Environmental Science & Technology

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An Episodic Assessment of Vehicle Emission Regulations on Saving Lives in California

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Cite This: *Environ. Sci. Technol.* 2021, 55, 547−552

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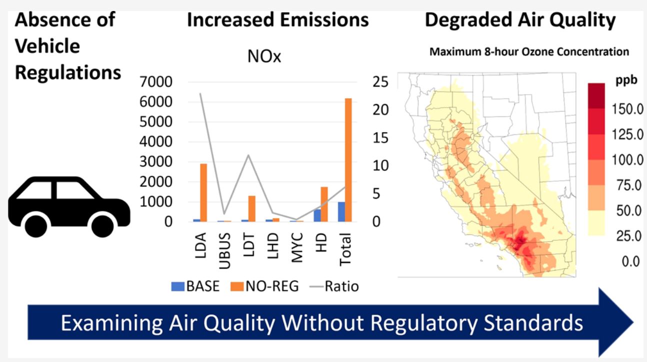
ABSTRACT: Historically, California has been a world leader in the development and application of environmental regulations. Policies to address air pollution have reduced criteria pollutant emissions, improved regional air quality, and benefited public health. To this end, California has imposed strict regulations on light-duty, medium-duty, and heavy-duty vehicles to reduce ambient concentrations of health-damaging pollutants such as ozone and fine particulate matter ($PM_{2.5}$). Here, we compare the impact on air quality in California should California not have adopted on-road vehicle regulations (No Regulations Case) with the air quality associated with current regulations (Regulated Case). Simulations of atmospheric chemistry and transport are conducted to evaluate the impact of emissions on ambient levels of ozone and $PM_{2.5}$, and a health impact assessment tool is used to quantify and monetize societal impairment. Compared with the "Regulated Case," the "No Regulations Case" results in a maximum peak 8 h ozone level of 162 ppb and 24 h $PM_{2.5}$ of 42.7 $\mu g/m^3$ in summer, and 107 $\mu g/m^3$ and 24 h $PM_{2.5}$ in winter. The associated increases in the daily incidence of human health outcomes are \$66 million per day and \$116 million per day during peak pollutant formation periods in summer and winter, respectively. Overall, the findings quantitatively establish the role and importance of on-road vehicle regulations in protecting societal well-being.



1. INTRODUCTION

The unexplained occurrence of eye- and lung-irritating atmospheric photochemical oxidants in Los Angeles in the 1940s was eventually determined to be predominantly associated with emissions from vehicles powered by combustion engines.[1] Combustion produces exhaust composed primarily of nitrogen, water, and carbon dioxide. The exhaust also has a small amount, virtually negligible on a mass basis, of other species that once emitted into the atmosphere result in measurable health effects, either directly (e.g., oxides of nitrogen and some forms of particulate matter) or through the generation of secondary pollutants in the atmosphere (e.g., ozone and other forms of particulate matter).

As a transformative event in environmental protection,[2] California began to aggressively regulate motor vehicle tailpipe emissions in 1966.[3] These efforts were followed by the Federal government developing the groundbreaking Clean Air Act of 1970, which established the formation of national ambient air quality standards (NAAQS).[4] In the subsequent years, California achieved historic success in reducing pollutant emissions to achieve air quality improvements, including reductions in concentrations of photochemical smog (in particular, ground-level ozone) and particulate matter (PM),[5,6] through regulatory controls, technological advancements, and improvements in energy efficiency.[7] In particular, these air quality improvements are attributed to reductions in mobile source emissions from on-road light-duty, medium-duty, and heavy-duty vehicles (LDV, MDV, and HDV).[8−10] The achievement is remarkable given the growth in California's population, energy demands, and economy.[11] Here, we use a photochemical air quality model and a health impact assessment tool to address the question "What might the costs to human health from air pollution be in California if on-road vehicles were not regulated?"

Regulatory efforts have had a profound impact on reducing air pollutant emissions and improving the safety of vehicles, and current vehicles emit 1−10% of pollutants that vehicles preceding control efforts had emitted.[12] Various methods have been used to achieve emission reductions, including pollutant control technologies (e.g., SCR for diesel heavy-duty vehicles),[13] reformulation of fuels,[14,15] periodic vehicle

Received: June 23, 2020
Revised: November 3, 2020
Accepted: November 6, 2020
Published: December 9, 2020

inspection and maintenance programs,[16] and the deployment of alternative zero-emitting technologies (e.g., electric vehicles and hydrogen fuel cell vehicles).[17] The phase-in of increasingly stringent emission limits has led to the successful implementation of control technologies—including the catalytic converter in 1975 and the three-way catalyst in 1981—which remain pivotal control technologies for on-road vehicles.[18] Additional examples include thermal management and onboard diagnostic systems, advanced catalyst technologies, and a range of engine control systems (e.g., exhaust gas recirculation and improved fuel injectors).[19] The success of vehicle regulations is also evident by the divergent trends of pollution and activity, with emission reductions coinciding with increases in total sector demands including the number of vehicles and vehicle miles traveled.[20]

Despite the historical success of air quality regulations, many California regions still experience levels of pollution in excess of the NAAQS, including the highly populated South Coast Air Basin (SoCAB) of California and the Central Valley.[21] In particular, exposure to fine particular matter ($PM_{2.5}$), nitrogen dioxide, and ground-level ozone causes an elevated risk of morbidity and premature death.[22−26] The health benefits achieved through meeting NAAQS have a significant monetary value to society.[27] For example, control programs executed in response to the 1990 Clean Air Act Amendments yielded air quality improvements with a value of potentially $2 trillion in total from 1990 to 2020.[28] In California, the economic savings from meeting NAAQS have been estimated at $22 and $6 billion (2007 dollars) for residents of the SoCAB and Central Valley, respectively.[29,30] On-road vehicle emission regulations incur monetary value to California through the avoidance of adverse health effects from cleaner air. Here, we attempt to quantify these benefits by assessing and valuing the air quality and health impacts of unregulated vehicles relative to the currently regulated vehicles.

The air quality impacts of potentially strengthening vehicle emission standards in SoCAB have been evaluated using atmospheric modeling.[31] Similarly, several studies have evaluated the impacts associated with current and future California and U.S. vehicle emission regulations.[32−34] However, these works generally consider the incremental reduction in emissions that future vehicles may have from current vehicles—which are already relatively clean. The current work is distinguished from these efforts by demonstrating the fundamental change associated with nonregulated vehicles relative to the currently regulated vehicles as well as including all on-road vehicles in the LDV, MDV, and HDV sectors where LDV includes light-duty automobiles (LDA) and light-duty trucks (LDT). The goal is to quantify the human health benefits that California's on-road vehicle regulatory standards have achieved (and conversely, the consequences of non-regulation) during a high pollutant formation period by demonstrating and valuing the effects of air quality degradation in the absence of on-road vehicle regulation. The results provide an important contribution by demonstrating the significant monetary value to the society that vehicle emission reductions provide, which may not be as well understood compared to the increased cost of emission control strategies.

2. METHODS

2.1. Emission Modeling. To estimate on-road vehicle emissions, we use the mobile source EMission FACtors (EMFAC2017)[35] model established by the California Air Resources Board to develop a baseline (i.e., "Regulated Case") and an adjusted (i.e., "No Regulations Case") inventory. EMFAC2017 calculates statewide on-road emissions for all vehicle categories (LDV, MDV, and HDV) based on fleet composition, vehicle activity data, and other factors including aging effects on vehicles. For the "No Regulations Case," model years (MY) for all vehicle types are replaced by the oldest MY available in the database to integrate vehicles representative of those prior to regulations being imposed (i.e., no control) or vehicles subjected to the oldest regulatory constraints (i.e., vehicles with less control than the current ones). Vehicle emissions are subjected to all other elements of EMFAC2017, including aging effects. Table S1 contains the MY assumed for each vehicle category, for example, MY 1968 is selected for all LDV and most HDV. Figure 1 shows both

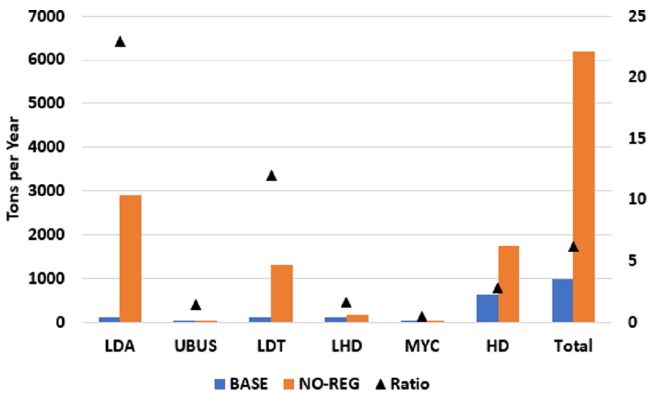


Figure 1. Total NO_x emissions (left axis) for different vehicle types and the ratio (right axis) of emissions in the "Regulated Case" and "No Regulations Case." LDA: Light-Duty Auto, UBUS: Urban Bus, LDT: Light-Duty Truck, LHD: Light−Heavy-Duty Truck, MYC: Motorcycle, HD: Heavy-Duty Truck.

the total amount and ratio of NO_x emissions between the "No Regulations" and "Regulated Cases," with LDA experiencing the largest increase exceeding 20 times the baseline. The same is true for VOC emissions (Figure S1); however, PM increases are largest for HDV (Figure S2). Emission changes estimated using EMFAC are reasonable with historical emission levels reported in the literature for both LDV (Table S2), HD trucks (Table S3), and buses (Table S4). It should be noted that only the direct vehicle emissions are adjusted, including exhaust, evaporative, and brake/tire wear. All other emissions from transportation, including the production and distribution of petroleum fuels, are the same as the "Regulated Case."

2.2. Air Quality Modeling. On-road inventories are merged into the CARB 2012 emissions inventory[36] to account for all other sources of emissions, and the Sparse Matrix Operator Kernel Emission (version 4.0) model[37] is used to generate anthropogenic emissions fields. The SAPRC-07 chemical mechanism[38] is used for speciation. The files are merged with biogenic emissions obtained from the Model of Emissions of Gases and Aerosols from Nature (version 2.1).[39]

The Community Multiscale Air Quality model (CMAQ) version 5.2[40] is used to simulate the chemistry and transport to determine the final ground-level concentrations of ozone and $PM_{2.5}$ during high formation periods in California. CMAQ is a widely accepted model for NAAQS attainment demonstration[41] and research purposes involving atmospheric chemistry processes.[42,43] The SAPRC-07 chemical mechanism[38] is used

https://dx.doi.org/10.1021/acs.est.0c04060
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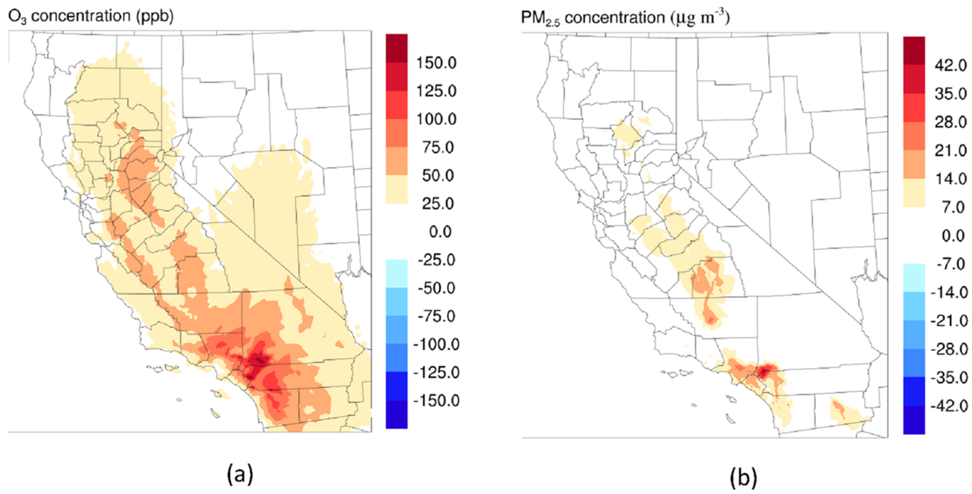

Figure 2. Peak differences in (a) maximum 8 h average ozone and (b) 24 h average $PM_{2.5}$ between the "No Regulations" and "Regulated Case" for the summer period.

for gas phase chemistry, and the AERO6 module[44] is used for aerosol dynamics with the latest SOA module.[45] The modeling domain covers California at a 4 km × 4 km resolution horizontal grid as shown in Figure S3 with two subdomains of special interest indicated for the SoCAB and Central Valley as they contain the most severe designated nonattainment areas for ozone and $PM_{2.5}$.[46] The initial and boundary conditions are generated from the Model for Ozone and Related Chemical Tracers (Mozart v4.0).[47] Meteorological inputs are downscaled from the (Final) Operational Global Analysis data (NCEP, 2000) for the year 2012 using the Advanced Research Weather Research and Forecasting Model (WRF-ARW, version 3.7).[48] The simulation period ranged from July 8−22 for the summer episode and January 1−15 for the winter episode with the first 4 days used as the spin-up period. The model performance for the baseline case has been evaluated and verified in previous studies.[49,50]

2.3. Health Impact Assessment. Following the study by Shen et al.,[51] we used a health impact assessment tool developed and maintained by the U.S. Environmental Protection Agency (EPA), the environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE),[52,53] to quantify and assess the monetary value of health impacts occurring across the California population from ozone and $PM_{2.5}$ concentration changes. We used population statistics from the Landscan data for 2012.[54] Baseline incidence rates for mortality and morbidity at the county level by five-year age groups are obtained from a comprehensive review of the literature.[55] Similarly, concentration-response (C-R) functions are selected based on suggested criteria from a thorough review of the literature.[56] Valuation functions for both morbidity and mortality incidence are similarly selected.[57] While these sources identify values appropriate for the four-county region of SoCAB, the results are suitable for application to the entire California state population as the available literature lacks data with improved granularity (e.g., specific C-R functions for northern or central California populations). It should also be noted that while BenMAP-CE can be used to estimate health impacts from long-term exposure such as those occurring from annual average $PM_{2.5}$, we report results for short-term exposure to ozone and $PM_{2.5}$ only as appropriate for the modeled episode (i.e., avoided health incidence and dollars per day). As a result, our estimates are conservative in that the C-R and valuation functions for short-term exposure yield significantly lower values than those for long-term exposure. For example, in Shen et al.'s study, the monetized mortality-related public health benefits from long-term $PM_{2.5}$ exposure are 14 times higher than those from the short-term one.[51] Similar magnitudes for the valuation of long-term exposure have been reported by others.[28]

3. RESULTS AND DISCUSSION

Increased vehicle emissions in the "No Regulations Case" increase the concentrations of maximum 8 h average summer ozone (up to 162 ppb) and 24 h average $PM_{2.5}$ (up to 42.7 μg/m^3) in California, particularly in the SoCAB with ozone levels exceeding 300 ppb (Figure 2). In the winter period, $PM_{2.5}$ impacts are pronounced in the Central Valley, with increases reaching 107 μg/m^3 (Figure S4). Increases in the number of NAAQS exceedances for both ozone and $PM_{2.5}$ are predicted as a result of concentration changes (Figures S5 and S6). This is important for both SoCAB and the Central Valley as both areas currently fail to achieve NAAQS for ozone and $PM_{2.5}$ in some areas. Maximum ground-level impacts occur in densely populated urban regions, notably the SoCAB, with heightened importance attributed to potential health impacts. The concentration increases predicted here are dramatic but must be considered conservative as (1) the emission forecasting method likely underestimates emission increases from MDV, HDV, and buses, and (2) emissions associated with petroleum fuel production and distribution are not considered and would likely impact results significantly. For example, 1 h average carbon monoxide (CO) concentrations reach 19 ppm in the "No Regulations Case," which is likely an underestimation as the 1960s average CO levels regularly reached 30−45 ppm.[58] As CO levels in the basin largely result from vehicle emissions, scaling with vehicle miles traveled (VMT) from the 1960s to today's yields expected average concentrations ranging from 87−130 ppm, although other factors must also be considered including variations in meteorology.[59]

Table 1 shows that air quality degradation in the "No Regulations Case" increases the incidence of deleterious short-term health impacts (Tables S5 and S7) corresponding to a mean cost of $66 million per day in summer and $116 million per day in winter. By far, the largest impacts occur from increased incidence of premature mortality in both summer

https://dx.doi.org/10.1021/acs.est.0c04060
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Table 1. Mean Incidence and Valuation of Increased Morbidity and Mortality Health Outcomes in the "No Regulations Case" Relative to Base[a]

	summer episode		winter episode	
end point	mean incidence (# /day)	mean valuation (thousand $/day)	mean incidence (# /day)	mean valuation (thousand $/day)
premature mortality, all cause				
short-term ozone exposure	3.7	−32,100.0	−0.3	2298.0
short-term $PM_{2.5}$ exposure	3.3	−28,800.0	12.6	−110,000.0
mortality total	7	−60,900.0	12.3	−107,702.0
morbidity, all				
short-term ozone exposure	35248.9	−2941.0	−2124.3	133.8
short-term $PM_{2.5}$ exposure	20227.2	−2173.9	89295.8	−8745.3
morbidity total	55476.1	−5114.9	87171.5	−8611.5
case total		−66,014.9		−116,313.5

[a]Negative values denote cost from increased health impacts.

($60.9 million mean) and winter ($107.7 million mean) calculated from the value of statistical life obtained from Robinson et al.'s study.[57] In contrast, health costs from increased morbidity incidence are approximately 1−2 orders of magnitude lower, depending on the season ($5.1 million summer mean and $8.6 million winter mean). Impacts are notably higher for the winter modeling period due to higher $PM_{2.5}$ exposure, demonstrating the importance of $PM_{2.5}$ for human health. The results show a "positive" value (i.e., benefits) for short-term ozone exposure in the winter corresponding to concentration reductions from titration reactions driving inverse relationships between ozone and precursor species in some regions of California including SoCAB.[43,49] It should also be considered that the average concentration of ozone in winter is significantly below the NAAQS, further limiting the importance of increases. Still, the results represent a net cost in winter due to the much higher health impact cost from $PM_{2.5}$ increases. In summer, the health cost impacts are relatively equivalent between exposure to ozone and $PM_{2.5}$.

Figure S9 shows the daily monetized health costs occurring from increases in ozone and $PM_{2.5}$ in the summer episode. It is extremely important to consider that these results are from short-term exposure estimates only and therefore highly conservative. The use of long-term exposure functions would be expected to provide approximately an order of magnitude higher savings annually. The highest costs are associated with areas of the SoCAB due to the extremely high vehicle populations and other aggravating features (i.e., geography and weather patterns). In particular, the highest combined benefits occur in the western portions of SoCAB due to the increased ozone concentrations. Prominent areas of added costs are shown in the Central Valley (Fresno and Bakersfield), with lesser impacts in the San Francisco Bay Area and Sacramento. Figure S8 shows the winter benefits demonstrating similar spatial patterns although peak benefits also occur in the Central Valley, in addition to SoCAB, as a result of significant $PM_{2.5}$ worsening. Of note, these areas often coincide with environmentally disadvantaged communities (Figure S7), particularly in the SoCAB (Figure S8).

4. SUMMARY

While many stakeholders fought the initial imposition and subsequent tightening of tailpipe emission regulations, the results reported herein suggest that not regulating combustion-powered vehicles would have led to severe health impacts. In the "No Regulations" case, we conservatively estimate the monetized economic damages associated with human health outcomes the absence of regulation would result today, that is, an increase in societal costs of $66 million per day and $116 million per day associated with short-term health effects of peak pollutant formation periods. Long-term exposure would only add to the public health impact.

While the impacts are evaluated for California due to the data available, the progressive regulatory efforts to reduce tailpipe emissions, and the model followed by the rest of the country, the results reflect the nation as a whole with the understanding that the level of the effects will vary based on local meteorology, geography, and spatial distribution of population.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.0c04060.

> Vehicle model year and emission information (Tables S1−S3), detailed BenMAP results for each endpoint with uncertainty ranges (Tables S5−S8), emission differences for different vehicle types (Figures S1,S2), simulation domain (Figure S3), winter ozone and $PM_{2.5}$ concentration differences (Figure S4), number of nonattainment days for ozone and $PM_{2.5}$(Figures S5,S6), census tract rankings for environmental burdens in California and SoCAB (Figures S7,S8), and monetized health benefits for summer and winter episode (Figure S9,S10) (PDF)

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https://dx.doi.org/10.1021/acs.est.0c04060
Environ. Sci. Technol. 2021, 55, 547−552

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https://pubs.acs.org/10.1021/acs.est.0c04060

Notes

The authors declare no competing financial interest.

■ ACKNOWLEDGMENTS

The authors would like to thank Victor Vila and Pau Balat for their contributions to this work, which was supported by the Balsells Fellowship. The authors would also like to acknowledge the contributions of the staff of the high-performance computing cluster at UCI for this work, including Joseph Farran and Harry Mangalam.

■ REFERENCES

(1) Haagen-Smit, A. J. Chemistry and Physiology of Los Angeles Smog. *Ind. Eng. Chem.* **1952**, *44*, 1342−1346.
(2) Bachmann, J. Will the Circle Be Unbroken: A History of the US National Ambient Air Quality Standards. *J. Air Waste Manage. Assoc.* **2012**, *57*, 652−697.
(3) California Air Resources Board. History https://ww2.arb.ca.gov/about/history.
(4) Clean Air Act Extension of 1970; 84 Stat. 1676, P.L. 91−604, 1970-12-31.
(5) Lurmann, F.; Avol, E.; Gilliland, F. Emissions Reduction Policies and Recent Trends in Southern California's Ambient Air Quality. *J. Air Waste Manage. Assoc.* **2015**, *65*, 324−335.
(6) Pollack, I. B.; Ryerson, T. B.; Trainer, M.; Neuman, J. A.; Roberts, J. M.; Parrish, D. D. Trends in Ozone, Its Precursors, and Related Secondary Oxidation Products in Los Angeles, California: A Synthesis of Measurements from 1960 to 2010. *J. Geophys. Res. Atmos.* **2013**, *118*, 5893−5911.
(7) Allison, J.; Press, D.; Horowitz, C.; Millard-Ball, A.; Pincetl, S. Chapter 7. Paths to Carbon Neutrality: Lessons from California. *In Bending the Curve: Ten scalable solutions for carbon neutrality and climate stability*; University of California Press, 2016; 2.
(8) Parrish, D. D.; Xu, J.; Croes, B.; Shao, M. Air Quality Improvement in Los Angeles—Perspectives for Developing Cities. *Front. Environ. Sci. Eng.* **2016**, *10*, 11.
(9) Hassler, B.; McDonald, B. C.; Frost, G. J.; Borbon, A.; Carslaw, D. C.; Civerolo, K.; Granier, C.; Monks, P. S.; Monks, S.; Parrish, D. D. Analysis of Long-term Observations of NOx and CO in Megacities and Application to Constraining Emissions Inventories. *Geophys. Res. Lett.* **2016**, *43*, 9920−9930.
(10) Bahadur, R.; Feng, Y.; Russell, L. M.; Ramanathan, V. Impact of California's Air Pollution Laws on Black Carbon and Their Implications for Direct Radiative Forcing. *Atmos. Environ.* **2011**, *45*, 1162−1167.
(11) Propper, R.; Wong, P.; Bui, S.; Austin, J.; Vance, W.; Alvarado, A.; Croes, B.; Luo, D. Ambient and Emission Trends of Toxic Air Contaminants in California. *Environ. Sci. Technol.* **2015**, *49*, 11329−11339.
(12) Sperling, D. The Price of Regulation. *ACCESS Mag.* **2004**, *1*, 9−18.
(13) Faiz, A.; Weaver, C. S.; Walsh, M. P. *Air Pollution from Motor Vehicles: Standards and Technologies for Controlling Emissions*; World Bank Publications, 1996.
(14) Kirchstetter, T. W.; Singer, B. C.; Harley, R. A.; Kendall, G. R.; Traverse, M. Impact of California Reformulated Gasoline on Motor Vehicle Emissions. 1. Mass Emission Rates. *Environ. Sci. Technol.* **1999**, *33*, 318−328.
(15) Harley, R. A.; Hooper, D. S.; Kean, A. J.; Kirchstetter, T. W.; Hesson, J. M.; Balberan, N. T.; Stevenson, E. D.; Kendall, G. R. Effects of Reformulated Gasoline and Motor Vehicle Fleet Turnover on Emissions and Ambient Concentrations of Benzene. *Environ. Sci. Technol.* **2006**, *40*, 5084−5088.
(16) Eisinger, D. S.; Wathern, P. Policy Evolution and Clean Air: The Case of US Motor Vehicle Inspection and Maintenance. *Transp. Res. Part D Transp. Environ.* **2008**, *13*, 359−368.
(17) Calef, D.; Goble, R. The Allure of Technology: How France and California Promoted Electric and Hybrid Vehicles to Reduce Urban Air Pollution. *Policy Sci.* **2007**, *40*, 1−34.
(18) Gerard, D.; Lave, L. B. Implementing Technology-Forcing Policies: The 1970 Clean Air Act Amendments and the Introduction of Advanced Automotive Emissions Controls in the United States. *Technol. Forecast. Soc. Change* **2005**, *72*, 761−778.
(19) Bertelsen, B. I. Future US Motor Vehicle Emission Standards and the Role of Advanced Emission Control Technology in Meeting Those Standards. *Top. Catal.* **2001**, *16/17*, 15−22.
(20) Parrish, D. D. Critical Evaluation of US On-Road Vehicle Emission Inventories. *Atmos. Environ.* **2006**, *40*, 2288−2300.
(21) U.S. EPA. The Green Book Nonattainment Areas for Criteria Pollutants. http//www.epa.gov/air/oaqps/greenbk/ (Accessed Febr. 11, 2014).
(22) Pope, C. A., III; Dockery, D. W. Health Effects of Fine Particulate Air Pollution: Lines That Connect. *J. Air Waste Manage. Assoc.* **2012**, *56*, 709−742.
(23) Laden, F.; Neas, L. M.; Dockery, D. W.; Schwartz, J. Association of Fine Particulate Matter from Different Sources with Daily Mortality in Six US Cities. *Environ. Health Perspect.* **2000**, *108*, 941.
(24) Moore, K.; Neugebauer, R.; Lurmann, F.; Hall, J.; Brajer, V.; Alcorn, S.; Tager, I. Ambient Ozone Concentrations Cause Increased Hospitalizations for Asthma in Children: An 18-Year Study in Southern California. *Env. Heal. Perspect* **2008**, *116*, 1063−1070.
(25) Jerrett, M.; Burnett, R. T.; Pope, C. A., III; Ito, K.; Thurston, G.; Krewski, D.; Shi, Y.; Calle, E.; Thun, M. Long-Term Ozone Exposure and Mortality. *N. Engl. J. Med.* **2009**, *360*, 1085−1095.
(26) Thurston, G. D.; Ito, K. Epidemiological Studies of Acute Ozone Exposures and Mortality. *J. Exposure Anal. Environ. Epidemiol.* **2000**, *11*, 286−294.
(27) Hubbell, B. J.; Hallberg, A.; McCubbin, D. R.; Post, E. Health-Related Benefits of Attaining the 8-Hr Ozone Standard. *Environ. Health Perspect.* **2005**, 73−82.
(28) U.S. EPA. *The Benefits and Costs of the Clean Air Act from 1990 to 2020: Summary Report*; Research Triangle Park, NC. Available at: http://www.epa.gov/cleanairactbenefits/feb11/summaryreport.pdf (Accessed September 20, 2018), 2011.
(29) Hall, J. V.; Brajer, V.; Lurmann, F. W.. *The Benefits of Meeting Federal Clean Air Standards in the South Coast and San Joaquin Valley Air Basins*; California State University 2008.
(30) Hall, J. V.; Brajer, V.; Lurmann, F. W. Air Pollution, Health and Economic Benefits—Lessons from 20 Years of Analysis. *Ecol. Econ.* **2010**, *69*, 2590−2597.
(31) Collet, S.; Kidokoro, T.; Sonoda, Y.; Lohman, K.; Karamchandani, P.; Chen, S.-Y.; Minoura, H. Air Quality Impacts of Motor Vehicle Emissions in the South Coast Air Basin: Current versus More Stringent Control Scenario. *Atmos. Environ.* **2012**, *47*, 236−240.
(32) Vijayaraghavan, K.; Lindhjem, C.; DenBleyker, A.; Nopmongcol, U.; Grant, J.; Tai, E.; Yarwood, G. Effects of Light Duty Gasoline Vehicle Emission Standards in the United States on Ozone and Particulate Matter. *Atmos. Environ.* **2012**, *60*, 109−120.
(33) Vijayaraghavan, K.; Lindhjem, C.; Koo, B.; DenBleyker, A.; Tai, E.; Shah, T.; Alvarez, Y.; Yarwood, G. Source Apportionment of Emissions from Light-Duty Gasoline Vehicles and Other Sources in the United States for Ozone and Particulate Matter. *J. Air Waste Manage. Assoc.* **2016**, *66*, 98−119.
(34) Collet, S.; Minoura, H.; Kidokoro, T.; Sonoda, Y.; Kinugasa, Y.; Karamchandani, P. Evaluation of Light-Duty Vehicle Mobile Source Regulations on Ozone Concentration Trends in 2018 and 2030 in the Western and Eastern United States. *J. Air Waste Manage. Assoc.* **2013**, *64*, 175−183.

(35) California Air Resources Board. EMFAC2017 Volume III - Technical Documentation; 2018.
(36) CARB. Current CARB Emissions Inventory (ver. 2012).
(37) EPA, U. S. SMOKE v4.5 User's Manual. Office of Air Quality Planning and Standards, Air Quality Assessment Division; 2017.
(38) Carter, W. P. L. Development of the SAPRC-07 Chemical Mechanism. *Atmos. Environ.* **2010**, *44*, 5324−5335.
(39) Guenther, a.; Karl, T.; Harley, P.; Wiedinmyer, C.; Palmer, P. I.; Geron, C. Estimates of Global Terrestrial Isoprene Emissions Using MEGAN (Model of Emissions of Gases and Aerosols from Nature). *Atmos. Chem. Phys.* **2006**, *6*, 107−173.
(40) US EPA Office of Research and Development. CMAQ (Version 5.2). *Zenodo* 2017.
(41) Carreras-Sospedra, M.; Williams, R.; Dabdub, D. Assessment of the Emissions and Air Quality Impacts of Biomass and Biogas Use in California. *J. Air Waste Manage. Assoc.* **2016**, *66*, 134−150.
(42) Zhu, S.; Horne, J. R.; Montoya-Aguilera, J.; Hinks, M. L.; Nizkorodov, S. A.; Dabdub, D. Modeling Reactive Ammonia Uptake by Secondary Organic Aerosol in CMAQ: Application to the Continental US. *Atmos. Chem. Phys.* **2018**, *18*, 3641.
(43) Mac Kinnon, M.; Shaffer, B.; Carreras-Sospedra, M.; Dabdub, D.; Samuelsen, G. S.; Brouwer, J. Air Quality Impacts of Fuel Cell Electric Hydrogen Vehicles with High Levels of Renewable Power Generation. *Int. J. Hydrogen Energy* **2016**, *41*, 16592−16603.
(44) Appel, K. W.; Pouliot, G. A.; Simon, H.; Sarwar, G.; Pye, H. O. T.; Napelenok, S. L.; Akhtar, F.; Roselle, S. J. Evaluation of Dust and Trace Metal Estimates from the Community Multiscale Air Quality (CMAQ) Model Version 5.0. *Geosci. Model Dev.* **2013**, *6*, 883−899.
(45) Murphy, B. N.; Woody, M. C.; Jimenez, J. L.; Carlton, A. M. G.; Hayes, P. L.; Liu, S.; Ng, N. L.; Russell, L. M.; Setyan, A.; Xu, L.; Young, J.; Zaveri, R. A.; Zhang, Q.; Pye, H. O. T.; Semivolatile, P. O. A. Parameterized Total Combustion SOA in CMAQv5.2: Impacts on Source Strength and Partitioning. *Atmos. Chem. Phys.* **2017**, *17*, 11107−11133.
(46) U.S. EPA. *Nonattainment Areas for Criteria Pollutants (Green Book)*.
(47) Emmons, S. E. F.; Emmons, L. K.; Walters, S.; Hess, P. G.; Lamarque, J.-F.; Pfister, G. G.; Fillmore, D.; Granier, C.; Guenther, A.; Kinnison, D.; Laepple, T.; Orlando, J.; Tie, X.; Tyndall, G.; Wiedinmyer, C.; Baughcum, S. L.; Kloster, S. Description and Evaluation of the Model for Ozone and Related Chemical Tracers, Version 4 (MOZART-4). *Geosci. Model Dev.* **2010**, *3*, 43−67.
(48) Skamarock, W. C.; Klemp, J. B.; Dudhi, J.; Gill, D. O.; Barker, D. M.; Duda, M. G.; Huang, X.-Y.; Wang, W.; Powers, J. G. A. Description of the Advanced Research WRF Version 3. *Tech. Rep.* **2008**, 113.
(49) Benosa, G.; Akshay, A.; Zhu, S.; Mac Kinnon, M.; Dabdub, D. T. Air Quality Impacts of Implementing Emission Reduction Strategies at Southern California Airports. *Atmos. Environ.* **2018**, *185*, 121−127.
(50) Mac Kinnon, M.; Zhu, S.; Carreras-Sospedra, M.; Soukup, J. V.; Dabdub, D.; Samuelsen, G. S.; Brouwer, J. Considering Future Regional Air Quality Impacts of the Transportation Sector. *Energy Policy* **2019**, *124*, 63−80.
(51) Shen, E.; Oliver, A.; Dabirian, S. *Final Socioeconomic Report ; South Coast Air Quality Management District*. Available at: http://www.aqmd.gov/docs/default-source/clean-air-plans/socioeconomic-analysis/final/sociofinal_030817.pdf?sfvrsn=2 , 2017.
(52) Sacks, J. D.; Lloyd, J. M.; Zhu, Y.; Anderton, J.; Jang, C. J.; Hubbell, B.; Fann, N. The Environmental Benefits Mapping and Analysis Program−Community Edition (BenMAP−CE): A Tool to Estimate the Health and Economic Benefits of Reducing Air Pollution. *Environ. Model. Softw.* **2018**, *104*, 118−129.
(53) Davidson, K.; Hallberg, A.; McCubbin, D.; Hubbell, B. Analysis of PM2. 5 Using the Environmental Benefits Mapping and Analysis Program (BenMAP)Â. *J. Toxicol. Environ. Heal. Part A* **2007**, *70*, 332−346.
(54) ORNL LandScan; Oak Ridge TN: Oak Ridge National Laboratory. Available at: http://web.ornl.gov/sci/landscan/, 2016.
(55) Industrial Economics. *Review of Baseline Incidence Rate Estimates for Use in 2016 Socioeconomic Assessment*; Memorandum. Massachusetts, MA: Industrial Economics, Inc Available at: http://www.aqmd.gov/docs/default-source/clean-air-plans/socioeconomic-analysis/iecmemos_november2016/scbaselineincidence_112916.pdf , 2016.
(56) Industrial Economics. *Literature Review of Air Pollution-Related Health Endpoints and Concetnration-Response Functions for Ozone, Nitrogen Dioxide, and Sulfur Dioxide: Results and Recommendations*; Memorandum. Cambridge, MA: Industrial Economics, Inc. Available at: http://www.aqmd.gov/docs/default-source/clean-air-plans/socioeconomic-analysis/iec_gasplitreview_092916.pdf , 2016.
(57) Industrial Economics and Lisa Robinson. *Review of Mortality Risk Reduction Valuation Estimates for 2016 Socioeconomic Assessment*; Memorandum. Massachusetts, MA: Industrial Economics, Inc. Available: http://www.aqmd.gov/docs/default-source/clean-air-plans/socioeconomic-analysis/iecmemos_november2016/evaluationcriteria_113016.pdf , 2016.
(58) Haagen-Smit, A. J. *Carbon Monoxide Levels in City Driving*. Taylor & Francis 1966.
(59) California Department of Transportation. Historical Monthly Vehicle Miles of Travel.

https://dx.doi.org/10.1021/acs.est.0c04060
Environ. Sci. Technol. 2021, 55, 547−552