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Monetized health benefits attributable to mobile source emission reductions across the United States in 2025

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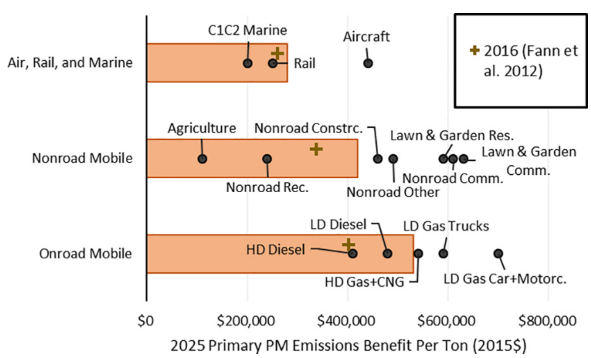
HIGHLIGHTS

- Mobile sources emit pollutants associated with adverse health outcomes.
- $PM_{2.5}$-related benefit per ton (BPT) values estimated for 16 mobile source sectors
- BPT estimates provide a reduced-form tool for monetizing health impacts.
- Can be used to assess health benefits of alternative air quality control scenarios
- Regional (East/West) mobile source BPT values also presented for each sector

GRAPHICAL ABSTRACT



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ABSTRACT

By-products of mobile source combustion processes, such as those associated with gasoline- and diesel-powered engines, include direct emissions of particulate matter as well as precursors to particulate matter and ground-level ozone. Human exposure to fine particulate matter with an aerodynamic diameter smaller than 2.5 μm ($PM_{2.5}$) is associated with increased incidence of premature mortality and morbidity outcomes. This study builds upon recent, detailed source-apportionment air quality modeling to project the health-related benefits of reducing $PM_{2.5}$ from mobile sources across the contiguous U.S. in 2025. Updating a previously published benefits analysis approach, we develop national-level benefit per ton estimates for directly emitted $PM_{2.5}$, SO_2/pSO_4, and NO_X for 16 mobile source sectors spanning onroad vehicles, nonroad engines and equipment, trains, marine vessels, and aircraft. These benefit per ton estimates provide a reduced-form tool for estimating and comparing benefits across multiple mobile source emission scenarios and can be applied to assess the benefits of mobile source policies designed to improve air quality. We found the benefit per ton of directly emitted $PM_{2.5}$ in 2025 ranges from $110,000 for nonroad agriculture sources to $700,000 for onroad light duty gas cars and motorcycles (in 2015 dollars and based on an estimate of PM-related mortality derived from the American Cancer Society cohort study). Benefit per ton values for SO_2/pSO_4 range from $52,000 for aircraft sources (including emissions from ground support vehicles) to $300,000 for onroad light duty diesel emissions. Benefit per ton values for NO_X range from $2100 for C1 and C2 marine vessels to $7500 for “nonroad all other” mobile sources, including industrial, logging, and oil field sources. Benefit per ton estimates increase approximately 2.26-fold when using an alternative concentration response function to derive $PM_{2.5}$-related mortality. We also report benefit per

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ton values for the eastern and western U.S. to account for broad spatial heterogeneity patterns in emissions reductions, population exposure and air quality benefits.

1. Introduction

The transportation sector, which includes on-road vehicles, nonroad vehicles, aircraft, trains, and marine vessels, emits pollutants that degrade air quality (Dallmann and Harley, 2010; Zawacki et al., 2018). The by-products of mobile source combustion processes include direct emissions of particulate matter as well as particulate matter and ozone precursors. Human exposure to fine particulate matter with an aerodynamic diameter smaller than 2.5 μm ($PM_{2.5}$) is associated with increased incidence of premature mortality and morbidity outcomes (Dockery et al., 1993; Pope et al., 2002; Krewski et al., 2009; Lepeule et al., 2012; West et al., 2016; U.S. EPA, 2009a). Characterizing the benefits of improved air quality resulting from reduced or avoided mobile source emissions is an important step in assessing operational procedures (Gouge et al., 2013; Ashok et al., 2017), technology adoption (Tessum et al., 2014), and policies designed to improve air quality (US EPA, 2014).

Full-scale benefits assessments entail detailed and complex analytical steps that characterize each stage of the emissions-to-impact pathway, including quantifying emissions, changes in ambient pollution concentrations and mixing rates, population exposure to pollutants, risks of adverse health outcomes and (often) the economic value of those outcomes. Understanding the effect of emissions on resulting ambient concentrations requires the use of computationally intensive atmospheric chemistry and transport models such as the Comprehensive Air Quality Model with Extensions (CAMx) or the Community Multi-Scale Air Quality model (CMAQ). These models simulate the physical and chemical processes that affect air pollutants and their precursors as they disperse and react in the atmosphere (Byun and Schere, 2006; ENVIRON, 2010). Monetizing the health impacts of changes in pollutant concentrations requires integrated benefits mapping tools that account for population distribution, baseline incidence rates of health endpoints, mortality and morbidity effect estimates, and incidence cost estimates associated with these health endpoints or detailed economic data (Davidson et al., 2007; Saari et al., 2015). The complexity of these models can make full-scale benefits assessments time and resource prohibitive.

Reduced-form approaches can make benefits assessments more tractable by providing computationally efficient techniques that can reasonably and appropriately approximate a full-scale analysis. Benefit per ton (BPT) estimates represent the monetized health benefit of avoiding one ton of emissions from a particular source or source sector. This approach is one example of reduced-form assessment instruments that have been used to characterize the benefits of emission reductions in the US, in Europe, and worldwide (Holland et al., 2005; Fann et al., 2009; Fann et al., 2012; Shindell, 2015; Heo et al., 2016). Benefit per ton estimates are typically generated by running an emissions scenario through a full-scale photochemical air quality model and estimating the environmental health burdens associated with the resulting air pollution. The sum of the monetized impact of these burdens is then divided by the mass of the emissions (or emission changes) associated with that scenario to characterize the marginal benefit of a unit reduction of that emission species (or the marginal cost of an additional unit emission) (Fann et al., 2012).

This study builds upon a detailed source-apportionment air quality study (Zawacki et al., 2018) to present projections of benefits from $PM_{2.5}$ attributable to mobile source emissions across the Contiguous U.S. in 2025. Updating a previous benefits analysis approach presented in Fann et al. (2012), we develop benefit per ton estimates of directly emitted $PM_{2.5}$, $SO_2 + SO_4$, and NO_X for 16 specific mobile source sectors spanning onroad vehicles, nonroad engines and equipment, trains, marine vessels, and aircraft. These self-consistent per-unit-emission benefit estimates provide a reduced-form tool for assessing emission reduction scenarios across multiple mobile source sectors. The benefit per ton estimates presented here improve upon previous estimates, which have been limited to a specific source sector such as aviation (Penn et al., 2017) or have aggregated mobile sources into broader sectoral categories (Fann et al., 2012). This paper describes the approach for calculating species-specific benefit estimates, highlighting advances over previously published benefit per ton estimates. Section 3 presents and summarizes model results while Section 4 discusses important implications and caveats.

2. Methods

To calculate benefit per ton estimates of mobile source emissions, we first modeled $PM_{2.5}$ air quality concentrations in the Contiguous United States using the source apportionment module in the CAMx photochemical air quality model to tag 17 unique mobile-source sectors. Further, this study estimates the extent of premature mortality and morbidity attributable to $PM_{2.5}$, monetizing these impacts using an established model of willingness-to-pay and cost-of-illness values of each health endpoint. Finally, for each sector and each $PM_{2.5}$-related emission species, the resulting monetized benefits are divided by the mass of the emissions to derive a cost-per-unit emission metric. The following section describes the methods and data sources used in these calculations in detail.

Emissions inputs for 2025 are projected from a 2011 emissions inventory generated from EPA's 2011 v6.2 emissions modeling platform, which is based on version 2 of the 2011 National Emissions Inventory (NEI) (US EPA, 2015a). Wildland fires were based on satellite information for location and timing (Baker et al., 2016a) and biogenic emissions were based on day and hour specific temperature and solar radiation (Bash et al., 2016). Mobile source emissions are categorized into 17 sectors based on in-use characteristics, fuel use, and vehicle type and are presented in Table 1.

Aviation emissions are classified as aircraft emissions, which cover commercial aircraft landing and take-off emissions up to 3000 ft, and aircraft ground support emissions at airports. Aircraft emission at altitudes above 3000 ft are not modeled, although there is increasing evidence that high-altitude emissions contribute to local air quality (Barrett et al., 2010). Due to the small amount of aircraft ground support emissions relative to other mobile source categories, ground support and landing and take-off emissions have been combined into one category for the purposes of estimating an aircraft-related benefit per ton value (presented in the next section). However, aviation emissions for these two flight phases are presented separately in Table 1. This explains the discrepancy between having categorized mobile source emissions into 17 sectors while only presenting 16 sector-specific benefit per ton values.

Marine vessel emissions from diesel engines above 800 hp with displacement less than 30 l per cylinder are designated as Category 1 and Category 2 (C1 & C2 marine). Category 3 (C3 marine) emissions come from engines above 30 l per cylinder, typically used for propulsion on ocean-going vessels. For both marine engine categories, emissions out to the U.S. Economic Exclusion Zone are included. We exclude C3 marine emissions that occur in Non-U.S. waters from the benefit per ton calculation since domestic policy will not directly control emissions

Table 1
Projected 2025 emissions (tons) from mobile source sectors.

Sector	Primary $PM_{2.5}$	NO_X	SO_2	pSO_4
Aircraft (excl. ground support)	8074	140,528	16,628	24
Aircraft ground support only	321	10,492	279	2
Marine vessels				
C1 & C2	9068	305,416	795	27
C3[a]	5647	537,038	14,004	2147
Nonroad				
Agriculture	11,310	191,440	340	33
Commercial	6173	74,653	179	36
Construction	12,708	189,821	461	34
Lawn & garden commercial	19,164	54,215	136	14
Lawn & garden residential	5675	19,777	86	3
Recreational (incl. pleasure craft)	13,051	163,443	423	9
All other (industrial, logging, mining, oil field)	4348	98,772	457	236
Onroad				
Heavy duty diesel	30,201	946,522	3748	5329
Heavy duty gas & CNG	1164	30,095	197	31
Light duty diesel	6692	173,650	1291	1612
Light duty gas cars and motorcycles	19,814	219,726	2487	522
Light duty gas trucks	22,274	337,035	4665	564
Rail	13,445	582,351	382	39
Mobile source total	195,548	4,371,692	90,648	10,662
All other sources total[b]	4,324,855	5,855,765	2,665,552	107,604

[a] Excludes emissions outside of the Exclusive Economic Zone (EEZ).
[b] "All other sources" includes emissions from the following sources: biogenics, fugitive dusts, agricultural ammonia, oil and gas exploration, non-Electricity Generating Unit point, Electricity Generating Unit point, non-point, fires (wild, prescribed, agricultural), biomass burning, and international (Canada, Mexico).

outside of U.S. waters. Nonroad emissions are separated into seven categories including commercial, construction, and recreational, while onroad emissions are separated into 5 categories based on vehicle characteristics (light duty and heavy duty) and fuel type (gas, diesel, and compressed natural gas [CNG]).

Mobile source emissions inventories for the entirety of the Contiguous U.S. except California were generated using the Motor Vehicle Emission Simulator (MOVES2014) for onroad emissions (US EPA, 2017a); 2002 and 2008 baseline inventories and sector-specific growth factors for rail and marine emissions respectively; and the National Mobile Inventory Model (NMIM) and NONROAD 2008 model for non-road emissions (US EPA, 2010a; US EPA, 2009b). California-specific emissions were modeled and provided by the state of California as described in Zawacki et al. (2018). Emissions for 2025 were projected from the 2011 inventory, and account for mobile and point source regulations that were final at the time that the platform was finalized. Additional descriptions of the emissions inventories are provided in US EPA (2015b) and Zawacki et al. (2018). Emission estimates for the mobile source sectors include only direct combustion emissions and do not consider upstream (e.g., production) or downstream (e.g., junking) emissions associated with those mobile source sectors.

We tracked the 17 emissions sectors for contributions to surface-level fine particulate concentrations using CAMx v6.2 with PM source apportionment technology (PSAT) extensions (Environ International Corporation, 2015). Photochemical model source apportionment implementations have been used to track specific sources (Baker and Kelly, 2014; Baker et al., 2016b; Baker and Woody, 2017) and groups of sources such as sectors (Fann et al., 2013; Zawacki et al., 2018) at local to continental scales. Limited evaluation comparing photochemical model source apportionment estimates of primary and secondary pollutants against plume measurements made downwind from specific industrial sources show good agreement (Baker and Kelly, 2014; Baker and Woody, 2017). Emissions input files were generated using the Sparse Matrix Operator Kernel Emissions (SMOKE) model v3.6.5. The CAMx model domain resolution consists of 12 km × 12 km grid cells and 25 vertical layers covering the entire Contiguous United States. Initial and boundary conditions including transport from global emissions are taken from GEOS-Chem version 8–03-02 with a 36 km × 36 km lateral resolution, and Input meteorological data are based on a WRF v3.4 simulation (Skamarock et al., 2008).

We applied the environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE) v1.3 to quantify the number and value of $PM_{2.5}$-attributable deaths and illnesses. BenMAP-CE is an open-source computer program that incorporates user-provided or pre-loaded datasets of air quality, demographics, concentration-response relationships, and economic values of mortality and morbidity. BenMAP-CE has been applied at local, regional, and national scales to assess the incidence and value of health impacts attributable to air quality (Fann et al., 2012; Sun et al., 2015; Jones and Berrens, 2017). Modeled air quality concentrations are overlaid on demographically striated population estimates for the emission model year as projected from 2010 U.S. Census block data (Woods and Poole, 2016; US EPA, 2017b). BenMAP-CE relates expected health endpoint incidences over the exposed population of a given geographic region or model grid cell (i) to changes in air quality through a health impact function approach using a concentration-response function (CRF) as in Eq. (1):

$$y_i = y_{0_i}\left(e^{\beta \Delta x_i} - 1\right) \cdot p_i \tag{1}$$

where Δx_i is the change in air quality in grid cell i; β is the risk coefficient of a specific health effect per unit concentration of the pollutant as drawn from relevant epidemiological studies; y_{0i} is the baseline incidence rate of that health endpoint over the population in grid cell i; and p is the exposed population of interest in grid cell i. Total incidences across the model domain can be estimated by summing y across all grid cells.

Two CRFs are applied to estimate premature mortalities attributable to changes in $PM_{2.5}$. The first CRF is drawn from Krewski et al. (2009), an extended follow-up and spatial analysis of the American Cancer Society (ACS) Cancer Prevention Study II (CPS-II) linking particulate air pollution and mortality. We apply the risk coefficient from the random effects Cox model that controls for 44 individual and seven ecological covariates, based on average exposure levels for 1999–2000 across 116 U.S. cities (Relative Risk = 1.06, 95% confidence interval 1.04–1.08). The second CRF is drawn from Lepeule et al. (2012), an extended follow-up of the Harvard Six Cities Study (Relative Risk = 1.14, 95% confidence interval 1.07–1.22). While concentration response functions derived from other studies exist (Lelieveld et al., 2015), the selection of these two landmark studies as the underlying basis for PM mortality benefit estimates provides perspective on the potential range of impacts and is supported by the Advisory Council on Clean Air Compliance Analysis Health Effects Subcommittee (SAB, 2010).

Modeled and monetized non-fatal health endpoints include heart attacks, respiratory and cardiovascular-related hospital admissions, emergency room visits, upper and lower respiratory symptoms, acute bronchitis, aggravated asthma, lost work days, and acute respiratory symptoms. Health impact functions for morbidity endpoints and baseline incidence rates (y_0) for all endpoints are described in detail in the BenMAP-CE documentation and in recent regulatory impact analyses (US EPA, 2017b; US EPA, 2012). Other health impacts such as air quality impact on low birth weight and non-health endpoints such as recreational and residential visibility, impacts on forestry and agriculture, and feedbacks or interactions with climate are not quantified.

The monetized value of avoided mortalities and morbidities associated with reductions in $PM_{2.5}$ exposure are estimated using a combination of willingness-to-pay (WTP) and cost of illness values from literature. The value of statistical life (VSL) approach, a summary measure of the willingness-to-pay for small changes in mortality risk experienced across a large population, is used to monetize avoided premature deaths. While other monetization approaches exist for valuing a reduction in mortality risk (such as the Value of Life Years [VOLY] method) and the VSL approach may not capture total welfare or distributional effects of estimated benefits, the VSL approach is consistent with EPA best-practice and the recommendations of the EPA Science Advisory Board and is supported elsewhere in the benefits assessment literature (US EPA, 2010b; Sunstein, 2013). A single population-uniform VSL, adjusted for inflation and income growth, is developed from willingness-to-pay estimates from a survey of both the stated and revealed preference literature. Results for 2025 emissions are presented in 2015 USD ($) using 2025 income assumptions resulting in a VSL of $10.4 million. The monetized value of $PM_{2.5}$-related mortality also accounts for a twenty-year segmented cessation lag (USEPA, 2012). To discount the value of premature mortality incidence that occurs at future points along the distributed lag, we apply both a 3 and 7 percent discount rate consistent with current regulatory practices and guidance (US EPA, 2010c). While other discounting approaches exist, the selected values give an indication of the sensitivity of benefit per ton estimates to the choice of discount rates.

WTP values are applied for morbidity endpoints where available. For example, the benefits for reduced acute respiratory impacts are monetized by applying a willingness-to-pay to avoid minor-restricted activity days not requiring hospitalization. For health effects without consistent WTP estimates, such as hospital admissions, we monetize the endpoint using the cost of treatment or mitigation. These cost of illness estimates may undervalue total benefits of emission reductions by not including the value of avoided pain and suffering. Morbidity endpoints generally account for less than 5% of total monetized benefits of $PM_{2.5}$ reductions using this approach. The WTP and cost of illness values used in this analysis, their derivation, and a discussion of their uncertainty are provided in detail in the BenMAP-CE documentation and in the benefits assessments presented in US EPA regulatory impact analyses (US EPA, 2017b; US EPA, 2012).

For each sector and emission specie, the monetized value of premature deaths and morbidity endpoints developed above divided by the total associated emissions mass (emission mass by sector and specie is shown in Table 1) yields a series of metrics that reflect the average economic value of avoided health impacts for a 1-ton reduction in directly emitted $PM_{2.5}$ or $PM_{2.5}$ precursors. For example, the monetized value of avoided health impacts associated with directly emitted $PM_{2.5}$ concentrations for a given sector is divided by the mass of primary $PM_{2.5}$ directly emitted from that sector to yield a per unit benefit per ton value. The same calculation is made for each source based on particulate nitrate-related avoided health impacts and NO_X emissions.

To estimate previous mobile source BPT values (Fann et al., 2012), the monetized value of avoided health impacts associated with particulate sulfate were divided by SO_2 emissions, assuming that SO_2 emissions were the primary precursor to the particulate sulfate fraction of total $PM_{2.5}$. This has been the case for most mobile and non-mobile sources in the past. However, as a result of updates to EPA's mobile source emissions estimation tool, the Motor Vehicle Emission Simulator (MOVES), significant changes were made to the modeling of the speciation and composition of sulfur emissions (USEPA, 2015c). These changes particularly increased the percentage of sulfur emissions emitted as particulate SO_4 from model year 2007 and later diesel engines. To account for this update, we now include the mass of both SO_2 emissions and directly emitted SO_4 in the denominator of the mobile source benefit per ton calculations to better account for the SO_4 contribution to the particulate sulfate fraction of total $PM_{2.5}$. While primary nitrate emissions may contribute to a small fraction of total atmospheric particulate nitrate concentrations, only the mass of NO_X emissions are included in the denominator of the nitrate benefit per ton calculation. This is consistent with prior modeling, as the relative contribution of primarily emitted nitrate emissions is expected to be exceedingly small.

Note that these metrics represent the average number of cases and monetized benefits of emission reductions among mobile sources within the Contiguous U.S. As national average values, these metrics may not sufficiently characterize the benefits of emission reductions in a particular location, given local variability in factors such as the density, distribution and baseline health status of populations exposed to $PM_{2.5}$ attributable to each mobile source as well as baseline atmospheric conditions.

Prior studies estimating benefit per ton values (for example, Fann et al., 2012) have not investigated geographic distributions of this reduced order technique. To explore the presence of spatial heterogeneity in the benefit per ton estimates, we estimate "eastern" and "western" regional average benefit per ton values for all mobile source categories. An eastern and western distribution provides improved resolution over these prior, national values. The regional approach uses the same methods described above to generate national average benefit per ton values, except that emissions, air quality concentrations, and monetized health impacts were apportioned to, and summed across, states that fall either in the west or the east. States were assigned to either the eastern region, which includes Texas and all states to the north and east, and the western region, which includes all other states in the Contiguous U.S. (Fig. 1 in the Supplementary materials). We chose this boundary based on the broad spatial characteristics observed in Zawacki et al. (2018) – that for most sources, emissions occur near urban centers and transportation corridors, with corresponding ambient pollutant concentrations. The Midwest, often with lower regional emissions and pollutant concentrations, provides a natural break in the domain. This east-west split, meant to better capture the match between emissions, ambient concentrations of particulate matter, and air quality health impacts on a broad regional basis, is consistent with other regional benefits analyses (US EPA, 2015d; US EPA, 2015e; US EPA, 2012; US EPA, 2011).

3. Results

3.1. National results

The monetized value of health-related benefits associated with reducing a ton of various emission species are shown in Table 2, assuming mortality derived from Krewski et al. (2009), and Table 3, assuming mortality derived from Lepeule et al. (2012). All values in Tables 2 and 3 are presented in year 2015 dollars and assume a 3% discount rate to account for mortality cessation lag. The benefit-per-ton of directly emitted $PM_{2.5}$ ranges from $110,000 for nonroad agriculture emissions and $190,000 for C3 marine vessels to $630,000 for nonroad commercial lawn and garden emissions and $700,000 for onroad light duty gas cars and motorcycles using the CRF for mortalities derived from Krewski et al. (2009). Benefit per ton values for SO_2/pSO_4 range from $29,000 for C3 marine vessels and $52,000 for aircraft emissions to $260,000 for onroad heavy duty diesel emissions and $300,000 for onroad heavy light diesel emissions using the CRF for mortalities derived from Krewski et al. (2009). Benefit per ton values for NO_X range from $1900 for C3 marine vessels and $2100 for C1 and C2 marine vessels to $7100 for onroad light duty gas cars and motorcycles and $7500 for nonroad mobile emissions from other sources including industrial, logging, and oil field using the CRF for mortalities derived from Krewski et al. (2009). Benefit per ton estimates increase approximately 2.26-fold when using the mortality CRF derived from Lepeule et al. (2012). Benefit per ton values using a 7% discount rate are 9–11% lower and are shown in Supplementary materials Tables S1 and S2.

The magnitude of direct $PM_{2.5}$ emission reduction benefits tends to correlate with average proximity of the source sector to locations with the highest population density. For example, direct marine emissions and direct nonroad agriculture emissions tend to have the lowest value per ton as they are, in general, emitted further away from population receptors. Because SO_2 and NO_X contribute to secondary $PM_{2.5}$ formation through physical and chemical properties in the atmosphere, not all SO_2 and NO_X emissions will produce atmospheric particulates and the particulates that do form may be more regionally dispersed. Thus, comparing benefit per ton values associated with these species may reflect broader regional population, background atmospheric

Table 2
Summary of the total dollar value (mortality based on Krewski et al., 2009 and morbidity) of benefits per ton reduction of directly emitted $PM_{2.5}$ and $PM_{2.5}$ precursors by each of 16 sectors in 2025 (2015$; 3% discount rate); results presented as average unit values for the Nation, Western US, and Eastern US.[a]

Benefit per ton values using mortality concentration response function derived from Krewski et al. (2009)									
Sector	Directly emitted $PM_{2.5}$[b]			SO_2/pSO_4[c]			NO_X[d]		
	West	National	East[e]	West	National	East	West	National	East
Aircraft (including ground support)	$570,000	**$440,000**	$390,000	$62,000	**$52,000**	$44,000	$9300	**$7000**	$6100
Marine vessels									
C1 & C2	$740,000	**$200,000**	$140,000	$190,000	**$98,000**	$53,000	$9300	**$2100**	$1500
C3	$540,000	**$410,000**	$370,000	$160,000	**$110,000**	$92,000	$6600	**$2900**	$1700
Non-road									
Agriculture	$110,000	**$110,000**	$110,000	$49,000	**$65,000**	$69,000	$3800	**$3700**	$3600
Commercial	$810,000	**$610,000**	$570,000	$190,000	**$190,000**	$200,000	$7300	**$7000**	$6900
Construction	$620,000	**$460,000**	$420,000	$82,000	**$93,000**	$95,000	$5800	**$5500**	$5400
Lawn & garden commercial	$750,000	**$630,000**	$610,000	$140,000	**$170,000**	$170,000	$5900	**$4800**	$4500
Lawn & garden residential	$660,000	**$590,000**	$580,000	$150,000	**$200,000**	$210,000	$3800	**$4500**	$4600
Recreational (incl. pleasure craft)	$300,000	**$240,000**	$230,000	$43,000	**$59,000**	$62,000	$2700	**$3100**	$3200
All other (industrial, logging, mining, oil field)	$710,000	**$490,000**	$440,000	$300,000	**$210,000**	$200,000	$11,000	**$7500**	$6400
On-road									
Heavy duty diesel	$580,000	**$410,000**	$360,000	$380,000	**$260,000**	$230,000	$5200	**$6100**	$6.500
Heavy duty gas & CNG	$750,000	**$540,000**	$460,000	$140,000	**$140,000**	$140,000	$5200	**$6100**	$6400
Light duty diesel	$750,000	**$480,000**	$380,000	$480,000	**$300,000**	$240,000	$6700	**$5700**	$5400
Light duty gas cars and motorcycles	$1,200,000	**$700,000**	$490,000	$190,000	**$130,000**	$95,000	$7300	**$7100**	$7100
Light duty gas trucks	$970,000	**$590,000**	$450,000	$140,000	**$100,000**	$83,000	$6900	**$6500**	$6300
Rail	$270,000	**$250,000**	$240,000	$57,000	**$77,000**	$89,000	$4100	**$6200**	$6900

Benefit per ton values in bold denote national average values, which are bound by the regional estimates.
[a] Values represent sum of the value of avoided morbidity impacts and mortality impacts quantified using the $PM_{2.5}$ mortality risk estimate noted. Estimates rounded to two significant figures.
[b] Value represents total benefits from reducing elemental and organic carbon exposure divided by the mass of emissions of elemental and organic carbon.
[c] Value represents total benefits from reducing particulate sulfate exposure divided by the mass of emissions of SO_2 and primarily emitted particulate SO_4.
[d] Value represents total benefits from reducing particulate nitrate exposure divided by the mass of emissions of NO_X.
[e] East includes Texas and those states to the north and east. West includes all other states in the Contiguous U.S.

concentrations of reactive species and availability of NH_3, and meteorological conditions proximate to the source sector locations as well as the relative molar masses of the emissions species.

The benefit per ton values of NO_X emissions are lower than the equivalent benefit per ton values of primary PM and SO_2/pSO_4 for each individual source sector; for example, the benefit per ton estimates of C3 marine and nonroad agricultural sulfate emissions are 47 and 18 times greater than the commensurate values for NO_X emissions. However, total health and welfare costs of NO_X emissions may still dominate those of sulfate emissions for a given source sector, as in the case of agricultural where NO_X emissions by mass are over two orders of magnitude greater than sulfate emissions.

Table 3
Summary of the total dollar value (mortality based on Lepeule et al., 2012 and morbidity) of benefits per ton reduction of directly emitted $PM_{2.5}$ and $PM_{2.5}$ precursors by each of 16 sectors in 2025 (2015$; 3% discount rate); results presented as average unit values for the Nation, Western US, and Eastern US.[a]

Sector	Directly emitted $PM_{2.5}$[b]			SO_2/pSO_4[c]			NO_X[d]		
	West	National	East[e]	West	National	East	West	National	East
Aircraft (including ground support)	$1,300,000	**$990,000**	$870,000	$140,000	**$120,000**	$110,000	$21,000	**$16,000**	$14,000
Marine									
C1 & C2	$1,700,000	**$440,000**	$320,000	$420,000	**$220,000**	$120,000	$21,000	**$4700**	$3500
C3	$1,200,000	**$930,000**	$850,000	$370,000	**$250,000**	$210,000	$15,000	**$6600**	$3700
Nonroad									
Agriculture	$260,000	**$240,000**	$240,000	$110,000	**$150,000**	$160,000	$8500	**$8300**	$8300
Commercial	$1,800,000	**$1,400,000**	$1,300,000	$420,000	**$440,000**	$440,000	$16,000	**$16,000**	$16,000
Construction	$1,400,000	**$1,000,000**	$960,000	$180,000	**$210,000**	$220,000	$13,000	**$12,000**	$12,000
Lawn & garden commercial	$1,700,000	**$1,400,000**	$1,400,000	$310,000	**$380,000**	$400,000	$13,000	**$11,000**	$10,000
Lawn & garden residential	$1,500,000	**$1,300,000**	$1,300,000	$330,000	**$450,000**	$470,000	$8600	**$10,000**	$10,000
Recreational (incl. pleasure craft)	$670,000	**$540,000**	$510,000	$98,000	**$130,000**	$140,000	$6200	**$7100**	$7300
All other (industrial, logging, mining, oil field)	$1,600,000	**$1,100,000**	$990,000	$670,000	**$480,000**	$450,000	$25,000	**$17,000**	$14,000
Onroad									
Heavy duty diesel	$1,300,000	**$930,000**	$820,000	$860,000	**$590,000**	$520,000	$12,000	**$14,000**	$15,000
Heavy duty gas & CNG	$1,700,000	**$1,200,000**	$1,00,000	$310,000	**$320,000**	$320,000	$12,000	**$14,000**	$14,000
Light duty diesel	$1,700,000	**$1,100,000**	$860,000	$1,100,000	**$680,000**	$550,000	$15,000	**$13,000**	$12,000
Light duty gas cars and motorcycles	$2,700,000	**$1,600,000**	$1,100,000	$440,000	**$290,000**	$220,000	$17,000	**$16,000**	$16,000
Light duty gas trucks	$2,200,000	**$1,300,000**	$1,000,000	$310,000	**$230,000**	$190,000	$16,000	**$15,000**	$14,000
Rail	$610,000	**$560,000**	$540,000	$130,000	**$170,000**	$200,000	$9200	**$14,000**	$16,000

Benefit per ton values in bold denote national average values, which are bound by the regional estimates.
[a] Values represent sum of the value of avoided morbidity impacts and mortality impacts quantified using the $PM_{2.5}$ mortality risk estimate noted. Estimates rounded to two significant figures.
[b] Value represents total benefits from reducing elemental and organic carbon exposure divided by the mass of emissions of elemental and organic carbon.
[c] Value represents total benefits from reducing particulate sulfate exposure divided by the mass of emissions of SO_2 and primarily emitted particulate SO_4.
[d] Value represents total benefits from reducing particulate nitrate exposure divided by the mass of emissions of NO_X.
[e] East includes Texas and those states to the north and east. West includes all other states in the Contiguous U.S.

Table 4
Population-weighted average concentration contribution (μg/m^3) by source for modeled $PM_{2.5}$ species[a].

Sector	Primary $PM_{2.5}$			SO_2/pSO_4			NO_X		
	West	National	East	West	National	East	West	National	East
Aircraft (including ground support)	0.0403	**0.0236**	0.0184	0.0090	**0.0059**	0.0049	0.0132	**0.0070**	0.0051
Marine									
C1 & C2	0.0172	**0.0111**	0.0093	0.0016	**0.0005**	0.0002	0.0064	**0.0041**	0.0033
C3	0.0026	**0.0017**	0.0015	0.0218	**0.0117**	0.0085	0.0304	**0.0111**	0.0052
Nonroad									
Agriculture	0.0048	**0.0065**	0.0070	0.0001	**0.0001**	0.0002	0.0035	**0.0041**	0.0044
Commercial	0.0230	**0.0190**	0.0178	0.0002	**0.0003**	0.0003	0.0033	**0.0032**	0.0032
Construction	0.0430	**0.0350**	0.0325	0.0002	**0.0003**	0.0003	0.0077	**0.0065**	0.0062
Lawn & garden commercial	0.0464	**0.0491**	0.0499	0.0001	**0.0002**	0.0002	0.0023	**0.0017**	0.0015
Lawn & garden residential	0.0094	**0.0128**	0.0138	0.0000	**0.0001**	0.0001	0.0003	**0.0005**	0.0006
Recreational (incl. pleasure craft)	0.0150	**0.0119**	0.0110	0.0001	**0.0002**	0.0002	0.0021	**0.0031**	0.0034
All other (industrial, logging, mining, oil field)	0.0131	**0.0096**	0.0085	0.0010	**0.0009**	0.0009	0.0090	**0.0048**	0.0035
Onroad									
Heavy duty diesel	0.0766	**0.0493**	0.0409	0.0242	**0.0158**	0.0131	0.0357	**0.0357**	0.0357
Heavy duty gas & CNG	0.0042	**0.0023**	0.0018	0.0003	**0.0002**	0.0002	0.0013	**0.0011**	0.0011
Light duty diesel	0.0236	**0.0116**	0.0079	0.0111	**0.0060**	0.0044	0.0094	**0.0063**	0.0054
Light duty gas cars and motorcycles	0.1180	**0.0496**	0.0285	0.0079	**0.0035**	0.0021	0.0115	**0.0097**	0.0091
Light duty gas trucks	0.1030	**0.0483**	0.0285	0.0076	**0.0036**	0.0024	0.0204	**0.0138**	0.0118
Rail	0.0293	**0.0198**	0.0168	0.0003	**0.0002**	0.0002	0.0210	**0.0215**	0.0217

Population-weighted concentrations in bold denote national average values, which are bound by the regional estimates.
[a] Population ages 0–99 used for weighting.

3.2. Regional results

Because the relationship between emissions reductions and human exposure to atmospheric pollution is dependent on background atmospheric composition, meteorological conditions, and proximity to population sources, the marginal benefits of reducing emissions show significant spatial heterogeneity (Fann et al., 2009; Heo et al., 2016). Looking at the regional results, we find that an Eastern and Western split, meant to better match emissions reductions and air quality benefits, captures some of this variability on a broad, regional basis. As we observed in the national average benefit per ton estimates, the magnitude of the regional estimates tends to correlate with average proximity of the source sector to locations with the highest population density. For example, for onroad sources of pollution, western benefit per ton values are generally larger than eastern values, consistent with the density of roads, vehicle traffic, and population centers in the West. This pattern is borne out by each source's population-weighted air quality contribution for the different components of PM modeled by Zawacki et al. (2018). Table 4 presents average regional contribution in air quality by source, taken from Zawacki et al. (2018), and weighted by population.

Other notable regional trends include larger average per-unit marine and aviation benefits per ton in the West, in part due to the proximity of large population centers to emissions sources. However, NOx-related benefit per ton estimates maintain relative consistency across regions and the national average benefit per ton value, indicating a more uniform distribution of NOx-related air quality impacts throughout the U.S.

3.3. Comparisons to other values

Due to resolution and data limits, previous assessments of the public health benefits of primary $PM_{2.5}$ and precursor emission reductions have presented mobile source values as coarse sectoral aggregates (Fann et al., 2012; Heo et al., 2016). Fig. 1 shows the benefit per ton values for 2025 emissions for three combined source sectors and their components.[1] The source-specific values show the significant variability in benefit per ton estimates across different combined sectoral categories. For example, the combined sectoral nonroad mobile benefit per ton values overestimates agriculture-specific benefit per ton values for all emission species and underestimate nonroad commercial and nonroad other benefits. While onroad mobile emissions show less sector-by-sector variability in primary $PM_{2.5}$ benefit per ton estimates compared to other combined sources, onroad sulfate benefit per ton values underestimate diesel-specific values and overestimate gas vehicle-specific estimates each by up to a factor of two.

For comparison, the combined sector benefit per ton values for the year 2016 from Fann et al. (2012) are also shown in Fig. 1. Though the source-apportionment modeling approach and the specification of the health impact and benefits assessment in Fann et al. (2012) is similar to that used here, it is challenging to directly infer trends across results as differences in underlying emission factors, modeling platform, and policies included in the emissions projections may confound comparisons. The benefit per ton estimates for primary $PM_{2.5}$ show consistent increasing year-on-year trends across all combined source sectors. The systematic growth in the benefit per ton estimates are driven by U.S. population growth and increases in personal income leading to increases in the willingness to pay to reduce the risk of premature death from exposure to air pollution. However, comparisons to prior modeling of benefits associated with secondary particulate species are more complicated as they are more dependent on chemical speciation and background emissions. Benefit-per-ton values for 2025 NO_X emissions developed in this work are lower than the associated 2016 combined sector estimates presented in Fann et al. (2012).

4. Discussion and conclusions

The benefit per ton values reported here express the economic value and human health impacts attributable to emissions for specific components of the mobile sector. These benefit per ton values are the most detailed representation of the mobile sector reported in the literature thus far. However, care must be taken when using or interpreting these values in a policy analysis context. The numerator of the benefit per ton value derivation equation (i.e. benefits) includes only the benefits that accrue to the exposed population within the CAMx modeling grid from only the health endpoints and pollutants expressly modeled, while the denominator (i.e., emission tons) includes only the emissions modeled and released within the modeling domain. For example, C3 marine emissions in the benefit per ton calculation include all ocean going marine vessels out to the bounds of the U.S. Exclusive Economic Zone (EEZ). Thus, the benefit per ton value would be expected to

[1] C3 marine vessels are considered ocean-going vessels and have typically been presented separately from the Aircraft, Locomotive (Rail), and Marine (ALM) combined sector. Aircraft emissions were not included in the ALM estimates from Fann et al. (2012).

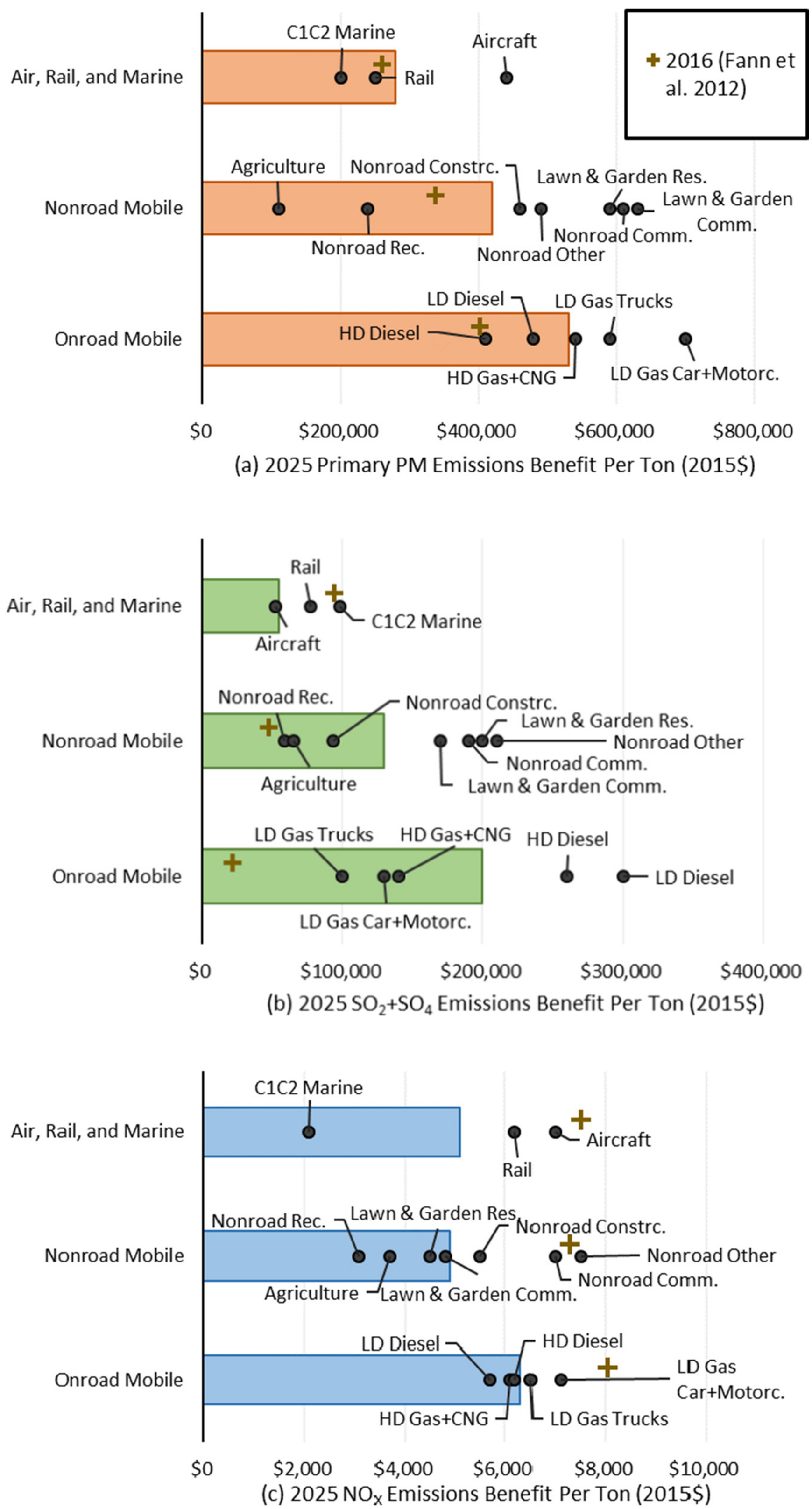

Fig. 1. Combined and individual source sector benefit per ton values for (a) primary $PM_{2.5}$, (b) SO_2 and SO_4, and (c) NO_X. Benefit per ton values for each specific mobile source sector represented by black dots. Colored bars represent average benefit per ton values for aggregated sector categories, which were defined to be consistent with the mobile source benefit per ton values published by Fann et al. (2012). For comparison, the Fann et al. combined mobile source sector benefit per ton values are represented by plus symbols. All values use Krewski et al. (2009) mortality estimates, a 3% discount rate, and are valued in 2015$. Note that the Fann et al. values were projected from the 2005 v4 modeling platform and assessed using BenMAP v4.044.

underestimate benefits if applied to a policy that focused on coastal emission reductions. Conversely, aircraft emissions in the benefit per ton calculation include only emissions below 3000 ft; this benefit per ton value could overestimate benefits if applied to emission reductions occurring across all phases of flight. The geographically differentiated benefit per ton values presented here significantly improve upon previously presented geographically uniform values, while also providing insight into some of the limitations of using benefit per ton values in benefits assessments. Future work could investigate optimal resolutions or boundaries for policy.

Across several sectors, additional benefits (and potential disbenefits) from emission reductions of the modeled species could include induced changes in surface ozone concentrations (Turner et al., 2016), acidic deposition (Menz and Seip, 2004), additional morbidity costs or health endpoints not currently modeled, broader economic feedbacks (Saari et al., 2015), interactions with climate change, and benefits to populations outside the modeling domain from intercontinental transport (Chin et al., 2007; Barrett et al., 2010). Benefit per ton estimates are not calculated for ammonia (NH_3) emissions, which may be important for automobile-induced secondary particulate matter formation and may have tradeoffs associated with reducing NO_X emissions (Dedoussi and Barrett, 2014). Further, while benefit per ton estimates indicate which emissions are most beneficial per unit reduction, they do not give an indication of which emission sectors and sources are the least costly to target to protect public health, which is critical for characterizing regulatory impacts (Dominici et al., 2014).

Uncertainties and limitations exist across the entire emissions-to-impact pathway. Inventories for nonroad emission sources, including rail and marine, are less certain than inventories for onroad sources, resulting in heterogeneous uncertainties in benefit per ton estimates across source sectors (Zawacki et al., 2018). Uncertainties associated with the health impact functions and the VSL can have a large influence on the magnitude of benefits, but previous work has found that their variation is comparable to other modeling assumptions in assessing the policy-to-impact chain (Thompson et al., 2014). Health impact function and VSL uncertainties are systematic across sectors and pollutants (Thompson et al., 2014; Shindell, 2015), but differential toxicity across fine particle size and composition may induce heterogeneous uncertainty across pollutant species. However, there currently is insufficient scientific evidence to differentiate the concentration response function for each health endpoint by emission species. Model evaluation for the projected emission year is performed in Zawacki et al. (2018). Detailed uncertainty evaluation of air quality modeling and benefits assessment are included as part of regulatory impact assessments and can inform the contributors to and magnitude of benefit per ton value uncertainties (US EPA, 2012; US EPA, 2015e).

Notwithstanding these uncertainties, the benefit per ton values presented here represent well the number and economic value of adverse health impacts associated with emissions from a broad class of mobile sources. As compared to values reported elsewhere in the literature, these benefit per ton values are better resolved by mobile sector and by geographic area—two features that make these especially useful to quantifying the benefits of reducing emissions from this sector.

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Disclaimer

The views expressed in this article are those of the authors and do not necessarily represent the views or the policies of the U.S. Environmental Protection Agency.

Appendix A. Supplementary data

Additional information about estimated benefits per ton is provided as the Supplementary materials. Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.09.273.

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