do not represent actual annual costs of the Tier 2/gasoline sulfur program for 2010. Rather, they represent an approximation of the steady-state cost per ton that would likely prevail in 2015 and beyond. The benefit cost ratio for the earlier years of the program would be expected to be lower than that based on these costs, since the fleet-adjusted costs are larger in the early years of the program while the benefits are smaller.

Since the long term costs are not representative of the per vehicle costs in the early phases of the program, we also estimated an adjusted cost based on the near term cost effectiveness value. Using the near term cost effectiveness value of $2134/per ton, the adjusted cost would be $4.3 billion. While no actual in-use fleet could consist entirely of vehicles experiencing this near term cost, this value does present an upper bound on the cost figure.

The resulting adjusted cost values are given in Table VII-19.

Table VII-19. Adjusted Cost for Comparison to Benefits

Cost Basis	*Cost per ton ratio*	*Tons of NOx + NMHC*	*Adjusted Cost (billions of dollars)*
Long term	1748	2,003,761	3.5

Chapter VII References:

Abbey, D.E., F. Petersen, P.K. Mills and W.L. Beeson, 1993. "Long-Term Ambient Concentrations of Total Suspended Particulates, Ozone, and Sulfur Dioxide and Respiratory Symptoms in a Nonsmoking Population." *Archives of Environmental Health*, **48**(1): 33-46.

Abbey, D.E., B.E. Ostro, F. Petersen and R.J. Burchette, 1995. "Chronic Respiratory Symptoms Associated with Estimated Long-Term Ambient Concentrations of Fine Particulates Less Than 2.5 Microns in Aerodynamic Diameter (PM2.5) and Other Air Pollutants." *Journal of Exposure Analysis and Environmental Epidemiology*, **5**(2): 137-159.

Abt Associates Inc., 1992. *The Medical Costs of five Illnesses Related to Exposure to Pollutants.* Prepared for U.S. EPA, Office of Pollution Prevention and Toxics. Washington, DC. See EPA Air Docket A-96-56, Document No. VI-B-09-(n).

Abt Associates Inc., 1995. *Urban Ornamental Plants: Sensitivity to Ozone and Potential Economic Losses*. Prepared for the U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. Research Triangle Park, NC. July.

Abt Associates Inc., 1998. *Air Quality Estimation for the NOx SIP Call RIA.* Prepared for U.S. EPA, Office of Air Quality Planning and Standards, under contract no. 68-D-98-001. Research Triangle Park, NC. September. See EPA Air Docket A-96-56, Document No. VI-B-09-(gggg)

Abt Associates Inc., 1999. *Tier 2 Proposed Rule: Air Quality Estimation, Selected Health and Welfare Benefits Methods, and Benefit Analysis Results*. Prepared for U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC. February.

Butler, R.J., 1983. "Wage and Injury Rate Responses to Shifting Levels of Workers' Compensation." In: *Safety and the Work Force*. Worrall, J.D., Ed. Cornell University, ILR Press, Ithaca, NY.

Chang, J.S., R.A. Brost, I.S.A. Isaksen, S. Madronich, P. Middleton, W.R. Stockwell and C.J. Walcek, 1987. "A Three-Dimensional Eulerian Acid Deposition Model: Physical Concepts and Formulation." *Journal of Geophysical Research*, **92**, 14,681-14,700.

Chang, J.S., P.B. Middleton, W.R. Stockwell, C.J. Walcek, J.E. Pleim, H.H. Lansford, F.S. Binkowski, S. Madronich, N.L. Seaman and D.R. Stauffer, 1990. "The Regional Acid Deposition Model and Engineering Model. NAPAP SOS/T Report 4." In: *Acidic Deposition: State of Science and Technology, Volume 1*. National Acid Precipitation Assessment Program, Washington DC. December.

Chestnut, L.G., 1997. "Draft Memorandum: Methodology for Estimating Values for Changes in Visibility at National Parks." April 15. See EPA Air Docket A-96-56, Document No. VI-B-09-

(ooo).

Chestnut, L.G. and R.D. Rowe, 1990. *Preservation Values for Visibility Protection at the National Parks: Draft Final Report*. Prepared for U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, Economic Analysis Branch. Research Triangle, NC. February, 16.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton and M. van den Belt, 1997. "The Value of the World's Ecosystem Services and Natural Capital." *Nature*, **387**, 253-259.

Cousineau, J., R. Lacroix and A. Girard, 1988. *Occupational Hazard and Wage Compensating Differentials*. University of Montreal Working Paper.

Crocker, T.D. and R.L. Horst, Jr, 1981. "Hours of Work, Labor Productivity, and Environmental Conditions: A Case Study." *The Review of Economics and Statistics*, **63**, 361-368.

Cummings, R., H. Burness and R. Norton, 1981. *Methods Development for Environmental Control Benefits Assessment, Volume V. Measuring Household Soiling Damages from Suspended Air Particulates, A Methodological Inquiry*. Prepared for U.S. Environmental Protection Agency. Washington, DC.

de Steiger, J.E., J.M. Pye and C.S. Love, 1990. " Air Pollution Damage to U.S. Forests." *Journal of Forestry*, **88**(8), 17-22.

Dennis, R. 1997. Personal communication. NOAA Atmospheric Research Lab, Research Triangle Park, NC.

Dennis, R.L., 1997. "Using the Regional Acid Deposition Model to Determine the Nitrogen Deposition Airshed of the Chesapeake Bay Watershed." In: *Atmospheric Deposition to the Great Lakes and Coastal Waters*. Baker, J.E., Ed. Society of Environmental Toxicology and Chemistry, Pensacola, FL. pp. 393-413. See EPA Air Docket A-96-56, Document No. VI-B-09-(ccc).

Dennis, R.L., W.R. Barchet, T.L. Clark and S.K. Seilkop, 1990. "Evaluation of Regional Acid Deposition Models (Part I). NAPAP SOS/T Report 5." In: *Acidic Deposition: State of Science and Technology, Volume 1*. National Acid Precipitation Assessment Program, Washington, DC. September.

Dillingham, A., 1985. "The Influence of Risk Variable Definition on Value of Life Estimates." *Economic Inquiry*, **24**, 277-294.

Dockery, D.W., C.A. Pope, X.P. Xu, J.D. Spengler, J.H. Ware, M.E. Fay, B.G. Ferris and F.E. Speizer, 1993. "An Association Between Air Pollution and Mortality in 6 United-States Cities." *New England Journal of Medicine*, **329**(24), 1753-1759.

Dockery, D.W., F.E. Speizer, D.O. Stram, J.H. Ware, J.D. Spengler and B.G. Ferris, Jr., 1989. "Effects of Inhalable Particles on Respiratory Health of Children." *American Review of Respiratory Disease*, **139**, 587-594.

Douglas, S.G. and R.K. Iwamiya, 1999. *Estimating the Effects of the Tier 2 Motor-Vehicle Standards on Air Quality: Ozone*. Prepared for Abt Associates Inc. Prepared by Systems Applications International, Inc. SYSAPP-99-98/50. January.

E.H. Pechan & Associates, I., 1996. *Regional Particulate Control Strategies Phase II*. Prepared for the U.S. Environmental Protection Agency, Office of Policy, Planning, and Evaluation. Washington, DC. September.

E.H. Pechan, 1999. *Emissions and Air Quality Impacts of Proposed Motor Vehicle Tier 2 and Fuel Sulfur Standards*. Prepared by The Pechan-Avanti Group under EPA Contract No. 68-D9-8052. Prepared for U.S. EPA, Office of Air Quality Planning and Standards, Innovative Strategies and Economics Group. Research Triangle Park, NC. January.

Empire State Electric Energy Research Corporation (ESEERCO), 1994. *New York State Environmental Externalities Cost Study. Report 2: Methodology*. Prepared by RCG/Hagler, Bailly, Inc. November.

Fox, S. and R.A. Mickler, Eds. 1996. Impact of Air Pollutants on Southern Pine Forests. Ecological Studies. Vol. 118. Springer-Verlag, New York.

Garen, J., 1988. "Compensating Wage Differentials and the Endogeneity of Job Riskiness." *The Review of Economics and Statistics*, **70**(1), 9-16.

Gegax, D., S. Gerking and W. Shulze, 1985. *Perceived Risk and the Marginal Value of Safety*. Working paper prepared for the U. S. Environmental Protection Agency.

Gerking, S., M. DeHaan and W. Schulze, 1988. "The Marginal Value of Job Safety: A Contingent Valuation Study." *Journal of Risk and Uncertainty*, **1**, 185-199.

Heck, W.W. and E.B. Cowling, 1997. "The Need for a Long Term Cumulative Secondary Ozone Standard--An Ecological Perspective." *EM*, January, 23-33.

Herzog, H.W., Jr., and A.M. Schlottmann, 1987. *Valuing Risk in the Workplace: Market Price, Willingness to Pay, and the Optimal Provision of Safety*. University of Tennessee Working Paper.

Industrial Economics Incorporated (IEc), 1993. Memorandum to Jim DeMocker, U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Policy Analysis and Review. September 30. See EPA Air Docket A-96-56, Document No. VI-B-09-(lll).

Industrial Economics Incorporated (IEc), 1994. Memorandum to Jim DeMocker, U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Policy Analysis and Review. March 31. See EPA Air Docket A-96-56, Document No. VI-B-09-(nnn).

Ito, K. and G.D. Thurston, 1996. "Daily Pm(10)/Mortality Associations - an Investigation of At-Risk Subpopulations." *Journal of Exposure Analysis and Environmental Epidemiology*, **6**(1), 79-95.

Jones-Lee, M.W. 1989. *The Economics of Safety and Physical Risk.* Basil Blackwell, Oxford.

Kinney, P.L., K. Ito and G.D. Thurston, 1995. "A Sensitivity Analysis of Mortality Pm-10 Associations in Los Angeles." *Inhalation Toxicology*, **7**(1), 59-69.

Kniesner, T.J. and J.D. Leeth, 1991. "Compensating Wage Differentials for Fatal Injury Risk in Australia, Japan, and the United States." *Journal of Risk and Uncertainty*, **4**(1), 75-90.

Korotney, D. 1998. "Tons reduced values for 812 inventory adjustment." Email communication, U.S. Environmental Protection Agency, Office of Mobile Sources, Ann Arbor, MI. November 9.

Krupnick, A.J. and M.L. Cropper, 1992. "The Effect of Information On Health Risk Valuations." *Journal of Risk and Uncertainty*, **5**(1), 29-48.

Krupnick, A.J., W. Harrington and B. Ostro, 1990. "Ambient Ozone and Acute Health Effects - Evidence From Daily Data." *Journal of Environmental Economics and Management*, **18**(1), 1-18.

Krupnick, A.J. and R.J. Kopp, 1988. *The Health and Agricultural Benefits of Reductions in Ambient Ozone in the United States*. Resources for the Future. Washington, DC. Discussion Paper QE88-10. August.

Latimer and Associates, 1996. *Particulate Matter Source - Receptor Relationships Between All Point and Area Sources in the United States and PSD Class/Area Receptors*. Prepared for Bruce Polkowsky, Office of Air Quality Planning and Standards, U.S. EPA. Research Triangle Park, NC. September.

Lee, E.H. and W.E. Hogsett, 1996. *Methodology for Calculating Inputs for Ozone Secondary Standard Benefits Analysis: Part II*. Prepared for U.S. EPA, Office of Air Quality Planning and Standards. March. See EPA Air Docket A-96-56, Document No. VI-B-09-(f).

Leigh, J.P., 1987. "Gender, Firm Size, Industry and Estimates of the Value-of-Life." *Journal of Health Economics*, **6**, 255-273.

Leigh, J.P. and R.N. Folsom, 1984. "Estimates of the Value of Accident Avoidance at the Job Depend on Concavity of the Equalizing Differences Curve." *The Quarterly Review of Economics and Business*, **24**(1), 56-66.

Manuel, E.H., R.L. Horst, K.M. Brennan, W.N. Lanen, M.C. Duff and J.K. Tapiero, 1982. *Benefits Analysis of Alternative Secondary National Ambient Air Quality Standards for Sulfur Dioxide and Total Suspended Particulates, Volumes I-IV.* Prepared for U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. Research Triangle Park.

Marin, A. and G. Psacharopoulos, 1982. "The Reward for Risk in the Labor Market: Evidence from the United Kingdom and a Reconciliation with Other Studies." *Journal of Political Economy*, **90**(4), 827-853.

McClelland, G., W. Schulze, D. Waldman, J. Irwin, D. Schenk, T. Stewart, L. Deck and M. Thayer, 1991. *Valuing Eastern Visibility: A Field Test of the Contingent Valuation Method.* Prepared for U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation. June. See EPA Air Docket A-96-56, Document No. VI-B-09-(ppp).

Miller, T. and J. Guria, 1991. *The Value of Statistical Life in New Zealand*. Report to the New Zealand Ministry of Transport, Land Transport Division.

Moolgavkar, S.H., E.G. Luebeck, T.A. Hall and E.L. Anderson, 1995. "Air Pollution and Daily Mortality in Philadelphia." *Epidemiology*, **6**(5), 476-484.

Moore, M.J. and W.K. Viscusi, 1988. "Doubling the Estimated Value of Life: Results Using New Occupational Fatality Data." *Journal of Policy Analysis and Management*, **7**(3), 476-490.

NAPAP, 1990. *Acidic Deposition: State of Science and Technology, Report 18: Response of Vegetation to Atmospheric Deposition and Air Pollution*. National Acid Precipitation Assessment Program, Office of Director. Washington, DC. NTIS document No. PB92-100544INZ (order at 1-800-553-6847).

NAPAP, 1991. *National Acid Precipitation Assessment Program: 1990 Integrated Assessment Report*. National Acid Precipitation Assessment Program, Office of Director. Washington, DC. NTIS document No. PB92-100346INZ (order at 1-800-553-6847).

National Research Council. Subcommittee on Nitrate and Nitrite in Drinking Water. 1995. *Nitrate and Nitrite in Drinking Water*. National Academy Press, Washington, DC. NTIS document No. PB95-267092INZ (order at 1-800-553-6847).

Olson, C.A., 1981. "An Analysis of Wage Differentials Received by Workers on Dangerous Jobs." *Journal of Human Resources*, **16**, 167-185.

Ostro, B.D., 1987. "Air Pollution and Morbidity Revisited: A Specification Test." *Journal of Environmental Economics and Management*, **14**, 87-98.

Ostro, B.D. and S. Rothschild, 1989. "Air Pollution and Acute Respiratory Morbidity - an Observational Study of Multiple Pollutants." *Environmental Research*, **50**(2), 238-247.

Pope, C.A., D.W. Dockery, J.D. Spengler and M.E. Raizenne, 1991. "Respiratory Health and Pm10 Pollution - a Daily Time Series Analysis." *American Review of Respiratory Disease*, **144**(3), 668-674.

Pope, C.A., M.J. Thun, M.M. Namboodiri, D.W. Dockery, J.S. Evans, F.E. Speizer and C.W. Heath, 1995. "Particulate Air Pollution As a Predictor of Mortality in a Prospective Study of Us Adults." *American Journal of Respiratory and Critical Care Medicine*, **151**(3), 669-674.

Pye, J.M., J.E. de Steiguer and C. Love, 1988. Expert Opinion Survey on the Impact of Air Pollutants on Forests of the USA. Proceedings of Air Pollution and Forest Decline, Interlaken, Switzerland. . October. See EPA Air Docket A-96-56, Document No. VI-B-09-(e).

Raucher, R.S., J.A. Drago, E. Trabka, A. Dixon, A. Patterson, C. Lang, L. Bird and S. Ragland, 1993. *An Evaluation of the Federal Drinking Water Regulatory Program Under the Safe Drinking Water Act as Amended in 1986. Appendix A: Contaminant-Specific Summaries.* Prepared for the American Water Works Association.

Samet, J.M., 1997. *Particulate Air Pollution and Daily Mortality: Analysis of the Effects of Weather and Multiple Air Pollutants. The Phase IB Report of the Particle Epidemiology Evaluation Project*. Health Effects Institute. March. Available through http://www.healtheffects.org/pubform.htm

Schwartz, J., 1993. "Particulate Air Pollution and Chronic Respiratory Disease." *Environmental Research*, **62**, 7-13.

Schwartz, J. and R. Morris, 1995. "Air Pollution and Hospital Admissions For Cardiovascular Disease in Detroit, Michigan." *American Journal of Epidemiology*, **142**(1): 23-35.

Schwartz, J., D.W. Dockery and L.M. Neas, 1996. "Is Daily Mortality Associated Specifically With Fine Particles." *Journal of the Air & Waste Management Association*, **46**(10), 927-939.

Schwartz, J., D.W. Dockery, L.M. Neas, D. Wypij, J.H. Ware, J.D. Spengler, P. Koutrakis, F.E. Speizer and B.G. Ferris, 1994. "Acute Effects of Summer Air Pollution On Respiratory Symptom Reporting in Children." *American Journal of Respiratory and Critical Care Medicine*,

150(5), 1234-1242.

Schwartz, J. and R. Morris, 1995. "Air Pollution and Hospital Admissions For Cardiovascular Disease in Detroit, Michigan." *American Journal of Epidemiology*, **142**(1), 23-35.

Sisler, J.F., 1996. *Spatial and Seasonal Patterns and Long Term Variability of the Composition of the Haze in the United States: An Analysis of Data from the IMPROVE Network.* Colorado State University, Cooperative Institute for Research in the Atmosphere. Fort Collins, CO. July. See EPA Air Docket A-96-56, Document No. VI-B-09-(ee).

Smith, R.S., 1974. "The Feasibility of an 'Injury Tax' Approach to Occupational Safety." *Law and Contemporary Problems*, **38**(4), 730-744.

Smith, R.S., 1976. *The Occupational Safety and Health Act: Its Goals and Achievements*. American Enterprise Institute. Washington, DC.

Smith, V.K., 1983. "The Role of Site and Job Characteristics in Hedonic Wage Models." *Journal of Urban Economics*, **13**, 296-321.

Smith, V.K. and C. Gilbert, 1984. "The Implicit Risks to Life: A Comparative Analysis." *Economics Letters*, **16**, 393-399.

Taylor, C.R., K.H. Reichelderfer and S.R. Johnson. 1993. *Agricultural Sector Models for the United States: Descriptions and Selected Policy Applications.* Iowa State University Press, Ames, IA.

Thurston, G.D., K. Ito, C.G. Hayes, D.V. Bates and M. Lippmann, 1994. "Respiratory Hospital Admissions and Summertime Haze Air Pollution in Toronto, Ontario - Consideration of the Role of Acid Aerosols." *Environmental Research*, **65**(2), 271-290.

Thurston, G.D., K. Ito, P.L. Kinney and M. Lippmann, 1992. "A Multi-Year Study of Air Pollution and Respiratory Hospital Admissions in 3 New-York State Metropolitan Areas - Results For 1988 and 1989 Summers." *Journal of Exposure Analysis and Environmental Epidemiology*, **2**(4), 429-450.

Tolley, G.S. and et al., 1986. *Valuation of Reductions in Human Health Symptoms and Risks*. Prepared for U.S. Environmental Protection Agency. January. See EPA Air Docket A-96-56, Document No. VI-B-09-(mmm).

U.S. Bureau of the Census. 1992. *Statistical Abstract of the United States: 1992*. 112 ed. Washington, DC.

U.S. EPA, 1992. *Regulatory Impact Analysis: Protection of Stratospheric Ozone. Volume II,* *Part I, Appendix F*. U.S. EPA, Stratospheric Protection Program. Washington, DC.

U.S. EPA, 1993. *Air Quality Criteria for Oxides of Nitrogen. Volume II*. U.S. EPA, Office of Research and Development. Washington, DC. EPA-/600/8-91/049bF. August. NTIS document No. PB92-176379INZ (order at 1-800-553-6847).

U.S. EPA, 1994. *Documentation for Oz-One Computer Model (Version 2.0).* Prepared by Mathtech, Inc., under Contract No. 68D30030, WA 1-29. Prepared for U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC. August.

U.S. EPA, 1995. *Acid Deposition Standard Feasibility Study Report to Congress*. U.S. EPA, Office of Air and Radiation, Acid Rain Division. Washington, DC. EPA 430-R-95-001a. May. See EPA Air Docket A-96-56, Document No. VI-B-09-(ll).

U.S. EPA, 1996a. *Proposed Methodology for Predicting PM2.5 from PM10 Values to Assess the Impact of Alternative Forms and Levels of the PM NAAQS*. Prepared by Terence Fitz-Simons, David Mintz and Miki Wayland (U.S. Environmental Protection Agency, Office of Air Quality Planning and Standard, Air Quality Trends Analysis Group). June 26. See EPA Air Docket A-96-56, Document No. VI-B-09-(u).

U.S. EPA, 1996b. *Review of National Ambient Air Quality Standards for Ozone: Assessment of Scientific and Technical Information. OAQPS Staff Paper*. U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC. EPA-452\R-96-007. June. NTIS document No. PB92-190446INZ (order at 1-800-553-6847).

U.S. EPA, 1996c. *Air Quality Criteria for Ozone and Related Photochemical Oxidants*. U.S. EPA, Office of Research and Development. Washington, DC. EPA-/600/P-93/004cF. July. NTIS document No. PC E99/MF E99, PB94-173119INZ (order at 1-800-553-6847).

U.S. EPA, 1996d. *Regulatory Impact Analysis for Proposed Particulate Matter National Ambient Air Quality Standard*. Prepared by: Innovative Strategies and Economics Group, Office of Air Quality Planning and Standards. Research Triangle Park, NC. December. Available at http://www.epa.gov/ttn/oarpg/t1ria.html

U.S. EPA, 1997a. *Deposition of Air Pollutants to the Great Waters: Second Report to Congress*. U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC. EPA-453/R-97-011. June. See EPA Air Docket A-96-56, Document No. VI-B-09-(ff).

U.S. EPA, 1997b. *The Effects of SOx and NOx Emission Reductions on Sulfate and Nitrate Particulate Concentrations*. Thomas Braverman, Air Quality Modeling Group, Office of Air Quality Planning and Standards. Research Triangle Park, NC. May. See EPA Air Docket A-96-56, Document No. VI-B-09-(cc).

U.S. EPA, 1997c. "Methodology Used to Create PM10 and PM2.5 Air Quality Databases for RIA Work." Memorandum from David Mintz, Air Quality Trends Analysis Group, Office of Air Quality Planning and Standards to Allyson Siwik Innovative Strategies and Economics Group, Office of Air Quality Planning and Standards. July 15. See EPA Air Docket A-96-56, Document No. VI-B-09-(kk).

U.S. EPA, 1997d. *National Air Pollutant Emission Trends Procedures Document 1990-1996, Section 4:0: National Criteria Pollutant Estimates, 1985-1996. Draft Document*. Office of Air Quality Planning and Standards. Research Triangle Park, NC. June. See EPA Air Docket A-96-56, Document No. VI-B-09-(hhh).

U.S. EPA, 1997e. *Regulatory Impact Analyses for the Particulate Matter and Ozone National Ambient Air Quality Standards and Proposed Regional Haze Rule*. U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC. July. See EPA Air Docket A-96-56, Document No. VI-B-09-(r).

U.S. EPA, 1997f. "Response to Comments Made by AISI on EPA Methodology for Predicting PM2.5 from PM10." Memorandum to the docket from Terence Fitz-Simons (Office of Air Quality Planning and Standards, Air Quality Trends Analysis Group). February 6. See EPA Air Docket A-96-56, Document No. VI-B-09-(w).

U.S. EPA, 1997g. *The Benefits and Costs of the Clean Air Act: 1970 to 1990*. U.S. EPA, Office of Air and Radiation, Office of Policy, Planning and Evaluation. Washington, DC. October. Available at http://www.epa.gov/airprogm/oar/sect812/copy.html

U.S. EPA, 1997h. *Technical Support Document for the Regulatory Impact Analyses for the Particulate Matter and Ozone National Ambient Air Quality Standards and Proposed Regional Haze Rule*. U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC. July 17.

U.S. EPA, 1998a. *The Regional NOx SIP Call & Reduced Atmospheric Deposition of Nitrogen: Benefits to Selected Estuaries*. September 22.

U.S. EPA, 1998b. *Regulatory Impact Analysis for the NOx SIP Call, FIP, and Section 126 Petitions*. U.S. EPA, Office of Air and Radiation. Washington, DC. EPA-452/R-98-003. December. See EPA Air Docket A-96-56, VI-B-09.

Viscusi, W.K., 1978. "Labor Market Valuations of Life and Limb: Empirical Estimates and Policy Implications." *Public Policy*, **26**(3), 359-386.

Viscusi, W.K. 1979. *Employment Hazards: An Investigation of Market Performance.* Harvard University Press, Cambridge.

Viscusi, W.K., 1981. "Occupational Safety and Health Regulation: Its Impact and Policy Alternatives." In: *Research in Public Policy Analysis and Management*. Crecine, J., Ed. JAI Press, Greenwich, CT. pp. 281-299.

Viscusi, W.K. 1992. *Fatal Tradeoffs: Public and Private Responsibilities for Risk*. Oxford University Press, New York.

Viscusi, W.K., W.A. Magat and J. Huber, 1991. "Pricing Environmental Health Risks - Survey Assessments of Risk - Risk and Risk - Dollar Trade-Offs For Chronic Bronchitis." *Journal of Environmental Economics and Management*, **21**(1), 32-51.

Ward, M.H., S.D. Mark, K.P. Cantor, D.D. Weisenburger, A. Correa-Villasenor and S.H. Zahm, 1996. "Drinking Water Nitrate and the Risk of Non-Hodgkin's Lymphoma." *Epidemiology*, **7**(September), 465-471.

Watson, W. and J. Jaksch, 1982. "Air Pollution: Household Soiling and Consumer Welfare Losses." *Journal of Environmental Economics and Management*, **9**, 248-262.

Woodruff, T.J., J. Grillo and K.C. Schoendorf, 1997. "The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States." *Environmental Health Perspectives*, **105**(6), 608-612.

Woolfolk, M.E., R.K. Iwamiya, C. Van Landingham, T.C. Myers, G.E. Mansell, H.H. Tunggal and S.G. Douglas, 1998. *A Prospective Analysis of Air Quality in the U.S.: Air Quality Modeling*. Systems Applications International, Inc. San Rafael, California. SYSAPP-97/69.