Diesel Emissions Quantifier Health Benefits Methodology

United States
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Agency

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Transportation and Regional Programs Division
Office of Transportation and Air Quality
U.S. Environmental Protection Agency

NOTICE

This technical report does not necessarily represent final EPA decisions or positions. It is intended to present technical analysis of issues using data that are currently available. The purpose in the release of such reports is to facilitate the exchange of technical information and to inform the public of technical developments.

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I. BACKGROUND

The Diesel Emissions Quantifier Benefits Module is a tool for estimating the health and monetary benefits that could result from a decrease in diesel exhaust emissions. The Benefits Module is a new component of EPA's existing web-based Diesel Emissions Quantifier (the Quantifier). The Benefits Module uses the 2002 National Emissions Inventory (NEI) data and the 2002 National Air Toxics Assessment (NATA) model results to estimate the relationship of changes in diesel emissions to changes in primary particulate matter air concentrations for each county in the U.S. The Benefits Module then uses previously-generated outputs from the Environmental Benefits Mapping and Analysis Program (BenMAP) model to estimate the value of changes in the incidence of avoided premature mortality and several excess morbidity endpoints.

The Quantifier, which was released on EPA's website in 2007, allows users to estimate the diesel emission reductions that result from implementing a variety of control strategies for mobile or stationary diesel engines that the user selects. It is designed for users who do not have technical expertise in emissions modeling or air pollution in general, but it does include a substantial amount of technical information for users who do have that expertise. The Quantifier's output includes tabular estimates of particulate matter emission reductions as well as estimates of emission reductions for NO_x, CO, CO_2, and hydrocarbons on both an annual and engine lifetime basis. These tables can be exported in spreadsheet format. It also includes a User's Guide that explains the data and calculations used to estimate the emission reductions. A more detailed description of the Quantifier, and access to the tool itself, can be found at www.epa.gov/cleandiesel/quantifier/.

The Benefits Module runs off of a county-scale "look-up table" within the larger Quantifier tool. The look-up table includes estimates of the monetary benefits per unit of reduction in emissions (tons/year) for each county in the United States. A user does not see this table directly but instead answers a set of questions about the type of engine being controlled, the emission control(s) used, and the location of the emission reductions. Once the Quantifier estimates the emission changes, users can choose to have the Benefits Module estimate the health and monetary impacts of reductions in fine particulate ($PM_{2.5}$) emissions. Those results are calculated from the lookup table and the combined monetary value of avoided mortality and morbidity is presented in tabular format for the counties the user identified. Monetary values are based on avoided incidences of the following health effects:

Premature mortality
Chronic bronchitis
Acute bronchitis
Upper and lower respiratory symptoms
Asthma exacerbation
Nonfatal heart attacks
Hospital admissions
Emergency room visits
Work loss days
Minor restricted-activity days

EPA has developed look-up tables for total diesel PM sources, as well as for on-road diesel sources and non-road diesel sources (diesel pleasure craft, diesel locomotives, diesel commercial marine vessels, and all other non-road diesel sources). The look-up table for total diesel PM sources was developed as part of the Quality Assurance for this tool; the tool uses the on-road and non-road look-up tables and sums the results for the total benefits for projects that include both types of engines. Due to the limitations of BenMAP (the benefits modeling component), the Benefits Module results are only available for the contiguous 48 states. Therefore, it cannot be used to provide benefits for diesel emission reductions strategies in Alaska, Hawaii or the U.S territories.

The purpose of the Quantifier and the Benefits Module is to provide a screening-level estimate of the emissions and health effects, respectively, of specific diesel engine emission reduction options. These options include adding post-combustion control technologies (also known as aftertreatment) to remove or reduce pollutants from the exhaust, replacing older engines with newer, cleaner engines, and/or switching to lower-emitting fuels. Emission reductions for any single project can be distributed in up to five counties. The Quantifier is not considered adequate or appropriate for SIP planning or credit calculation purposes. Users wanting to estimate the air quality or health benefits of a large number of diesel emission reduction programs spread out over many counties should use more complex air quality modeling tools that account for longer range transport of pollution and secondary pollutant formation.

The Quantifier allows the user to enter the size of the fleet affected by the strategy and the year in which the changes will take effect, as well as the location (county) of the engines. For engines used in multiple counties, such as long-haul trucks, the user should specify the county where the majority of the emissions are located. (While the Benefits Module allows the user to allocate emission reductions among multiple counties for the purpose of estimating monetary benefits, currently the Quantifier requires users to pick a single county for the purposes of calculating the effectiveness of each emission reduction strategy.) The Quantifier includes assumptions about the effectiveness at reducing emissions of various emission control technologies; the Benefits Module does not make any changes to those data. The Quantifier also includes scrappage estimates to inform lifetime engine emission reduction estimates. In contrast, the Benefits Module does not include estimates of the health impacts over the lifetime of an engine. Results are instead presented for the single year in which the emission reduction strategy was implemented. More information on the Quantifier can be found in the Users Guide at www.epa.gov/cleandiesel/documents/420b10033.pdf

II. ESTIMATING CHANGES IN $PM_{2.5}$ AIR QUALITY CONCENTRATIONS RESULTING FROM DIESEL EMISSIONS

A. Data Inputs

i. National Emissions Inventory

The NEI is a comprehensive inventory covering criteria pollutants and hazardous air pollutants (HAPs) for the 50 states, Washington DC, Puerto Rico, and the U.S. Virgin Islands. The NEI is assembled and reported every three years by EPA's Emission Inventory and Analysis Group.

Sources in the NEI are described as either stationary or mobile sources. Mobile sources are categorized as either on-road or non-road sources. On-road sources include motorized vehicles that are normally operated on public roadways. This includes passenger cars, motorcycles, minivans, sport-utility vehicles, light-duty trucks, heavy-duty trucks, and buses. Non-road sources include recreational marine and land-based vehicles, farm and construction machinery, industrial, commercial, logging, and lawn and garden equipment, aircraft, airport ground support equipment (GSE), locomotives, and rail maintenance equipment. These sources are powered by diesel, gasoline, compressed natural gas (CNG), and liquefied petroleum gas (LPG) -fueled engines, among others.

In developing the 2002 draft mobile source NEI, EPA provided state, local, and tribal agencies the opportunity to review and provide comment on the preliminary NEI. EPA's National Mobile Inventory Model (NMIM, www.epa.gov/oms/nmim.htm) was used to generate the preliminary non-road estimates for the 2002 NEI. The preliminary on-road estimates were developed by E.H. Pechan & Associates, Inc. using many of the same data and methods being used in NMIM (U.S. EPA, 2004). The on-road emission estimates in the NEI are based on running EPA's MOBILE6 model (http://www.epa.gov/oms/m6.htm) to generate emission factors in grams per mile and then determining total annual tons using annual vehicle miles traveled (VMT). The Highway Performance Monitoring System (HPMS), on which VMT estimates are based, uses sampling frames based on states, metropolitan areas, and non-metropolitan areas within states. EPA then allocates VMT to the county level. The annual VMT used in the preliminary version of the NEI was based on preliminary national 2002 VMT estimates made by the Federal Highway Administration (FHWA). Thirteen states submitted revised VMT data to EPA for incorporation in the final 2002 NEI. Once state, local, and tribal agencies submitted their review of preliminary NEI information to EPA, these data were logged, reviewed, and quality-assured by EPA.

Documentation for the 2002 NEI is provided at www.epa.gov/ttn/chief/net/2002inventory.html

Documentation for the 2002 Mobile NEI is located at ftp://ftp.epa.gov/EmisInventory/2002finalnei/documentation/mobile/2002_mobile_nei_version_3_report_092807.pdf

ii. National-Scale Air Toxics Assessment

The degree to which a reduction in diesel PM emissions results in a change in ambient diesel PM concentrations has been determined based on the results of EPA's 2002 National-Scale Air Toxics Assessment (NATA) (www.epa.gov/ttn/atw/natamain/). NATA, which is often referred to as a "screening model" due to limitations in the underlying data and methodology, predicts

ambient concentrations of diesel PM at the census tract level. NATA does this by performing dispersion modeling of diesel PM emissions taken from the 2002 National Emissions Inventory (NEI). The 2002 NATA includes 292 air pollutants, including all 187 hazardous air pollutants and diesel PM. The assessment includes four steps:

1. Compiling a national emissions inventory of air toxics emissions from outdoor sources.
2. Estimating ambient concentrations of air toxics across the United States.
3. Estimating outdoor population exposures across the United States.
4. Characterizing potential public health risk due to inhalation of air toxics including both cancer and non-cancer effects.

The first step, developing emissions inventories for the NEI, is described above. Since the NEI only provides county-scale emissions for mobile sources and area-wide stationary sources, the emissions must be apportioned to the census tract level for NATA modeling purposes. For diesel emission sources, the emissions are apportioned based on source category, for example:

On-road diesel emissions use roadway miles (urban primary roads, rural primary roads, urban secondary roads, rural secondary roads) for all roads except local roadways. This is because information on local roadway miles is not generally available so population was instead used as a surrogate for roadway miles

Locomotive diesel emissions use railroad miles

Commercial marine diesel emissions use port locations and underway miles, i.e. miles traveled under engine power

Construction diesel emissions use population change (according to the census, i.e. 1990 - 2000)

Diesel pleasure craft emissions use miles of water coastline

Types of Particulate Matter

The Quantifier estimates changes in diesel particulate matter, or diesel PM. This is all diesel particles, regardless of size. Likewise, NATA models diesel PM.

Health effects and monetary benefits, however, are related to exposures to fine particulate matter, or $PM_{2.5}$. $PM_{2.5}$ includes diesel particles as well as other types of small particles (the "2.5" means the particles are smaller than 2.5 microns).

Most diesel particulate matter is $PM_{2.5}$; some, however, is larger than 2.5 microns.

The Benefits Module uses the same estimate as EPA's MOBILE 6.2 model (the latest mobile source inventory development tool) and assumes that 96% of diesel particulate matter is $PM_{2.5}$. Therefore, the Benefits Module benefits are calculated on 96% of the total diesel PM emission reductions estimated by the Quantifier.

A complete list of surrogates is available on Tables C-7 and C-8 of "User's Guide for the Emissions Modeling System for Hazardous Air Pollutants (EMS-HAP) Version 3.0" (www.epa.gov/scram001/userg/other/emshapv3ug.pdf).

For step 2, a computer simulation model called the Assessment System for Population Exposure Nationwide (ASPEN; www.epa.gov/ttn/atw/nata1999/aspen99.html) is used to estimate toxic air pollutant concentrations. This model is based on EPA's Industrial Source Complex Long Term model (ISCLT) which simulates the behavior of the pollutants after they are emitted into the atmosphere. ASPEN uses estimates of toxic air pollutant emissions and meteorological data from National Weather Service Stations to estimate air toxics concentrations nationwide by census

tract. The ASPEN model takes into account important determinants of pollutant concentrations, such as:

Rate of release
Location of release
The height at which the pollutants are released
Wind speeds and directions at the meteorological stations closest to the release
Breakdown of the pollutants in the atmosphere after release (i.e., reactive decay)
Settling of pollutants out of the atmosphere (i.e., deposition)
Transformation of one pollutant into another (i.e., secondary formation)

ASPEN estimates toxic air pollutant concentrations for every census tract in the continental United States, Puerto Rico and the Virgin Islands. Census tracts are land areas defined by the U.S. Bureau of the Census and typically contain about 4,000 residents each. Census tracts in cities are usually smaller than 2 square miles in size but are much larger in rural areas (U.S. Bureau of Census, 2000) The ASPEN user's guide is available at www.epa.gov/scram001/userg/other/aspenug.pdf.

For emissions apportioned from the county-level to the census tract, such as on-road and non-road diesel sources, the emission locations within each census tract are treated as pseudo-point sources at locations in a radial grid around the census tract centroid. Pseudo-point sources are assumed to be vented point sources with an effective stack height of 5 meters and for which no plume rise calculations are made. ASPEN modeling was carried out to 40 km. Annual average emissions rates were used – no diurnal patterns were assumed. Because of this approximation in emissions source location, ASPEN was deemed sufficiently accurate for purposes of modeling on-road and non-road diesel sources. The 2002 NATA uses a more sophisticated dispersion model, AERMOD (see www.epa.gov/scram001/dispersion_prefrec.htm#aermod), to model large stationary sources where more detailed emissions information is available, but the AERMOD analysis does not apply to the module described here.

For some pollutants, the concentration estimates include a "background" concentration which is based on monitored values. Background concentrations are the contributions to outdoor air toxics concentrations resulting from natural sources, persistence in the environment of past years' emissions, and long-range transport from sources that are more than 50 kilometers away. In other words, background concentrations are levels of pollutants in the atmosphere that would be present if there had been no anthropogenic emissions in the area being modeled. (www.epa.gov/ttn/atw/nata1999/background.html). For diesel PM, NATA does not use monitored air quality concentrations to estimate background concentrations. Instead, it uses a modeling-based approach that provides a rough approximation of air concentrations resulting from transport from sources located between 50 km and 300 km from the receptors. These estimates are included in the source category concentration estimates instead of being treated as a separate source category in the 2002 NATA.

The results of NATA step 2, predicted ambient concentration for diesel PM at the census tract level, are used in the Benefits Module. The results from steps 3 and 4 of the NATA analysis (estimating population exposures and characterizing public health risk) have not been used to support the Benefits Module and thus are not further described here. Instead, the Quantifier uses the Environmental Benefits Mapping and Analysis Program (BenMAP) to estimate the health

impacts. Further information on NATA's use of the Hazardous Air Pollutant Exposure Model 5 (HAPEM5) can be found at www.epa.gov/ttn/atw/nata1999/ted/teddraft.html. For further information summarizing the 2002 NATA and past results, see www.epa.gov/ttn/atw/nata1999/natafinalfact.html.

NATA results are publicly available on EPA's website at www.epa.gov/ttn/atw/natamain/. Results can be found for the entire United States, at the county or census tract level, and by source type or pollutant. These results are best used for comparing counties or census tracts to one another, and do not define "hotspots" or areas of significantly higher concentrations within a single census tract, or answer epidemiological questions such as whether proximity to sources causes increased adverse health effects or higher risks.

The NATA methodology has undergone Science Advisory Board (SAB) peer review. Details of the review, including slide presentations and user documentation for each step of the NATA approach, are available at www.epa.gov/ttn/atw/sab/sabrev.html.

B. Analysis and Calculations

The census tract level NATA-predicted ambient concentrations of diesel PM were used to create the lookup tables that are the basis for the Benefits Module. These predicted ambient concentrations of diesel PM are used in conjunction with standard $PM_{2.5}$ concentration-response functions used in the BenMAP benefits modeling tool.

In order to create county-level ambient concentrations, the census tract level ambient concentration NATA results (c) have been population-weighted to county values:

For a county *i* and tract *a*:

$$c_i = \frac{\sum (c_{i,a} * Population_{i,a})}{Population_i}$$

This analysis was performed for total ambient diesel PM, as well as for on-road diesel PM and non-road diesel PM. County-level, versus census-tract level, concentrations have been used for the Benefits Module because (a) county-level results are a better match for the standard $PM_{2.5}$ concentration-response functions used in the health benefits analysis and (b) mobile source emissions, taken from NEI, are estimated at the county-level and the use of census tract level results would introduce additional uncertainty.

As an additional note, long range dispersion of diesel PM may contribute to an increase in diesel PM concentrations in one county due to emissions from a neighboring county. In general, this effect is likely to be insignificant because the large majority of diesel impacts occur in close proximity to the source, but it is a potential concern for low-emitting counties in close proximity to or downwind of one or more high-emitting counties. In areas where long-range transport is more important, the uncertainty resulting from the approach used here may be more significant and remains inadequately accounted for in this methodology. An inspection of the resulting

county ratios, described in the Quality Assurance section below, reveals that there were only a few counties that had very low emissions and large ambient PM diesel concentrations, none of which were clearly an inaccurate result. Nonetheless, given the uncertainty in the results for these counties, benefits have been calculated, but a flag has been added in the Benefits Module to indicate where benefit per ton estimates for low-emitting counties may be underestimates, and also where benefit per ton estimates for high-emitting counties may be overestimates (due to transport of emissions into surrounding counties). The method used to identify and flag counties, based on a ratio of predicted ambient concentration to emissions density in each county, is described further in Section IV.C below.

III. ESTIMATING THE HUMAN HEALTH BENEFITS OF CHANGES IN $PM_{2.5}$ AIR QUALITY

A. Overview

Having first estimated change in $PM_{2.5}$ ambient concentrations resulting from a change in diesel $PM_{2.5}$ emissions, the Benefits Module then estimates the per-ton benefit of reducing ambient diesel $PM_{2.5}$. To perform this benefits analysis, the Benefits Module uses the "damage function" approach, which is a peer-reviewed technique for estimating the human health impacts associated with exposure to ambient pollutants (Levy et al., 1999). As a result, the Benefits Module calculates the benefit-per-ton of $PM_{2.5}$ emission reduction in a manner generally consistent with the methods found in the recently published Regulatory Impact Analysis (RIA) for the Ozone NAAQS (U.S. EPA, 2008a).

Estimating $PM_{2.5}$ benefit-per-ton entails three basic steps:

1. Estimating the change in $PM_{2.5}$ air quality for the geographic area of interest
2. Loading the estimated air quality changes into the Environmental Benefits Mapping and Analysis Program (BenMAP) and estimating the resulting change in the incidence of health outcomes and monetizing the benefits of those outcomes (Abt Associates Inc., 2005a)
3. Dividing the total monetized benefit by the total estimated emission reduction

The discussion in the preceding section described how the estimates of change in ambient diesel PM air quality concentrations were derived for each county, which constitutes the first step above. The following sections detail how we estimated health benefits of $PM_{2.5}$ exposure and performed the final benefit-per-ton calculations.

B. Data Inputs and Health Endpoints

The Benefits Module uses the BenMAP model to estimate the health endpoints (the health effects that are caused, exacerbated, or otherwise affected by exposure to $PM_{2.5}$ such as premature mortality or asthma attacks) resulting from a unit change in diesel emissions in each county. Table 1 below summarizes the health endpoints quantified and the health impact functions applied for this analysis.

Modeling was done for each of three air quality modeling scenarios—on-road, non-road and total diesel PM. The model compared baseline air quality for each scenario (reflecting total county level ambient $PM_{2.5}$ from that particular source type alone) and a control air quality scenario (reflecting a zero-out of ambient $PM_{2.5}$) The modeling predicted relatively small incremental changes in $PM_{2.5}$ in each county. Because most of the health impact functions (equations that explain the relationship between exposure and changes in health endpoints) used for our analysis are log-linear (and thus produce different estimates of health impacts depending on the baseline level of air quality change), the benefits are somewhat sensitive to the baseline levels of air quality. For this reason, we modified the air quality inputs slightly by adding 10 $\mu g/m^3$ to the baseline and control air quality files—ensuring that the benefits were calculated higher on the curve. Because it is not possible to know ex-ante what the baseline air quality levels will be in the counties in which users apply the benefit-per-ton estimates, this seemed like a reasonable adjustment.

In general, the benefits assessment used techniques, health impact functions and valuation functions that are consistent with the $PM_{2.5}$ health impacts assessments supporting the $PM_{2.5}$ and the Ozone National Ambient Air Quality Standards (U.S. EPA, 2006; U.S. EPA 2008a), with two major exceptions. First, in contrast to those analyses, this assessment applies non-threshold adjusted $PM_{2.5}$ health impact functions. Some researchers have hypothesized the presence of a threshold relationship between $PM_{2.5}$ exposure and the risk of adverse health effects, including premature mortality. For this reason, EPA has traditionally applied an assumed 10 $\mu g/m^3$ cutpoint to the long-term mortality and short-term morbidity concentration-response functions. We determined that such a threshold would be inappropriate for this analysis because we do not know, ex ante, which areas would receive air quality improvements above or below this hypothesized threshold. Further, we did not believe it appropriate to assign zero benefits to counties where ambient PM levels were below a threshold level of 10 $\mu g/m^3$.

The second major divergence from the two RIAs noted above is that we estimated current year population exposure (2008), rather than a projected exposure. We anticipated that most users would wish to estimate the near-term benefits of diesel control strategies, which called for using current year population to generate exposure estimates in BenMAP. Users interested in additional details regarding the health impact assessment may refer to the most recent $PM_{2.5}$ and ozone RIAs (U.S. EPA 2006; U.S. EPA 2008a).

Table 1: Summary of health endpoints and health impact functions

Endpoint	Pollutant	Study and Functional Form	Study Population
Premature Mortality			
Premature mortality — cohort study, all-cause	$PM_{2.5}$ (annual)	Laden et al. (2006), log-linear	>25 years
Premature mortality — all-cause	$PM_{2.5}$ (annual)	Woodruff et al. (1997), logistic	Infant (<1 year)
Chronic Illness			
Chronic bronchitis	$PM_{2.5}$ (annual)	Abbey et al. (1995), logistic	>26 years
Nonfatal heart attacks	$PM_{2.5}$ (daily)	Peters et al. (2001), logistic	Adults
Hospital Admissions			
Respiratory	$PM_{2.5}$ (daily)	Pooled estimate: Moolgavkar (2003)—ICD 490-496 (COPD), log-linear Ito (2003)—ICD 490-496 (COPD), log-linear	>64 years
	$PM_{2.5}$ (daily)	Moolgavkar (2000)—ICD 490-496 (COPD), log-linear	20–64 years
	$PM_{2.5}$ (daily)	Ito (2003)—ICD 480-486 (pneumonia), log-linear	>64 years
	$PM_{2.5}$ (daily)	Sheppard (2003)—ICD 493 (asthma), log-linear	<65 years
Cardiovascular	$PM_{2.5}$ (daily)	Pooled estimate: Moolgavkar (2003)—ICD 390-429 (all cardiovascular), log-linear Ito (2003)—ICD 410-414, 427-428 (ischemic heart disease, dysrhythmia, heart failure), log-linear	>64 years
	$PM_{2.5}$ (daily)	Moolgavkar (2000)—ICD 390-429 (all cardiovascular), log-linear	20–64 years
Asthma-related ER visits	$PM_{2.5}$	Norris et al. (1999), log-linear	0–18 years
Other Health Endpoints			
Acute bronchitis	$PM_{2.5}$	Dockery et al. (1996), logistic	8–12 years
Upper respiratory symptoms	PM_{10}	Pope et al. (1991),	Asthmatics, 9–11 years
Lower respiratory symptoms	$PM_{2.5}$	Schwartz and Neas (2000)	7–14 years
Asthma exacerbations	$PM_{2.5}$	Pooled estimate: Ostro et al. (2001) (cough, wheeze and shortness of breath) Vedal et al. (1998) (cough)	6–18 years[a]
Work loss days	$PM_{2.5}$	Ostro (1987), log-linear	18–65 years
Minor restricted activity days (MRADs)	$PM_{2.5}$	Ostro and Rothschild (1989), log-linear	18–65 years

[a] The original study populations were 8 to 13 for the Ostro et al. (2001) study and 6 to 13 for the Vedal et al. (1998) study. Based on advice from the SAB-HES, we extended the applied population to 6 to 18, reflecting the common biological basis for the effect in children in the broader age group.

The final stage of the benefits analysis is to estimate the monetary value of the health impacts for each county and each of the three scenarios. As in the health incidence stage of the benefits analysis, here we follow techniques that are generally consistent with previous EPA RIA benefits analyses. As in those analyses, mortality benefits are estimated using the EPA standard Value of Statistical Life of $5.5 million (1990 dollars income levels, 1999$). We also apply an EPA Science Advisory Board-recommended 20-year distributed lag between exposure and premature mortality.[1] When calculating monetized benefits, it is necessary to discount over this time period. Hence, we discount the mortality benefits at 3% and then sum the monetary value of each independent endpoint. We estimated valuation for a cost year of 2006 and adjusted the Willingness to Pay (WTP) valuation functions to reflect 2008 projected income levels. Users interested in the complete technical details of the valuation stage may refer to the most recent $PM_{2.5}$ and ozone RIAs (U.S. EPA 2006; U.S. EPA 2008a).

i. Annual versus Annualized Monetized Benefits

The steps above produce an annual estimate of the benefits of reducing an incremental ton of $PM_{2.5}$ from various emission sources for the year 2008. However, we expect that diesel retrofits will provide a stream of benefits over a number of years. Moreover, the costs of these controls are frequently expressed in annualized terms that take into account the expected "lifetime" of the investment. Annualizing costs is the process of combining capital and operating-and-maintenance costs and then distributing these costs on an annual basis over the life of the equipment.

Thus, the benefits and costs are expressed in somewhat different temporal scales. Ideally, the benefits should also be annualized as well. However, this process would require a year-to-year estimate of the change in emissions and air quality over the life of each piece of equipment. For this same time period we would calculate year-to-year benefits, and this stream of future benefits would then be discounted back to the original year in which the emission control was installed. Moreover, we would account for year-to-year changes in population growth and distribution. We would also project changes in income growth to account for the increasing willingness to pay to reduce mortality risk. These were not practical analyses for this project.

Instead, we have made the assumption that the annual benefits are a fair surrogate for the annualized benefits. On one hand, we have neither modeled future population growth and distribution, nor accounted for future income growth; these are factors that should increase benefits over time. On the other hand, this stream of benefits would be discounted, which would reduce the annualized benefits. In our judgment, these countervailing factors more or less balance out such that the annual benefits are comparable to the annualized benefits. Each of the tables and maps in this document treat annual benefits as annualized benefits.

ii. Calculating the $PM_{2.5}$- Benefit per-ton Estimate

The final step is to simply divide the county level benefit estimate by the total change in emissions—resulting in a benefit-per-ton estimate. The benefit-per-ton estimate can also be represented as follows:

[1] This lag reflects the hypothesis that some reductions in premature mortality from exposure to ambient $PM_{2.5}$ will occur over short periods of time in individuals with compromised health status, but other effects are likely to occur among individuals who, at baseline, have reasonably good health that will deteriorate because of continued exposure.

$$BPT_{ii} = \left(\frac{\Delta w_i}{\Delta e_i} \right)$$

where

BPT_{ii} = average health benefits (in 2002 dollars) in county *i* per ton of reduced diesel PM emissions in county *i*,

Δe_i = total reduction in diesel PM emissions (in tons) in county *i*,

Δw_i = health benefits (in 2006 dollars) in county *i* as a result of Δc_i.

For this Benefits Module, no factor was used to convert the ambient diesel PM concentrations (Δc_i) in each county to ambient $PM_{2.5}$ concentration, prior to calculating health benefits (Δw_i). Similarly, BPT_{ii} were calculated by dividing by county diesel PM emissions (Δe_i), and not just the $PM_{2.5}$ component. Diesel PM consists primarily of $PM_{2.5}$, generally 96% by mass (U.S. EPA MOBILE 6.2). Without additional information about how the percentage of $PM_{2.5}$ to total diesel PM may vary between sources and locations within a county, and because Δw_i generally scales linearly with Δc_i, any factor that describes the proportion of diesel PM that is $PM_{2.5}$ would be multiplied in both the numerator (Δw_i*factor*) and denominator (Δe_i*factor*) of the BPT_{ii} calculation, and would cancel out. Thus, for purposes of deriving BPT_{ii}, the relative proportion of diesel PM that is $PM_{2.5}$ is unimportant.

When applying the BPT_{ii} to determine the health benefits for specific diesel exhaust reductions, it is important to remember that the health functions to derive Δw_i are specific to $PM_{2.5}$. Thus, the derived BPT_{ii} most accurately describes health benefits per ton of $PM_{2.5}$ reduced, and not total diesel PM. The emissions changes predicted by the Quantifier are presented in the Quantifier as changes in particulate matter (PM). The Benefits Module converts the Quantifier diesel PM into changes in $PM_{2.5}$ using the 96% conversion factor identified above before the health benefits can be calculated.

IV. ESTIMATING ANNUALIZED COSTS

The Quantifier estimates the cost-effectiveness of each project over the average remaining lifetime of the engine. These values are not easily comparable to the annual benefits presented in the Benefits Module. Therefore, the Benefits Module also estimates the annualized cost of each project.

The annualized cost is based on project cost data the user inputs into the Quantifier. Users can enter two different costs into the Quantifier: the total project cost and the capital costs. The total project costs refer to the entire cost of a retrofit project (for example, the amount of grant funding received to do the project) whereas the capital costs refer to the portion of the total costs that go towards purchasing and installing the retrofit equipment. Capital costs do not include any on-going maintenance costs. To calculate the annualized cost, the Benefits Module uses the value the user enters for the capital cost of the project.

The formula for calculating annualized costs in essence "spreads out" the initial investment costs of the project over the remaining lifetime of the engine being retrofitted. The remaining lifetime is calculated from the existing scrappage tables in the Quantifier. These are the same data used to calculate the cost-effectiveness estimates in the existing version of the Quantifier. This process is used because although the costs are usually paid upfront, the benefits are spread out each year over the remaining lifetime of the engine. By annualizing costs and benefits, the values can be more easily compared.

The formula used for annualizing costs is:

$$AC = (P * r)/(1-(1+r)^{-n})$$

Where:

AC = Annualized Cost
P = Principal (or upfront capital cost)
r = Discount rate
n = Years (remaining life of the engine)

In this case we use a discount rate of 3%. This rate is recommended by EPA draft guidance ftp://ftp.epa.gov/EmisInventory/2002finalnei/documentation/mobile/2002_mobile_nei_version_3_report_092807.pdf regarding discounting of future costs and benefits in situations where all costs and benefits occur as changes in consumption flows rather than changes in capital stocks, i.e., capital displacement effects are negligible. As of the date of publication, current estimates of the consumption rate of interest, based on recent returns to Government-backed securities, are close to 3%.

Since the remaining lifetime of engines in a given retrofit project may vary, the annualized costs must be calculated separately for each type and model year of engine in any given project. These values are then summed to calculate the total annualized cost for each project.

V. UNCERTAINTIES, LIMITATIONS, AND QUALITY ASSURANCE

The Benefits Module represents a new way to bring together existing tools and databases to provide information to state and local agencies, the public and other parties as they seek to implement diesel reduction strategies. These existing data and tools have at various times been subjected to comment and peer review and reflect the recommendations of many experts in multiple disciplines. Nonetheless, the approach and data used by the Benefits Module contain multiple uncertainties and limitations that can limit the application of this tool. These uncertainties and limitations are discussed in more detail below.

A. Input Data

The emissions inventory for diesel PM from the 2002 NEI includes uncertainties associated with the emissions factors, particularly those built into NMIM and the activity information either included by default by EPA or provided by state and local agencies. It also includes the

methodology used to apportion diesel PM emissions to the census tract level using surrogates in NATA

The NATA modeling approach has a series of limitations as well. First, the results are considered most reliable at comparing geographic areas, not analyzing specific locations. The assessment focused on variation between geographic areas such as census tracts, counties and states. It cannot be used to identify "hot spots" where the air concentration, exposure and/or risk might be significantly higher within a census tract or county. In addition, this kind of modeling assessment cannot address the kinds of questions an epidemiology study might, such as the relationship between asthma and proximity of residences to point sources, roadways and other sources of air toxics emissions.

Second, the results do not include impacts from sources in neighboring countries (i.e., Canada or Mexico). Since the assessment did not include the emissions of sources in Canada and Mexico, the results for states that border either of these countries would not reflect these potentially significant sources of transported emissions.

Third, the assessment does not fully reflect variations in background ambient air concentrations. This includes both emissions from natural sources unrelated to anthropogenic emissions as well as transport of emissions from other counties. The assessment uses background ambient air concentrations that are average values over broad geographic regions. Much more research is needed before an accurate estimate of background concentrations at the level of census tracts, or even at the higher geographic scales (i.e. counties or states), can be made. Since background levels are significant contributors to the overall exposure in this assessment, the lack of detailed information on variations in background exposures probably causes the amount of variation in total exposure and risk between census tracts to be smaller than would otherwise be the case.

It is also important to keep in mind that NATA might systematically underestimate ambient air concentration for some compounds. A comparison of the 1996 and 1999 NATA results with ambient monitoring found good agreement for benzene (which primarily comes from gasoline engines), but underestimates for several other species, especially metals. Diesel PM monitoring is not generally available, so no comparison between NATA-predicted diesel PM concentrations and ambient monitoring has been made. There are several possible reasons for the underestimation of pollutant concentrations by NATA:

> *The National Emissions Inventory (NEI) may be missing specific emissions sources* (for many of the sources in the NEI some of the emissions parameters are defaulted or missing). Where data were missing or of poor quality, NATA uses default, or simplified assumptions.
>
> *If the emission rates are underestimated in many locations*. EPA believes the ASPEN model itself is contributing in only a minor way to the underestimation. This is mainly due to output from the predecessor of the ASPEN model comparing favorably to monitoring data in cases where the emissions and meteorology were accurately characterized and the monitors took more frequent readings.

If there are problems in monitor siting. Sites are normally situated to find peak pollutant concentrations, which imply that errors in the characterization of sources would tend to make the model underestimate the monitor values.

Uncertainty in the accuracy of the monitor averages, which, in turn, have their own sources of uncertainty. The results suggest that the model estimates are uncertain on a local scale (i.e., at the census tract level). EPA believes that the model estimates are more reliably interpreted as being a value likely to be found within 30 km of the census tract location.

With respect to diesel PM specifically, the ASPEN modeling used in NATA does not take into account secondary formation of $PM_{2.5}$ (i.e. atmospheric transformation into $PM_{2.5}$ of other pollutants present in diesel exhaust such as oxides of sulfur and nitrogen along with volatile organic carbons). Many of the emission controls included in the Quantifier will reduce mobile source NOx, which is an important precursor to the formation of ambient $PM_{2.5}$. By not modeling the influence of NOx reductions on $PM_{2.5}$ formation, our benefit-per-ton estimates may be biased downward. While we are aware of no published estimates quantifying this bias, it is possible to generate a bounding estimate by using previously published $PM_{2.5}$ benefit-per-ton estimates.

EPA published a series of $PM_{2.5}$ benefit-per-ton estimates in 2008 that relate changes in PM precursors to monetary benefits (U.S. EPA 2008b). These estimates vary by precursor reduced and source type affected. These estimates indicate that the value in 2015 of reducing one ton of directly emitted carbonaceous particles from a mobile source is about $380,000 (Laden et al. mortality estimate, 3% discount rate). Conversely, the value of reducing one ton of NOx emissions from mobile sources is about $10,000 (Laden et al. mortality estimate, 3% discount rate). The significant difference in valuation estimates reflects the differing potential for these precursors to form $PM_{2.5}$ in the atmosphere. This difference suggests, in turn, that not modeling NOx emissions may bias our estimates of $PM_{2.5}$ formation by only a small degree.[2]

In summary, the uncertainties and limitations associated with several key components of the analysis propagate through the analysis. The estimated health effects are calculated based on an array of "upstream" data and assumptions, the most significant of which relate to the change in ambient PM concentrations resulting from changes in emissions. We note that diesel PM is predominately but not exclusively $PM_{2.5}$, and $PM_{2.5}$ includes but is not limited to diesel particles. Based on these predicted air quality changes, we draw upon the vast body of $PM_{2.5}$ health effects literature to apply well-established benefit estimation techniques.

There are several key limitations and uncertainties associated with the benefit-per-ton estimates as well:

Estimating benefits at the local scale carries special uncertainties. This benefits analysis combines county-level air quality data with a substantial amount of national- and regional-level baseline incidence data to estimate the change in $PM_{2.5}$-related health outcomes. With the exception of baseline incidence rates for mortality, the health inputs

[2] In addition, as noted further in the document, we do not estimate ozone-related benefits or other benefits categories such as visibility. As such, the benefit-per-ton estimates likely understate total benefits.

to the analysis are defined at a much broader geographic scale than the air quality data. Moreover, the study we use to estimate $PM_{2.5}$ mortality benefits (Laden et al., 2006) is based upon population exposure data in six cities across the U.S. To the extent that populations in that study and the populations exposed to diesel PM are different, we may under- or over-state total benefits. For these reasons, this analysis is unlikely to have completely characterized the spatial variability in benefits.

The benefit-per-ton metrics contain each of the uncertainties inherent in a $PM_{2.5}$ benefits analysis. As discussed in the $PM_{2.5}$ NAAQS RIA (Table 5.5; U.S. EPA 2006), there are a variety of uncertainties associated with calculating PM benefits; these uncertainties are passed through to the benefit-per-ton estimates included in the Benefits Module. To some extent these uncertainties are exacerbated when applied at smaller scales.

These estimates omit certain benefits categories. Reductions in $PM_{2.5}$ precursors may provide visibility benefits, which are not expressed in the benefit-per-ton metrics. Certain unquantified benefit categories, described fully in the $PM_{2.5}$ NAAQS RIA (U.S. EPA 2006), are also omitted. These categories include ecological benefits, changes in pulmonary function, low birth weight, and non-asthma respiratory ER visits.

The full description of the limitations and uncertainties of the BenMAP modeling tool are available in the BenMAP User's Guide Technical Appendices, Appendix I: Uncertainty and Pooling (pg 254-263) (Abt, 2005a) and online at www.epa.gov/air/benmap/models/BenMAPappendicesSept08.pdf

B. Appropriate Use of This Application

For all of these factors, the uncertainty may lead to either a positive or negative bias in the results. The potential magnitude of the uncertainty in results is difficult to quantify. Past experience with emissions inventories would suggest that the magnitude of emissions, a product of emissions factors and activity, would be one of the largest uncertainties associated with the use of these data. However, basing our estimate on the ratio of the ambient concentration to total emissions, as is done for the Benefits Module, tends to minimize the importance of uncertainties in the emissions. For example, doubling emissions in a specific area would tend to double ambient concentrations, but keep the ratio relatively static, and thus the absolute uncertainty in emissions is not as significant a concern as other uncertainties in this analysis. Conversely, to the extent that these emissions transport to other areas, the uncertainty may be larger.

One of the main factors determining magnitude of health benefits associated with a given emissions reduction is the proximity of the emissions to people. Thus, uncertainty in the apportionment of emissions could be an important factor in this analysis. There are two things to consider for this uncertainty. First, if emissions are assigned to a larger census tract, then the same level of emissions will result in a lower ambient concentration, on average (the pollution, in effect, being spread out over a larger area means that there is less of it at any given point in that area). The opposite is true as well (i.e., assigning emissions to a smaller census tract will result in higher average concentrations). Second, if emissions are assigned to a less populated census tract, fewer people will be exposed to the resulting concentration of air pollution and the population-weighting at the county scale will predict a lower concentration and thus a lower

ratio. Again, the opposite is true (i.e., emissions assigned to higher-populated tracts leads to an overestimate of concentration and ratio).

We do not anticipate a high degree of uncertainty associated with treating mobile sources as a series of radial points within census tracts, although this may be more of a concern for counties and census tracts that cover a large geographic area. The Benefits Module uses average concentrations at a much larger geographic scale (i.e., county-level), which would tend to underestimate the importance of local hotspot impacts that are not detected by the NATA approach. Some bias may result, however, if the population within a census tract is located closer to and therefore more exposed to pollution from major roads or other low-level releases than our analysis assumes.

The health benefits in the Benefits Module are for $PM_{2.5}$ generically and are not dependent on the precise chemical composition of the $PM_{2.5}$ emissions in a particular area. Therefore the only likely significance associated with not considering atmospheric chemistry is if chemical reactions could lead to either loss or formation of $PM_{2.5}$. The loss of directly-emitted diesel PM through chemical reactions is unlikely, since the impacts from diesel PM tend to be highly local for these source types (e.g., no high stacks, minimal exit velocity) and there is insufficient time for reactions to occur before concentrations have been diluted by dispersion alone. Dilution of diesel PM occurs in less than 1 mile, or less than 20 minutes at even slow wind speeds, which is much faster than the typical atmospheric half-life of $PM_{2.5}$, which is considered to be on the order of days to weeks (e.g. Wilson and Suh, 1997).

In addition, the exposure and benefit-per-ton values to not include highly localized exposures, such as those that occur when diesel exhaust “self-pollutes” the cabin on the vehicle from which it has been emitted. This phenomenon has been studied extensively in diesel school buses, and the data indicate it can be a significant source of exposure from older diesel engines (e.g. Marshall and Behrentz, 2005). This Benefits Module does not capture this type of micro-scale exposure and thus the benefits estimate does not include the benefits of reducing these types of exposures.

Uncertainties in the use of NEI emissions and NATA-predicted ambient concentration may be reduced by considering the following when calculating health benefits using the Benefits Module:

> The highest uncertainties in the Benefits Module’s emissions, dispersion, health, and monetary benefits calculations are likely all associated with considering only a single location or project. Uncertainties that may have either a positive or negative bias when considered together are more likely to be substantially smaller when considering multiple emissions reductions over larger geographic areas, to the extent that such bias is not highly correlated with population.
> The results of the Benefits Module may be used to characterize the relative benefits of diesel emission reduction projects between areas, but comparisons are likely to be more uncertain when comparing areas in different states, where differences in underlying methodology (e.g., local submission of emissions information to NEI) are likely to be more significant.
> The benefits module is most appropriate when used to estimate scale and relative distribution of results (as opposed to precise predictions) and thus should be used for

purposes where this type of estimate is appropriate only. These results are not an adequate substitute for a more refined emissions, dispersion, and health impacts analysis in support of broader decision-making.

Both the calculation of air concentrations from emissions estimates and the subsequent estimation of the health benefits of those improvements in air quality are subject to significant uncertainty. As stated earlier, these estimates should be considered just that: estimates, and not precise calculations or predictions.

C. Quality Assurance

Figure 1 and Tables 2 through 4 are designed to examine whether the highest predicted benefit-per-ton results are reasonable. One of the primary concerns with our methodology is with counties that may experience substantial diesel impacts due to atmospheric transport from surrounding counties, but may not themselves have substantial emissions. This would likely skew the results towards unusually high benefit-per-ton numbers in those counties (i.e., skewed higher ratios of NATA-predicted diesel PM concentrations versus county emissions would be used as inputs for benefits calculations in BenMAP).

Figure 1 is a plot of monetary benefit-per-ton of diesel emissions reduced (expressed in $/ton) for each county in the United States versus total emissions (tons/year), by source, in that county. This figure illustrates two main points. First, there are few, if any, outliers with high benefit-per-ton but low local emissions. Although this figure cannot illustrate sufficiently whether the low-emitting counties are nonetheless skewed higher by atmospheric transport than would otherwise be expected, no low-emitting counties have benefit-per-ton results beyond what is observed for higher emitting, and thus more certain, counties. Second, the distributions show a relative positive trend; that is, benefit-per-ton estimates increase with county emissions. This result is reasonable because higher emitting counties also tend to be more populated counties and the combination of a higher density of sources and population in proximity to each other would lead to higher anticipated health benefits for diesel exhaust reductions.

Another way to consider the impacts of atmospheric transport either into or out of a county is to estimate the import/export factor. This factor describes the relationship between the change in NATA-predicted ambient concentration to the change in emissions density for that county. Figure 2 shows a plot of monetary benefit-per-ton of diesel emissions reduced (expressed in $/ton) for each county in the United States versus the ratio of change in concentration versus change in emissions density. This can be indicated by. $c_i/(e_i/a_i)$, where c_i, e_i, and a_i are the concentration, emissions, and area of county *i*. Counties that are highest in $c_i/(e_i/a_i)$ would be indicative of those that are most likely to import a relatively large portion of diesel PM, while counties that are lowest in $c_i/(e_i/a_i)$ would be indicative of those that are most likely to export a relatively large portion of diesel PM. A high import/export value indicates the air concentrations in the county are likely affected by imports of diesel PM from other counties. A low value indicates the county is likely to export a large portion of the diesel PM emitted there to other counties.

Figure 1: Monetary benefit-per-ton of diesel emissions reduced ($/ton) for all counties in the United States plotted versus source-specific diesel emissions (tons/year) in that county. Results are presented for (a) total diesel sources, (b) on-road diesel sources, and (c) non-road diesel sources.

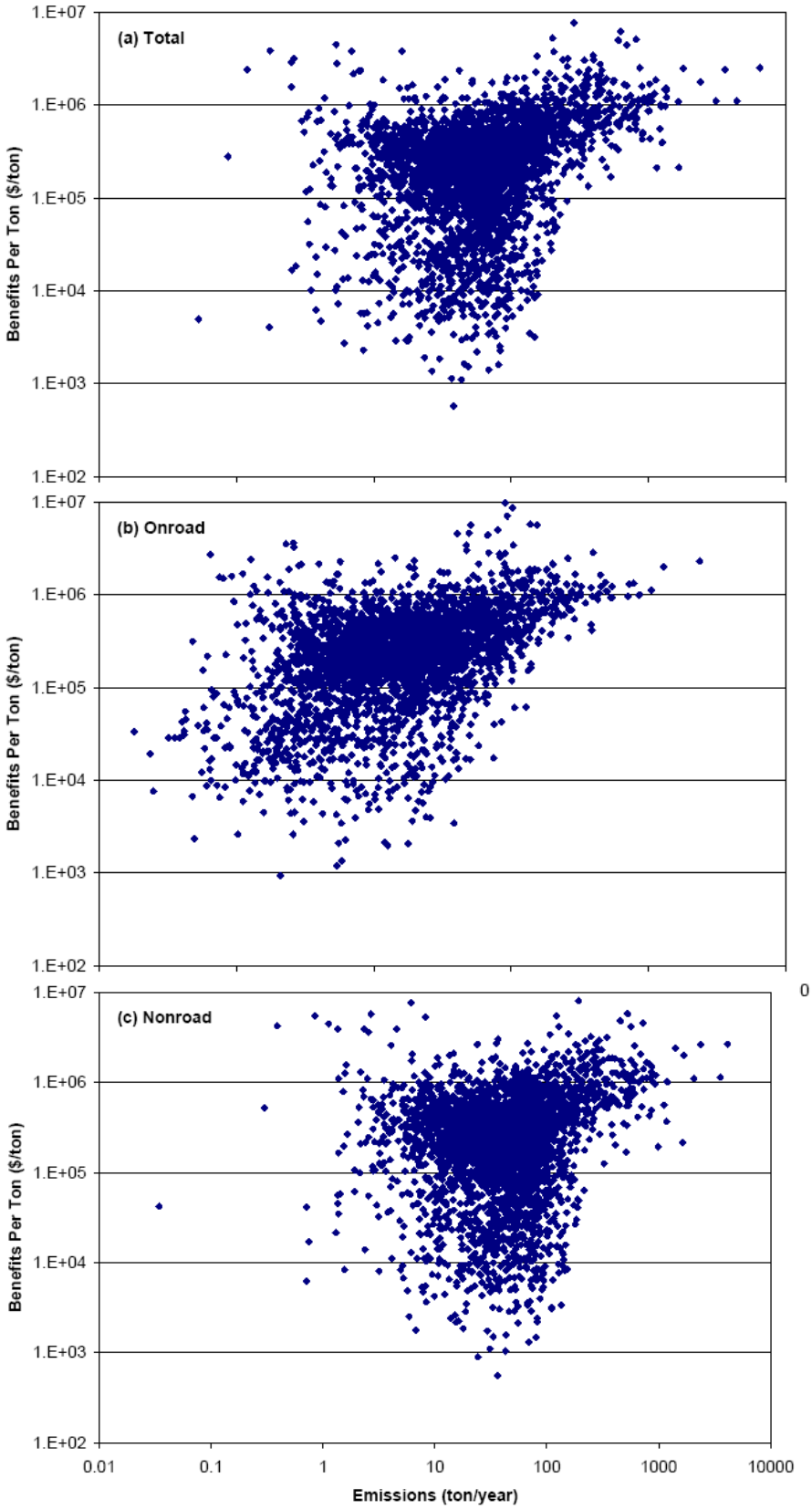
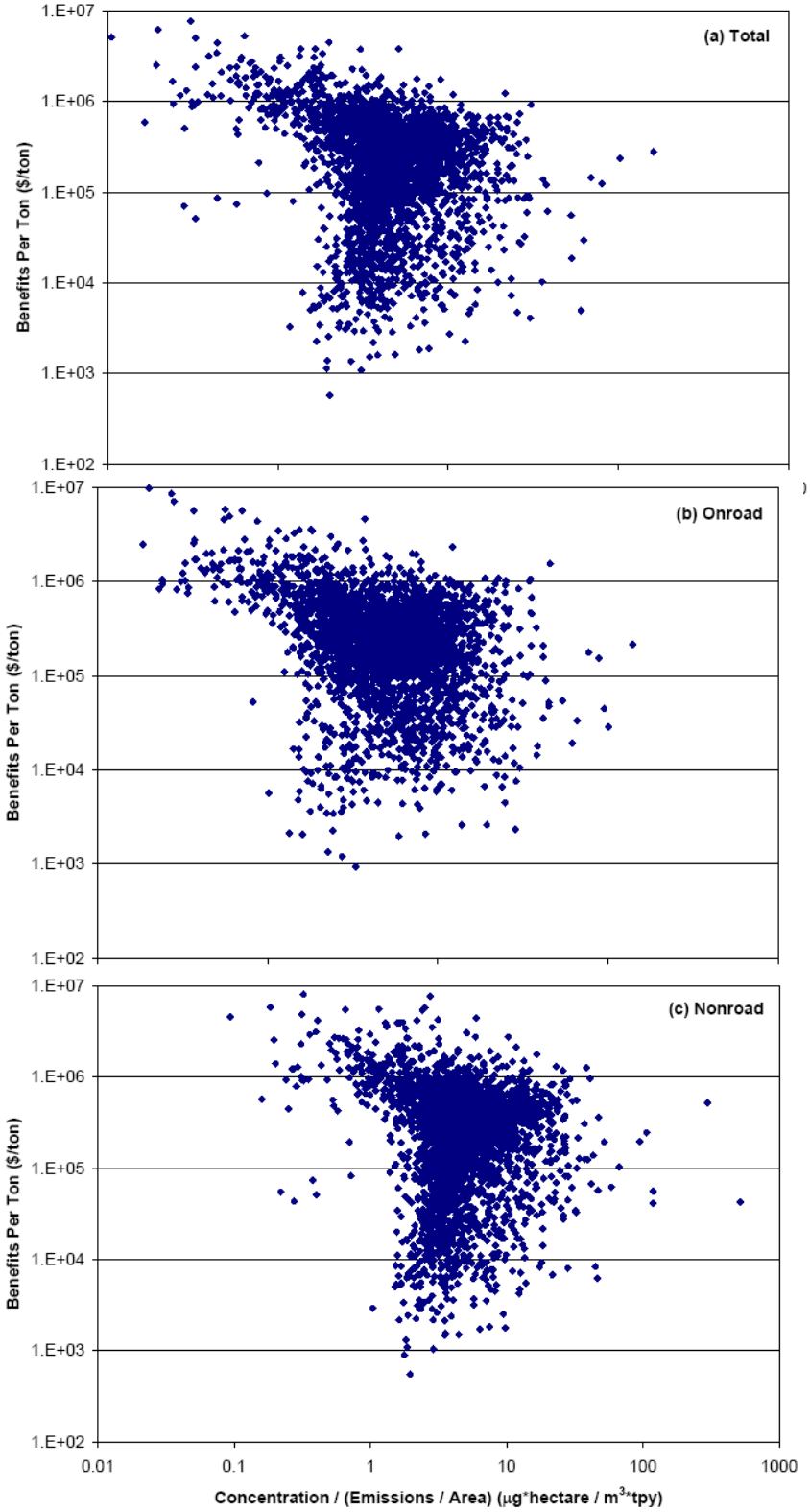


Figure 2: Monetary benefit-per-ton of diesel emissions reduced ($/ton) for all counties in the United State versus the import/export factor. This factor is a ratio of change in concentration versus change in emissions density for each county, i.e. $c_i/(e_i/a_i)$, where c_i, e_i, and a_i are the concentration, emissions, and area of county *i*. Results are presented for (a) total diesel sources, (b) on-road diesel sources, and (c) non-road diesel sources.



Tables 2 through 4 show the counties with highest predicted benefit-per-ton due to reductions from total diesel sources, on-road diesel sources, and non-road diesel sources, respectively. Tables 5 through 7 show the benefit-per-ton for counties with the lowest emissions for total diesel sources, on-road diesel sources, and non-road diesel sources, respectively. Tables 8 through 10 show the counties with the highest import/export factor (i.e. counties likely to import) and Tables 11 through 13 show the counties with the lowest import/export factor (i.e. counties likely to export) nationally.

A closer examination of the counties with the highest-predicted benefit-per-ton estimates (Tables 2 through 4) shows that counties with a high density of sources and/or high population density (such as Bronx, Kings, New York, Manhattan, and Queens Counties, which are part of the City of New York) have some of the highest benefit-per-ton estimates, which is expected. The independent cities of Virginia, i.e. Fairfax, Poquoson, Portsmouth, Winchester, Franklin, Lexington, and Falls Church, also show very high benefit-per-ton results, especially relative to their local emissions. These results do not appear unreasonable since these cities tend to be fairly dense with both sources and receptors. Many of these same counties have the lowest import/export factors in Tables 11 through 13, supporting the assertion that, if anything, the counties are mostly exporters of diesel emissions and the benefit-per-ton estimates may be underestimates.

Most of the instances of unusually high or low benefit-per-ton results are for non-road emissions. For example, the Loving County, TX, benefits of $42,000 per ton (Table 7), while small, is most likely due entirely to transport of outside pollutants, because there are essentially no local sources. Similarly, the $520,000 per ton for Alpine County is quite large, given the minimal local sources (0.3 tons/year) and sparsely populated, low density county. The import/export factor analysis supports both of these assertions, since both counties have a very high ratio (Table 13), and could thus be interpreted as diesel importers. Two other counties with low emissions (<2 tons/year) but high predicted benefit-per-ton (> $500,000 per ton) are Owsley County, KY, and Clay County, WV.

In order to acknowledge this uncertainty, the diesel benefits calculator takes the following approach. First, in addition to reporting results for the county selected, the results are also calculated and reported using statewide benefit-per-ton values in order to provide context. Second, for all counties with import/export factors in the lowest 5th percentile – for either on-road or non-road sources, depending on the query – the results are flagged with the following message:

> Benefits estimates are "flagged" for this county, indicating that we have less confidence in these results due to a large amount of inter-county transport of emissions. The impacts estimation tool may be <u>underestimating</u> benefits for emissions reduction projects in this county because it has a relatively high density of emissions compared to surrounding areas. As a result, this county is likely to be a net <u>exporter</u> of diesel emissions, and many of the benefits of reducing these emissions are likely to take place in downwind counties. Please take this increased uncertainty into account when interpreting your results.

Also, for all counties with import/export factors in the highest 5th percentile – for either on-road or non-road sources, depending on the query – the results are flagged with the following message:

> Benefits estimates are "flagged" for this county, indicating that we have less confidence in these results due to a large amount of inter-county transport of emissions. The impacts estimation tool may be overestimating the benefits for emissions reduction projects in this county because it has relatively few emissions compared to surrounding areas. As a result, this county is likely to be a net importer of diesel emissions, and air quality is significantly affected by emissions in upwind counties. Please take this increased uncertainty into account when interpreting your results.

EPA also calculated a population-weighted average of the county benefit-per-ton values within each state and within the entire United States. The procedure was identical to the population-weighting performed for averaging census tract ambient concentrations to the county level.

For total diesel sources, we calculated a range from $3.2 million per ton for New York State to $68,000 per ton for Wyoming. The national population-weighted average was $1.2 million per ton. The national benefit-per-ton value is somewhat higher than the national mobile source benefit-per-ton from carbonaceous particles from all mobile sources of $730,000 that was calculated as part of the ozone NAAQS RIA (U. S. EPA, 2008a). For on-road diesel sources we calculated a range from $3.8 million per ton for New York State to $63,000 per ton for Wyoming. For non-road diesel sources we calculated a range from $3.2 million per ton for New York State to $73,000 per ton for Wyoming. The national population-weighted average for on-road sources and non-road sources are $1.2 million per ton of diesel reduced. This is also somewhat higher than the on-road and non-road estimates calculated as part of the ozone NAAQS RIA, which are $740,000 per ton and $720,000, respectively.

The benefit-per-ton estimates from this project are clearly very different from those in the most recent ozone RIA. However, the divergence may be due to the fact that the diesel PM benefit-per-ton estimates reflect air quality changes from diesel sources alone. Conversely, the benefit-per-ton estimates developed for the ozone RIA reflect air quality changes from reductions in carbonaceous particles across all on-road and non-road mobile sources. Finally, these two benefit-per-ton estimates may diverge due to inherent differences in the model used to estimate air quality impacts. As described above, EPA used a dispersion model to estimate diesel PM air quality changes; conversely, EPA used a photochemical grid model to generate air quality estimates for the benefit-per-ton estimates that supported the ozone RIA.

Table 2: Counties with Highest Predicted Benefit-per-ton Estimates ($/ton) for Total Diesel Sources.

County	State	2000 Population	Emissions input (tons/year)	County area (hectares)	Import/ export factor	Benefits output ($/ton)
Bronx County	NEW YORK	1,332,650	290	40	0.31	7,800,000
Kings County	NEW YORK	2,465,326	630	60	0.20	6,200,000
Baltimore city	MARYLAND	651,154	200	85	0.64	5,300,000
New York County	NEW YORK	1,537,195	820	23	0.11	5,100,000
Queens County	NEW YORK	2,229,379	610	110	0.33	5,000,000
Fairfax city	VIRGINIA	21,498	5.3	10	0.99	4,500,000
Philadelphia County	PENNSYLVANIA	1,517,550	700	150	0.44	4,500,000
Poquoson city	VIRGINIA	11,566	1.8	20	5.1	3,900,000
Portsmouth city	VIRGINIA	100,565	16	35	1.4	3,800,000
Winchester city	VIRGINIA	23,585	6.9	11	1.7	3,800,000
Ocean County	NEW JERSEY	510,916	210	620	3.1	3,800,000
Hudson County	NEW JERSEY	608,975	400	57	0.44	3,500,000
Passaic County	NEW JERSEY	489,049	150	200	1.8	3,400,000
Falls Church city	VIRGINIA	10,377	2.6	3.5	1.3	3,200,000
Richmond County	NEW YORK	443,728	260	48	0.39	3,200,000
Bergen County	NEW JERSEY	884,118	400	250	1.0	3,100,000
Camden County	NEW JERSEY	508,932	240	230	1.6	3,100,000
Essex County	NEW JERSEY	793,633	340	130	0.61	3,100,000
Franklin city	VIRGINIA	8346	2.5	3.2	0.60	2,900,000
Hopewell city	VIRGINIA	22,354	5.4	8.8	1.0	2,800,000

Table 3: Counties with Highest Predicted Benefit-per-ton Estimates ($/ton) for On-road Diesel Sources.

County	State	2000 Population	Emissions input (tons/year)	County area (hectares)	Import/ export factor	Benefits output ($/ton)
New York County	NEW YORK	1,537,195	91	23	0.20	9,900,000
Kings County	NEW YORK	2,465,326	100	60	0.27	8,700,000
Bronx County	NEW YORK	1,332,650	94	40	0.28	7,000,000
Philadelphia County	PENNSYLVANIA	1,517,550	140	150	0.56	5,800,000
Queens County	NEW YORK	2,229,379	150	110	0.37	5,700,000
Hudson County	NEW JERSEY	608,975	50	57	0.71	5,700,000
Baltimore city	MARYLAND	651,154	79	85	0.60	5,000,000
Ocean County	NEW JERSEY	510,916	49	620	3.7	4,600,000
Richmond County	NEW YORK	443,728	41	48	0.55	4,500,000
Essex County	NEW JERSEY	793,633	68	130	0.87	4,400,000
Bristol County	RHODE ISLAND	50,648	2.6	23	1.8	3,600,000
Winchester city	VIRGINIA	23,585	2.3	11	1.5	3,500,000
Bergen County	NEW JERSEY	884,118	100	250	1.2	3,500,000
Passaic County	NEW JERSEY	489,049	47	200	1.8	3,400,000
Fairfax city	VIRGINIA	21,498	2.6	10	1.4	3,300,000
Providence County	RHODE ISLAND	621,602	48	430	2.3	3,000,000
Orange County	CALIFORNIA	2,846,289	400	800	1.3	2,900,000
Union County	NEW JERSEY	522,541	69	110	0.73	2,800,000
District of Columbia	DISTRICT OF COLUMBIA	572,059	90	66	0.37	2,800,000
Delaware County	PENNSYLVANIA	550,864	83	190	1.0	2,800,000

Table 4: Counties with Highest Predicted Benefit-per-ton Estimates ($/ton) for Non-road Diesel Sources.

County	State	2000 Population	Emissions input (tons/year)	County area (hectares)	Import/ export factor	Benefits output ($/ton)
Bronx County	NEW YORK	1,332,650	190	40	0.32	8,100,000
Portsmouth city	VIRGINIA	100,565	6.2	35	2.7	7,800,000
Kings County	NEW YORK	2,465,326	530	60	0.18	5,800,000
Fairfax city	VIRGINIA	21,498	2.7	10	2.5	5,800,000
Baltimore city	MARYLAND	651,154	121	85	0.66	5,500,000
Franklin city	VIRGINIA	8346	0.84	3.2	1.2	5,500,000
Hampton city	VIRGINIA	146,437	8.2	51	2.4	5,400,000
Queens County	NEW YORK	2,229,379	460	105	0.31	4,800,000
New York County	NEW YORK	1,537,195	730	23	0.093	4,600,000
Poquoson city	VIRGINIA	11,566	1.1	20	6.0	4,500,000
Lexington city	VIRGINIA	6867	0.39	5.1	3.1	4,300,000
Camden County	NEW JERSEY	508,932	130	230	2.2	4,200,000
Philadelphia County	PENNSYLVANIA	1,517,550	560	150	0.41	4,200,000
Winchester city	VIRGINIA	23,585	4.6	11	1.7	4,000,000
Falls Church city	VIRGINIA	10,377	1.4	3.5	1.6	3,900,000
Staunton city	VIRGINIA	23,853	2.3	13	1.6	3,900,000
Colonial Heights city	VIRGINIA	16,897	2.4	7.0	1.3	3,900,000
Hopewell city	VIRGINIA	22,354	2.6	8.8	1.3	3,600,000
Ocean County	NEW JERSEY	510,916	160	620	2.8	3,500,000
Passaic County	NEW JERSEY	489,049	110	200	1.8	3,400,000

Table 5: Benefit-per-ton of Diesel Emissions Reduced ($/ton) for Counties with the Lowest Emissions of Total Diesel Sources.

County	State	2000 Population	Emissions input (tons/year)	County area (hectares)	Import/ export factor	Benefits output ($/ton)
Loving County	TEXAS	67	0.53	660	60	4900
Alpine County	CALIFORNIA	1208	0.87	730	160	280,000
Lexington city	VIRGINIA	6867	1.2	5.1	1.8	2,400,000
Hinsdale County	COLORADO	790	1.7	1100	30	4100
Poquoson city	VIRGINIA	11,566	1.8	20	5.1	3,900,000
Franklin city	VIRGINIA	8346	2.5	3.2	0.60	2,800,000
Buena Vista city	VIRGINIA	6349	2.5	4.9	0.87	1,600,000
Daggett County	UTAH	921	2.5	710	14	17,000
Falls Church city	VIRGINIA	10,377	2.6	3.5	1.3	3,200,000
Edwards County	TEXAS	2162	2.7	2100	53	19,000
Owsley County	KENTUCKY	4858	3.0	200	23	670,000
Robertson County	KENTUCKY	2266	3.1	110	21	510,000
Real County	TEXAS	3047	3.2	690	19	120,000
Wirt County	WEST VIRGINIA	5873	3.3	230	26	740,000
McMullen County	TEXAS	851	3.3	1100	53	56,000
Norton city	VIRGINIA	3904	3.3	5.0	0.79	820,000
Irion County	TEXAS	1771	3.4	1100	20	32,000
Mineral County	NEVADA	5071	3.4	3800	80	120,000
Esmeralda County	NEVADA	971	3.5	3600	36	10,000
Glascock County	GEORGIA	2556	3.6	150	14	230,000

Table 6: Benefit-per-ton of Diesel Emissions Reduced ($/ton) Results for Counties with the Lowest Emissions of On-road Diesel Sources.

County	State	2000 Population	Emissions Input (tons/year)	County Area (hectares)	Import/ export factor	Benefits Output ($/ton)
Arthur County	NEBRASKA	444	0.18	720	66	34,000
McPherson County	NEBRASKA	533	0.24	870	62	19,000
Petroleum County	MONTANA	493	0.25	1700	29	7600
Loup County	NEBRASKA	712	0.32	570	29	28,000
Esmeralda County	NEVADA	971	0.36	3600	100	29,000
Thomas County	NEBRASKA	729	0.39	700	22	28,000
Hooker County	NEBRASKA	783	0.40	720	21	43,000
Keya Paha County	NEBRASKA	983	0.41	780	21	34,000
Blaine County	NEBRASKA	583	0.41	710	23	31,000
Banner County	NEBRASKA	819	0.42	750	54	55,000
Harding County	NEW MEXICO	810	0.42	2100	95	45,000
Slope County	NORTH DAKOTA	767	0.47	1200	19	6700
Storey County	NEVADA	3399	0.48	260	24	320,000
Loving County	TEXAS	67	0.49	660	29	2300
Greeley County	KANSAS	1534	0.53	790	19	39,000
Grant County	NEBRASKA	747	0.55	770	17	12,000
Alpine County	CALIFORNIA	1208	0.57	730	88	150,000
Buffalo County	SOUTH DAKOTA	2032	0.58	500	12	61,000
Stanley County	SOUTH DAKOTA	2772	0.58	1500	24	34,000
Logan County	NEBRASKA	774	0.58	560	16	22,000

Table 7: Benefit-per-ton of Diesel Emissions Reduced ($/ton) Results for Counties with the Lowest Emissions of Non-road Diesel Sources.

County	State	2000 Population	Emissions Input (tons/year)	County area (hectares)	Import/ export factor	Benefits Output ($/ton)
Loving County	TEXAS	67	0.034	660	520	42,000
Alpine County	CALIFORNIA	1208	0.30	730	300	520,000
Lexington city	VIRGINIA	6867	0.39	5.1	3.1	4,300,000
Edwards County	TEXAS	2162	0.70	2100	120	41,000
Hinsdale County	COLORADO	790	0.71	1100	46	6200
San Juan County	COLORADO	558	0.75	400	13	17,000
Franklin city	VIRGINIA	8346	0.84	3.2	1.2	5,500,000
Poquoson city	VIRGINIA	11,566	1.1	20	6.0	4,500,000
Daggett County	UTAH	921	1.3	710	19	22,000
Irion County	TEXAS	1771	1.4	1100	28	45,000
Falls Church city	VIRGINIA	10,377	1.4	3.5	1.6	3,900,000
Catron County	NEW MEXICO	3543	1.4	7000	120	56,000
Norton city	VIRGINIA	3904	1.4	5.0	1.1	1,100,000
Sterling County	TEXAS	1393	1.4	920	32	35,000
Real County	TEXAS	3047	1.4	690	27	170,000
Crockett County	TEXAS	4099	1.4	2800	47	57,000
Owsley County	KENTUCKY	4858	1.5	196	27	790,000
Kimble County	TEXAS	4468	1.5	1300	52	200,000
King County	TEXAS	356	1.6	940	45	8300
Clay County	WEST VIRGINIA	10,330	1.6	350	38	1,300,000

Table 8: Counties with Highest Import/Export Factors for Total Diesel Sources

County	State	2000 Population	Emissions Input (tons/year)	County Area (hectares)	Import/ export factor	Benefits Output ($/ton)
Alpine County	CALIFORNIA	1208	0.87	730	160	280,000
Nye County	NEVADA	32485	24	18,000	100	240,000
Mineral County	NEVADA	5071	3.4	3800	80	120,000
Inyo County	CALIFORNIA	17945	20	10,000	69	140,000
Catron County	NEW MEXICO	3543	4.5	7000	63	29,000
Loving County	TEXAS	67	0.53	660	60	4900
Edwards County	TEXAS	2162	2.7	2000	53	19,000
McMullen County	TEXAS	851	3.3	1100	53	56,000
Moffat County	COLORADO	13184	27	4800	38	62,000
Hamilton County	NEW YORK	5379	7.8	1800	38	120,000
Sierra County	CALIFORNIA	3555	4.9	960	36	140,000
Esmeralda County	NEVADA	971	3.5	3600	36	10,000
Graham County	NORTH CAROLINA	7993	4.0	300	31	930,000
Hinsdale County	COLORADO	790	1.7	1100	30	4100
Malheur County	OREGON	31615	75	9900	30	85,000
Highland County	VIRGINIA	2536	4.1	420	29	250,000
Coconino County	ARIZONA	116320	260	19,000	29	93,000
Mono County	CALIFORNIA	12853	15	3100	29	59,000
Pendleton County	WEST VIRGINIA	8196	7.8	690	29	380,000
Greenlee County	ARIZONA	8547	4.1	1800	28	86,000

Table 9: Counties with Highest Import/Export Factors for On-road Diesel Sources

County	State	2000 Population	Emissions Input (tons/year)	County Area (hectares)	Import/ export factor	Benefits Output ($/ton)
Mineral County	NEVADA	5071	0.61	3800	140	220,000
Esmeralda County	NEVADA	971	0.36	3600	100	29,000
Harding County	NEW MEXICO	810	0.42	2100	95	45,000
Alpine County	CALIFORNIA	1208	0.57	730	88	153,000
Nye County	NEVADA	32485	3.9	18,000	77	180,000
Arthur County	NEBRASKA	444	0.18	720	66	34,000
McPherson County	NEBRASKA	533	0.24	870	62	19,000
Banner County	NEBRASKA	819	0.42	750	54	55,000
Hancock County	TENNESSEE	6786	0.75	220	46	1,600,000
Brewster County	TEXAS	8866	2.3	6100	45	52,000
McMullen County	TEXAS	851	1.4	1100	45	48,000
Inyo County	CALIFORNIA	17945	10	10,000	43	89,000
Skamania County	WASHINGTON	9872	4.2	1700	42	210,000
Sierra County	CALIFORNIA	3555	1.3	960	42	160,000
Meagher County	MONTANA	1932	0.60	2400	42	36,000
Catron County	NEW MEXICO	3543	3.1	7000	39	18,000
Lincoln County	NEVADA	4165	1.3	11,000	38	14,000
Highland County	VIRGINIA	2536	1.2	420	38	320,000
Mariposa County	CALIFORNIA	17130	4.3	1500	36	460,000
Webster County	WEST VIRGINIA	9719	1.5	560	36	680,000

Table 10: Counties with Highest Import/Export Factors for Non-road Diesel Sources

County	State	2000 Population	Emissions Input (tons/year)	County Area (hectares)	Import/ export factor	Benefits Output ($/ton)
Loving County	TEXAS	67	0.034	660	520	42,000
Alpine County	CALIFORNIA	1208	0.30	730	300	520,000
Catron County	NEW MEXICO	3543	1.4	7000	120	56,000
Edwards County	TEXAS	2162	0.70	2100	120	41,000
Nye County	NEVADA	32485	20	18,000	110	250,000
Inyo County	CALIFORNIA	17945	10	10,000	96	200,000
Mineral County	NEVADA	5071	2.8	3800	67	100,000
McMullen County	TEXAS	851	1.9	1100	59	62,000
Kimble County	TEXAS	4468	1.5	1300	52	195,000
Brooks County	TEXAS	7976	2.1	970	47	360,000
Crockett County	TEXAS	4099	1.4	2800	47	57,000
Hinsdale County	COLORADO	790	0.71	1100	46	6200
King County	TEXAS	356	1.6	940	45	8300
Hamilton County	NEW YORK	5379	4.0	1800	43	140,000
Moffat County	COLORADO	13184	18	4800	42	67,000
St. Helena Parish	LOUISIANA	10525	4.2	410	41	960,000
Coconino County	ARIZONA	116320	130	19,000	39	120,000
Clay County	WEST VIRGINIA	10330	1.6	350	38	1,300,000
Blanco County	TEXAS	8418	3.6	720	37	310,000
Park County	COLORADO	14523	8.2	2200	37	130,000

Table 11: Counties with Lowest Import/Export Factors for Total Diesel Sources

County	State	2000 Population	Emissions Input (tons/year)	County Area (hectares)	Import/ export factor	Benefits Output ($/ton)
New York County	NEW YORK	1537195	820	23	0.11	5,200,000
Norfolk city	VIRGINIA	234403	460	48	0.16	590,000
San Francisco County	CALIFORNIA	776733	870	47	0.19	2,500,000
Kings County	NEW YORK	2465326	630	60	0.20	6,200,000
Suffolk County	MASSACHUSETTS	689807	370	69	0.24	1,700,000
Denver County	COLORADO	554636	400	100	0.24	940,000
Bristol city	VIRGINIA	17367	17	5.1	0.27	1,200,000
San Juan County	WASHINGTON	14077	52	56	0.28	70,000
Newport News city	VIRGINIA	180150	180	75	0.28	510,000
Arlington County	VIRGINIA	189453	180	26	0.29	1,300,000
Bronx County	NEW YORK	1332650	290	40	0.31	7,700,000
Emporia city	VIRGINIA	5665	10	3.5	0.31	1,000,000
Fredericks-burg city	VIRGINIA	19279	26	7.3	0.31	880,000
Williamsburg city	VIRGINIA	11998	9.7	4.8	0.32	890,000
District of Columbia	DISTRICT OF COLUMBIA	572059	370	6	0.32	2,400,000
Dukes County	MASSACHUSETTS	14987	180	95	0.33	51,000
Queens County	NEW YORK	2229379	610	110	0.33	5,000,000
Manassas Park city	VIRGINIA	10290	6.6	1.7	0.34	990,000
Lynchburg city	VIRGINIA	65269	48	23	0.38	1,200,000
Richmond County	NEW YORK	443728	260	48	0.39	3,200,000

Table 12: Counties with Lowest Import/Export Factors for On-road Diesel Sources

County	State	2000 Population	Emissions Input (tons/year)	County area (hectares)	Import/ export factor	Benefits Output ($/ton)
San Francisco County	CALIFORNIA	776733	260	47	0.19	2,500,000
New York County	NEW YORK	1537195	91	23	0.20	9,900,000
Norfolk city	VIRGINIA	234403	33	48	0.23	840,000
Bristol city	VIRGINIA	17367	8.5	5.1	0.24	1,100,000
Denver County	COLORADO	554636	140	100	0.24	940,000
Kings County	NEW YORK	2465326	100	60	0.27	8,600,000
Bronx County	NEW YORK	1332650	94	40	0.28	7,100,000
Fredericks-burg city	VIRGINIA	19279	9.7	7.3	0.29	830,000
Emporia city	VIRGINIA	5665	2.9	3.5	0.31	1,000,000
Danville city	VIRGINIA	48411	16	17	0.32	1,200,000
Lynchburg city	VIRGINIA	65269	22	23	0.33	1,000,000
Franklin city	VIRGINIA	8346	1.7	3.2	0.33	1,600,000
Hampton city	VIRGINIA	146437	25	51	0.34	750,000
Harrisonburg city	VIRGINIA	40468	12	11	0.34	870,000
Queens County	NEW YORK	2229379	150	110	0.37	5,700,000
Suffolk County	MASSACHUSETTS	689807	84	69	0.37	2,600,000
District of Columbia	DISTRICT OF COLUMBIA	572059	90	66	0.37	2,800,000
St. Louis city	MISSOURI	348189	140	72	0.37	1,700,000
Arlington County	VIRGINIA	189453	35	26	0.38	1,700,000
Pinellas County	FLORIDA	921482	210	310	0.41	1,400,000

Table 13: Counties With Lowest Import/Export Factors for Non-road Diesel Sources

County	State	2000 Population	Emissions Input (tons/year)	County area (hectares)	Import/ export factor	Benefits Output ($/ton)
New York County	NEW YORK	1537195	730	23	0.094	4,600,000
Norfolk city	VIRGINIA	234403	430	48	0.16	580,000
Kings County	NEW YORK	2465326	530	60	0.18	5,800,000
San Francisco County	CALIFORNIA	776733	610	47	0.20	2,500,000
Suffolk County	MASSACHUSETTS	689807	280	69	0.20	1,400,000
San Juan County	WASHINGTON	14077	51	56	0.22	55,000
Denver County	COLORADO	554636	260	100	0.24	940,000
Newport News city	VIRGINIA	180150	150	75	0.25	450,000
Arlington County	VIRGINIA	189453	140	26	0.27	1,200,000
Dukes County	MASSACHUSETTS	14987	170	95	0.28	43,000
Williamsburg city	VIRGINIA	11998	8.6	4.8	0.29	800,000
Bristol city	VIRGINIA	17367	8.1	5.1	0.29	1,300,000
District of Columbia	DISTRICT OF COLUMBIA	572059	280	66	0.31	2,300,000
Emporia city	VIRGINIA	5665	7.2	3.5	0.31	1,000,000
Queens County	NEW YORK	2229379	460	110	0.31	4,800,000
Manassas Park city	VIRGINIA	10290	5.3	1.7	0.32	920,000
Bronx County	NEW YORK	1332650	190	40	0.32	8,100,000
Fredericks-burg city	VIRGINIA	19279	17	7.3	0.33	920,000
Salem city	VIRGINIA	24747	17	10	0.35	940,000
Richmond County	NEW YORK	443728	220	48	0.36	2,900,000

Figures 3 and 4 below illustrate the geographic distribution of county-level $PM_{2.5}$ benefit-per-ton estimates by source type. Two key summary conclusions may be drawn:

> *There is a high degree of spatial heterogeneity.* For example, the states of California, New Jersey and Florida contain among the highest benefit-per-ton estimates, while interior states such as North and South Dakota contain very low estimates. Human health benefit estimates are strongly influenced by population exposure. Other things being equal, counties with higher population density will exhibit larger benefit-per-ton estimates.
>
> *Estimates are not equally accurate for all counties.* The Benefits Module "flags" results for counties where the non-road and on-road benefit-per-ton values are likely to be more uncertain due to transport of fine particle concentrations into or out of a county. These counties, identified by the import/export factors, are hashed on these maps. They are often but not always counties with very high and very low emissions.

Figure 3: Distribution of County-Level Benefit-per-Ton of Diesel PM Emission Reductions: <u>On-road</u> Sources (Laden et al. mortality estimate, 2006$)

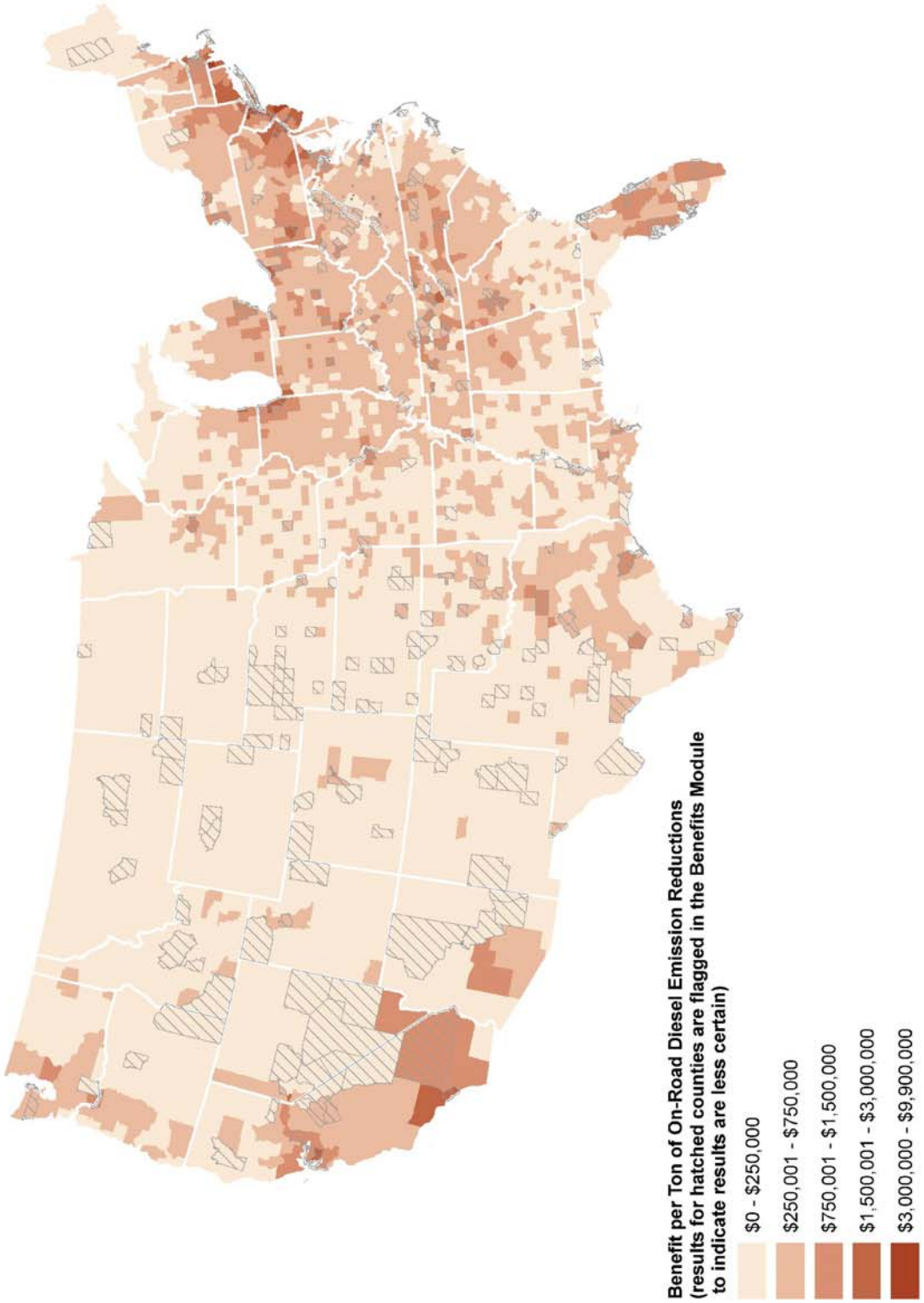
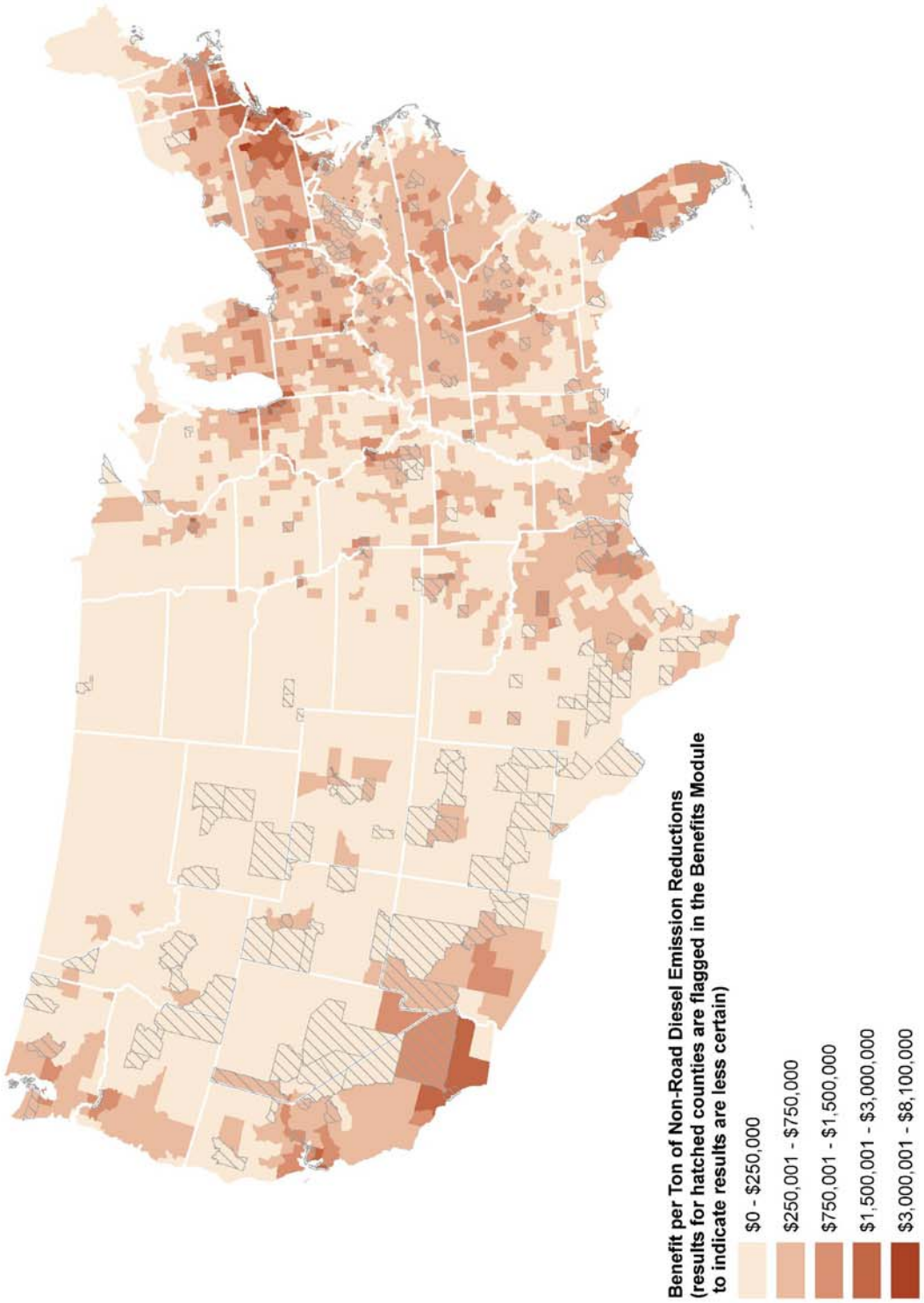


Figure 4: Distribution of County-Level Benefit-per-Ton of Diesel PM Emission Reductions: <u>Non-road</u> sources (Laden et al. mortality estimate, 2006$)



VI. EXAMPLE RESULTS

An example set of results for the Benefits Module are presented to assist users in understanding how the tools works. To provide example results, the Quantifier was run twice with two different scenarios. Results are presented for the "current" year – the year the emission reductions take place -- and dollar values are presented in 2006 dollars. These benefits would be expected to be similar in subsequent years, assuming that the performance of the emission reduction technology stays constant (for example, installed diesel catalysts continue to perform at the same efficiency). This is based on existing assumptions inherent in the Quantifier and some field research (Chandler et al., 2003). Given the scales and uncertainty in this analysis, we assume that population growth would slightly increase the benefits at roughly the same rate that discounting future benefits would reduce them. Therefore, this annual benefits number can be used as a rough estimate of annual benefits for each year of the lifetime of the engine retrofit.

To calculate these example results, the Quantifier was run for two counties: Cook County, IL and Anderson County, Texas. Cook County is a highly urban county, including the city of Chicago (land area 1,635 square miles and population in 2000 of 5.3 million), while Anderson County is a highly rural county southeast of Dallas (land area 1,078 square miles and population in 2000 of 55,109).

In the example scenario, 100 school buses were retrofitted in 2008 with diesel particulate filters and began using ultra low-sulfur diesel fuel (15 ppm sulfur). The buses were model year 2002 and traveled 13,000 miles per year. Before the retrofit, these 100 buses emitted a total of 0.32 tons/year of diesel PM. The retrofit reduced emissions 85%, or 0.27 tons per year.

In addition, 10 pieces of construction equipment (e.g. tractors, loaders, backhoes) were retrofitted in 2008 with diesel particulate filters and began using low-sulfur diesel fuel (500 ppm sulfur). The equipment was all model year 2000. Before the retrofit, the construction equipment emitted 0.20 tons/year. The retrofit reduced emissions 85%, or 0.17 tons per year.

Table 14 presents the estimates of the economic value of the emission reductions from both scenarios.

Table 14. Example Quantifier and Benefits Module results for Cook County, IL and Anderson County, TX

Benefits Module Results			
county	annual tons diesel PM reduction	annualized costs	annual benefits
Cook County, IL	0.44	$15,203	$1,000,000
Anderson County, TX	0.45	$15,203	$224,000

When reporting benefits estimates, we believe that there are two key uncertainties:

The assumptions used by EPA to derive the benefits-per-ton may differ significantly from the policy scenario in which users apply the benefit-per-ton. Specifically, the types of emission sources controlled, the temporal distribution of emission controls, the types of emissions, the source locations and background $PM_{2.5}$ levels may differ between the modeling scenario used to generate the benefit-per-ton estimates and the user-defined scenario.

The benefits-per-ton do not reflect certain non-linear relationships. Because the benefit-per-ton estimates are averages, they may not reflect non-linear relationships between air quality changes and background $PM_{2.5}$ levels. For example, because the concentration-response functions are non-linear, the estimated change in health impacts is sensitive to the background levels of $PM_{2.5}$ in the atmosphere. Overall we expect this to contribute a small amount to total uncertainty because the functional form of the mortality estimate (which represents the great majority of total benefits) is a nearly flat log-linear form.

VII. LITERATURE CITED

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VIII. WEBSITE INDEX/INTERNET RESOURCES

AERMOD model: www.epa.gov/scram001/dispersion_prefrec.htm#aermod

Assessment System for Population Exposure Nationwide (ASPEN) user's guide is available at www.epa.gov/scram001/userg/other/aspenug.pdf

BenMAP User's Guide Technical Appendices, Appendix I: Uncertainty and Pooling: www.epa.gov/air/benmap/models/BenMAPappendicesSept08.pdf

Diesel Emissions Quantifier

A more detailed description of the Quantifier, and access to the tool itself, can be found at www.epa.gov/cleandiesel/quantifier/

More information on the Quantifier can be found in the Users Guide, which is available on the website at www.epa.gov/cleandiesel/documents/420b10033.pdf

Documentation for the 2002 NEI is provided at www.epa.gov/ttn/chief/net/2002inventory.html

Documentation for the 2002 Mobile NEI is located at ftp://ftp.epa.gov/EmisInventory/2002finalnei/documentation/mobile/2002_mobile_nei_version_3_report_092807.pdf

EPA's MOBILE6 model is used to generate emission factors in grams per mile and then determining total annual tons using annual vehicle miles traveled (VMT): www.epa.gov/oms/m6.htm

EPA's National Mobile Inventory Model: NMIM, www.epa.gov/oms/nmim.htm

EPA's 2002 National-Scale Air Toxics Assessment (NATA): www.epa.gov/ttn/atw/natamain/

EPA's Draft Guidance for Discounting Future Costs and Benefits http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0516-06.pdf/$File/EE-0516-06.pdf?OpenElement

NATA's use of the Hazardous Air Pollutant Exposure Model 5 (HAPEM5) can be found at www.epa.gov/ttn/atw/nata1999/ted/teddraft.html

2002 NATA and past results summarized: www.epa.gov/ttn/atw/nata1999/natafinalfact.html

NATA results: www.epa.gov/ttn/atw/natamain/

Science Advisory Board (SAB) peer review of the NATA approach: www.epa.gov/ttn/atw/sab/sabrev.html

Technology Transfer Network 1999 National-Scale Air Toxic Assessment: www.epa.gov/ttn/atw/nata1999/background.html

User's Guide for the Emissions Modeling System for Hazardous Air Pollutants (EMS-HAP) Version 3.0: www.epa.gov/scram001/userg/other/emshapv3ug.pdf